Monitoring Tailor-made II

Proceedings

An international workshop on information strategies in water management

Edited by

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PROCEEDINGS

Monitoring Tailor-made II

information strategies in water management

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PREFACE

Monitoring tailor-made has become a concept in water management. Decades ago, water monitoring used to reflect periodical measurements of single, simple variables and of flow. Nowadays, monitoring is the major tool in gathering information to tackle changing environmental issues in surface and groundwater management. Not only environmental issues have changed, but also the information needed - to make (management) decisions or to take measures - changed and it becomes more complex to collect the right information. Therefore our monitoring activities comprise more than merely sampling water, measuring flow or performing laboratory analysis. Monitoring comprises a chain of successive activities: specification of information needs, development of monitoring strategies, network design, sampling, laboratory analyses, assessment, reporting and information utilisation. Water monitoring is therefore an intrinsic part of water management and it has become the working field of a growing number of specialists.

The "Monitoring Tailor-made" concept started its life, when experiences and ideas from various groups in the field of monitoring were brought together in an international workshop "Monitoring Tailor-made I" , in September 1994, at Beekbergen, the Netherlands. Driving forces behind this workshop were experts from The Colorado State University, Fort Collins, USA; from the Water Transport Company WRK and from the RIZA. The leading principle was that (re)design of monitoring networks should be based on and tuned to specific and well defined information needs, in other words: a tailor-made approach. Their attention was mostly focused on the specification of information needs, the first step in the above mentioned chain of successive monitoring activities, the monitoring cycle. They also started the development of a more general framework for this tailor-made approach in order to make it applicable for transboundary waters in order to meet the needs of international organisations as the United Nations Economic Commission for Europe, UN/ECE and the European Environment Agency, EEA.

From the experiences with MTM-I, we learned that it is not an easy task we have set ourselves. Ample experience in tuning the objectives of our monitoring programmes towards water management needs is not yet present. And there is no such thing as a recognised general framework, which can be used to implement the tailor-made approach for every problem or region. However, we did learn from comparing all those different monitoring strategies and programmes. We also learned to decrease the number of monitored ingredients and similarly to increase the information content. Yet, we still had to find ways/methods to better fill out, in a sound way, all the successive steps of the monitoring cycle. From this need, Monitoring Tailor-made II evolved.

This time an even larger number of groups/institutions were involved in the preparation, as can be derived from the composition of the Scientific Committee. The subjects to be addressed included: integrated monitoring, indicator use, cost-effectiveness and indicative methods, and recommendations on "Best Available Practices" or guidelines. Integrated monitoring comprises a variety of different, relatively new subjects as: integration of surface water and groundwater monitoring, integration of water quality and quantity monitoring, integration of different monitoring approaches; e.g. biological with chemical monitoring, and integration of different disciplines, that are included in the different steps of the monitoring cycle; e.g. water managers, policy makers and network designers. Most of these subjects have not yet been implemented in monitoring practice, therefore the major principles of these subjects were to be discussed and presented at Monitoring Tailor-made II.

We did not want to be over ambitious and we focused on the upper part of the monitoring cycle: water management, specification of information needs, monitoring strategy and further assessment, information utilisation and water management, to complete the cycle. In short, we named this approach "information strategies in water management".

With this scope, as compared to MTM-I, an even larger number of specialists participated and

With this scope, as compared to MTM-I, an even larger number of specialists participated and made this workshop "Monitoring Tailor-made II" a truly international, multidisciplinary forum, where ideas and experiences were exchanged. Part of our questions regarding information strategies in water management were answered, but also many more questions were raised. The contributions show many examples of how integrated monitoring can become reality, or how indicative methods can help us to make monitoring more efficient. Also, some building blocks were presented to start the use of indicators in water management. Guidelines and directives were presented.

Again we learned a lot. But also, from the same presentations and discussions, a need evolved for more, a follow-up, to learn how we should proceed with all the starting points presented at this workshop. How can we implement these principles in our monitoring programmes? What will be the result of testing the guidelines in pilot-projects? How can we make way for experts from different disciplines to work together? We would like to have answers on these questions and to have the results presented of putting these principles into practice. Therefore, it is planned to have a Monitoring Tailor-made III in 1999. We hope to meet you there!

From this proceedings a selection of papers has been published in a special issue of European Water Pollution Control in July this year. The papers in this "Monitoring Tailor-made" issue reflect the different monitoring approaches, which are requested when dealing with different problems or regions as is also reflected by the cover photographs. We warmly acknowledge the help and co-operation of Jan van de Kraats, editor-in-chief of this magazine, in finding referees and bringing the results of this workshop under the attention of even more people.

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CONTENTS

Posters - surface waters

OPENING AND INTRODUCTION

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Mr Chairman, ladies and gentlemen,
Welcome to the second international workshop Monitoring Tailor-made. Over 30 countries are represented here, so we can really speak of an international event.

WATER IS ESSENTIAL

The subject of our workshop is monitoring for water management. Water is essential. Without water, life as we know it would not exist. Nevertheless, good quality fresh water is scarce. Fresh water is needed for drinking, for hygiene, for agriculture, for industry and with a growing world population the need is growing (Saeijs, 1995). The increasing use of fresh water resources threatens the ecological functioning of the fresh water systems.

For several countries, rivers and groundwater reservoirs are the main sources of fresh water. They are highly depending upon other, upstream countries. Major disagreements may arise between countries over this issue. This is one of the reasons why an international agreement, such as the Helsinki Convention on the Protection and Use of Transboundary Watercourses and International Lakes, is so important.
This agreement arranges for the prevention, control ánd reduction of waterpollution in transboundary water systems and for the assurance that water resources are used in a reasonable and equitable way. It also ensures the protection or restoration of aquatic ecosystems at local, national and transboundary levels. To be able to do so, proper information on the status of the water systems is indispensable. Good monitoring strategies will therefore become more and more important.

OUTCOME OF THE FIRST WORKSHOP MONITORING TAILOR-MADE

As some of you will remember, two years ago, the first workshop Monitoring Tailor-made was held. During this workshop it was concluded that a better fit should be found between the data produced by the monitoring networks and the information that is necessary for management actions and future developments in water *management* policy.
This notion is visualised in the monitoring cycle, where monitoring is regarded as a sequence of related activities that ultimately leads to management decisions based on all relevant information (Adriaanse et al., 1994). The cycle represents the repeating evaluation and adaptation of the monitoring network. Monitoring programmes are most effective when they are 'tailor-made'. As the body of the information need grows, or looses weight, the monitoring programme should regularly be tailored to keep a perfect fit.

Another conclusion of the workshop was that monitoring and assessment becomes increasingly complex. It is not enough anymore that the specialists in different disciplines, physicists, chemists and biologists, collect an enormous amount of data and report these separately. Incre-

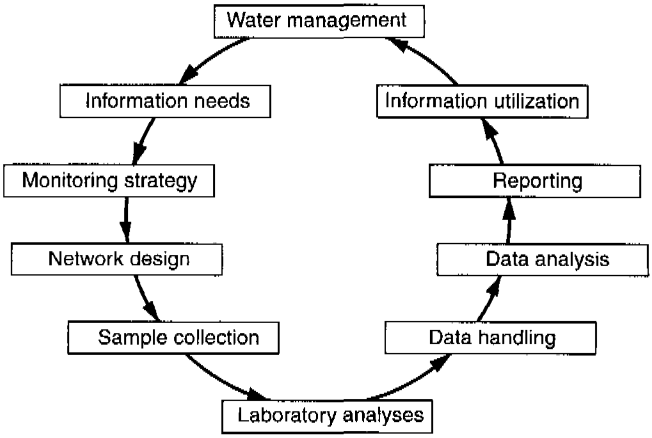

Figure 1: The monitoring cycle

asingly, water is being viewed in an integrated manner. A systematic analysis and assessment of water quality, flow regimes and water levels, habitats, biological communities, sources and fate of pollutants, as well as mass balances, should be conducted in order to provide reliable and useful information (UN/ECE, 1996). Only with this integrated information, a good understanding of the situation of the aquatic system at hand will be possible.

As the borders of the ecosystem are independent of national administrative borders, international co-operation becomes increasingly important. We observe that more and more countries come to agreements on joint monitoring and assessment activities. Water management becomes river basin management, taking into account all the riparian countries' interests.

Next to this, the growing knowledge of the processes in the aquatic environment leads to the understanding that the existing monitoring networks provide limited information. The development of the aquatic system is not always explained by the determinands that are measured. On the other hand it is not possible to measure all possibly influencing substances, simply because there are too many of them.

This growing knowledge also leads to the need for more comprehensive information. But as budgets are limited, all this need for extra information can only be met by more efficient and cost-effective methods.

Finally it was concluded that there is a growing public consciousness of environmental issues. This leads to the demand for more and better information and it is up to us to provide this in a comprehensible way.
We left the workshop with many new ideas and the challenge to develop a highly integrated and cost-effective, optimised monitoring network.

Now we are two years later. Presumably you all have individually made progress in improving your monitoring. In this workshop you will exchange your experiences and develop new ideas that should again help us some steps further in our quest for the ideal monitoring programme.

THE IMPORTANCE OF GROUNDWATER

Integrated water management is concerned with both groundwater and surface water systems. The monitoring of these waters can be quite different due to their different properties and the

human use of the system (Claessen, 1996). However, they are both part of the same hydrological cycle and especially the 'information'-parts of the monitoring cycle, -such as the definition of information needs and the translation of monitoring results in the desired information-, are relatively similar.
To be able to set up a true integrated monitoring of our water systems, one has to learn not only more about surface water and groundwater monitoring, but also about specific processes, by which they interrelate. For this reason we will pay special attention to the specific aspects of groundwater monitoring and its relation to surface water monitoring in this workshop.

OPTIMISATION OF THE MONITORING CYCLE

The growing need to integrate disciplines, but also the need to harmonise between countries and to comply with international agreements put a heavy burden on the monitoring networks. How can we be certain that we can provide all the necessary information? How can we be certain that the data really provides for the information needed? Should we collect more data? How can we do this without the needed capacity, without the needed finances? How can we escape this rat-race for more?

There are several possible answers to these questions.
One idea is that there should be an equilibrium between the goal of the information and the costs to obtain it. Monitoring networks should be cost-effective. We should not go for the best possible if less is sufficient. By looking closer at the 'specification of the information need', the first activity in the monitoring cycle, and its counterpart, 'information utilisation', which is the final step, the attention may be focused on the development of environmental indicators or indices, which can provide valuable information, without adding more measurements. By optimising the information parts of the monitoring cycle, monitoring can become more effective. With the use of indicators, policy objectives will become more clear and transparent, thus increasing public support. The clear specification of information needs will help to balance profits and costs of the monitoring network.

Another solution is to look for methods that provide much information against little cost. Toxicological tests, for instance, may provide information on the effects of the numerous substances in the water that are not measured individually. Development of indicative methods will reduce the number of specific analyses. These methods may not give precise information, but will act as a lever to start an inquiry whenever there is an indication that something is wrong.

The use of models is a third way of contributing to the reduction of data-gathering. They may be used to predict flow or concentrations, thus reducing the number of sampling locations or sampling frequencies. Models may also be used to help the designer of the monitoring network to optimise it by showing where impacts can be expected and under what conditions.

INTERNATIONAL CO-OPERATION

Water management is not a sole matter of countries. Rivers and groundwater resources do not stop at national borders. Therefore, international co-operation is essential. What is needed is a basis for this co-operation. The Helsinki convention is such a basis, as are collective alliances like the European Community, but the point is to make these effective. Guidelines, directives, or codes of practice are needed to implement the co-operation. But since issues and priorities to solve these matters tend to vary per country, guidelines or directives cannot be too specific. For instance, the guidelines on water-quality monitoring and assessment of transboundary rivers, that have been drawn up by the UN/ECE Task Force on monitoring and assessment under the Helsinki convention and that recently have been adopted by the UN/ECE Committee on Environmental Problems, describe a strategy for developing joint monitoring programmes, leaving

details to be filled in by the riparian countries. Under the EC legislation, a Framework Directive is under development, in which regional variation is accounted for. Also, the activities of the European Topic Centre on Inland Waters provide significant starting points to strengthen this strategy. In my opinion the work of these different organisations has to be complementary and cooperation of these organisations is imperative for international water management.
We will discuss frameworks to support international co-operation in this workshop. In the mean time, the challenge for riparian countries is to find their mutual interests.

As international co-operation becomes more important, the importance of quality assurance is increasing.
As the monitoring cycle shows, it is a chain of activities. The strength of this chain is determined by its weakest link. From this viewpoint, quality assurance of the monitoring and assessment process as a whole is fundamental for an effective monitoring network. In this respect, quality assurance may be a basis for successful co-operation and joint monitoring.

As we have seen, there is no single solution, but many ways to meet our challenge. Clearly specified information-needs, the use of indicative methods and models will contribute to an integrated, cost-effective assessment as part of a truly optimised monitoring network. This monitoring network will be the key to assure that fresh water may remain available for a growing and developing world population and still leaves enough room for valuable ecosystems.

THE THEME OF THIS WORKSHOP

Essentially, monitoring comes down to these two questions: what information do we need and how can we get this information to ascertain proper water management. And this brings us to the theme of this workshop: 'information strategies in water management'.

In the coming days you will again look at the need to integrate information on aquatic systems and discuss the ways to do so. You will study the data management and data analysis and means to present the information in a comprehensible way. You will be informed on indicators and indicative methods that recently have been developed. You will learn about the use of models and the development of guidelines. Experiences with national and international information strategies will be brought before us. This all should bring us closer to integrated and cost-effective, optimised monitoring networks.

I hope and expect that this workshop will be an opportunity for you to openly discuss the growing importance of tailor-made monitoring and assessment and that you may all learn from each others' experiences. I wish you interesting, fruitful and, not in the least, very enjoyable days here in Nunspeet.

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MONITORING PROGRESS TOWARD "SUSTAINABLE DEVELOPMENT": IMPLICATIONS FOR WATER QUALITY MONITORING

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ABSTRACT

As "sustainable development" increasingly becomes the goal of environmental management efforts, the regulatory-driven, media-based environmental management approaches employed in the past must be modified. Nowhere will this modification be more evident than with "water quality monitoring" - the traditional information system for regulatory water quality management. While some aspects of traditional water quality monitoring will remain the same, there will be a need for "indicators" of environmental and economic health that can be placed side-by-side in a meaningful (i.e. sustainable development) context. There will be a need to distribute data via computer technology in ways that promote its analysis and use (and value) by segments of society that have not had access to the data before. There will be a need to combine existing data sets in creative ways to produce more integrated information for lay audiences. There will also be a need to "standardize" water quality data handling, analysis, interpretation and reporting in order to facilitate ready consumption by the public. This evolution of water quality monitoring efforts toward more emphasis on data analysis, interpretation and reporting, while not unlike evolutionary steps taken in the past coordinating and standardizing sampling and laboratory analysis, will require water quality monitoring achieve sustainable development.

INTRODUCTION

Media-based (particularly water and soil) environmental management has been conceptualized and implemented in a variety of formats around the world. In the late 1800s and early 1900s, natural resource management was organized within a number of government agencies in the United States (agricultural departments, forestry agencies, soil conservation districts, and water development bureaus). A large number of academic disciplines and professional societies have evolved around this division of the environment. Each of the government departments and disciplines has evolved their own means of obtaining information relative to the resource being managed. For example, a fish management agency will obtain water quality information relative to operating and maintaining an optimum sport fishery. The goal of most of these early natural resource management efforts was maximum benefit of the resources being managed to the most people.

Before World War II and just after the war, the need to control pollution, both industrial and municipal, had reached the point where national coordination of pollution control efforts was being legislated. The resulting implementation was often weak, but the concept of nationally driven pollution control was being recognized and accepted. Monitoring efforts at this time were focused mainly on solving specific problems as they arose.

During the 1950s and 1960s, industrial development surged at a rapid rate causing a number of flagrant environmental abuses. In the 1960s laws and water quality management agencies were established to more aggressively reduce the excessive pollution from industries and cities using proactive management tools, such as legally enforceable water quality standards and discharge permits. Monitoring was used to check compliance and support enforcement actions. Some monitoring efforts were devoted to assessing general water quality conditions, but these were not well developed (Ward, 1995a).

Over the three decades since these initial water quality management efforts were undertaken, it has become obvious that management of one environmental medium, such as water, can have unintended consequences to other media, such as air and soil. The tendency of government to manage the environment in well-defined components, permitted some water quality problems to avoid regulatory attention. Perhaps the best example of this is nonpoint source pollution, which often results from the actions of many individuals rather than one large company. These sources of pollution do not lend themselves to the "command and control" philosophy underpinning existing water quality regulatory approaches. In addition, a strict regulatory approach to environmental management is often seen as counter to the economic well-being of citizens.

Thus, in the 1980s there were again considerable discussion and conceptualization of the need for a new approach to environmental management. The resulting approach is generally referred to as "sustainable development", although other terms such as "integrated watershed management" and "ecosystem management" are also being used to express many of the concepts involved. This more integrated view of the world, and human activities in it, was articulated in the Brundtland Report and highlighted and, generally, accepted by many nations of the world at the Earth Summit in Rio de Janeiro in 1992.

The recently published U.S. report from the President's Council on Sustainable Development (entitled Sustainable America) [1996], notes the need to move environmental management toward a more cost-effective system based on performance, flexibility and accountability. Government agencies are encouraged to "create more flexible, cost-effective approaches to attain the human and ecosystem health goals of existing programs while maintaining monitoring and verification functions." This philosophy has also been articulated in a primer for watershed assessment issued by the U.S. EPA (1994).

This more integrated view of environmental management is being used by some countries to completely overhaul the government structure employed to manage their environment. New Zealand, to illustrate, passed the Resource Management Act of 1991 which created elected Regional Councils to manage air, land and water within a watershed. The Regional Councils were given the power to tax and contract with the existing national, media-based agencies to perform needed work. The national agencies obtain their justification, in many ways, for continued existence from the contracts they perform for the Regional Councils. Palmer (1995) gives an excellent review of the thinking and actions that led to this overhaul of New Zealand's approach to environmental management.

The report Sustainable America, while not recommending as dramatic a change as was implemented in New Zealand, is attempting to articulate what sustainable development should mean to the United States. Scores of wide-ranging recommendations, and actions to implement them, are included in the report. The report speaks in terms of environmental and economic goals and ways to measure progress toward their attainment.

The purpose of this paper is to examine the move toward "a new way of thinking" (advocated in the report Sustainable America) about integrated environmental and economic management and examine the ramifications to water quality monitoring. First, the role of water quality monitoring within environmental management will be reviewed. Second, implications of the management changes on monitoring programs will be discussed.

WATER QUALITY MONITORING

Water quality monitoring has, historically, been focused on solving water quality problems, not reporting on the general status of environmental health, from either the ecosystem or human health perspectives. This focus stemmed from the legal mission assigned to the various resource management and environmental protection agencies. In meeting legal mandates, a considerable amount of monitoring performed within a water quality management organization is associated with setting standards, discharge limits, and checking compliance. In recent years there have been efforts to collect and analyze regulatory data to assess general water quality conditions. In the U.S. the best example of this effort is described in the guidelines used to produce biannual water quality assessments, also referred to as 305(b) reports (US-EPA, 1995a). However, it is only recently, in the U.S. for example, that a national effort has been devoted solely to the purpose of assessing national water quality status and trends. The initial planning for this National Water Quality Assessment (NAWQA) program are described by Hirsch et al. (1988).

As the need to shift water quality management efforts away from strictly point sources of pollution control to nonpoint source sources is recognized, the limits of the current reliance on discharge permits and compliance monitoring are quickly exposed. The control of pollution generated by individuals is beyond the current management structure. This realization has put added pressure on today's water quality management programs to educate the public about their individual contributions to water quality degradation. It has also led to many of the calls for a more integrated approach to water quality management with sustainable development goals.

Some form of "regulatory" monitoring will continue to be necessary, but the new goal of sustainable development, as noted above, places much more emphasis on the need for water quality information to inform and, in the process, educate the public. More specifically, such information needs to describe, for example, the health of the aquatic ecosystem, the impact of citizens' individual (and collective) actions on water quality, the quality of public water supplies, the safety of agricultural practices, as well as the overall status and trends in water quality conditions in the nation and community. To provide this broader view of water quality, from more of a watershed perspective, will require water quality monitoring professionals view their purpose and tasks differently.

No longer will a monitoring program be restricted to producing data for implementing, for example, the Safe Drinking Water Act or the Clean Water Act. The collected data, when combined with other water quality data sets and with corresponding ecological and economic data, have the ability to "draw" much larger pictures of environmental/ecological quality and human impact on that quality. Water quality data, today, are, in many ways, just "lying about" waiting to be combined, analyzed and interpreted in more meaningful and relevant ways for the public.
In fact, there is a patchwork of environmental, economic and social data sets, collected from narrow mission-oriented agencies, which can be combined using new information technologies (e.g., GIS) to draw pictures of sustainable development. The U.S. Environmental Protection Agency's recent report on environmental indicators (US-EPA, 1996) illustrates the variety of data sets that can be combined to form a suite of indicators. Private companies are recognizing the economic profit associated with combining existing, government-generated natural resource data sets into information that can be sold in the environmental marketplace.

Cities are recognizing the ability to describe their collective "footprint" on the water quantity and quality of their watershed by comparing the data currently collected at their water treatment plant with the data collected at their wastewater treatment plant. These data sets are generally created under different laws to meet different legal mandates, but it is possible to "mine" this data for additional information.

No one person or agency is charged to take the data from the different water quality monitoring programs, as well as the ecological and economic data collection efforts, and analyze and interpret the results in an integrated fashion. Unfortunately, when there is a call for sustainable development as a guiding principle for environmental management, there are rarely identified the expertise, resources and responsibility (legal or agency) to bring all the data together and begin to draw the larger pictures and better direct the "mission" oriented agencies to act in a more sustainable development manner. This task, in large measure, will fall upon those employed in water quality monitoring, acting in new, expanded roles. The Intergovernmental Task Force (ITFM) has attempted to integrate the monitoring efforts of approximately twenty agencies in the U.S. (1995).

IMPLICATIONS TO WATER QUALITY MONITORING

The move to a more integrated approach to environmental management will impact water quality monitoring most directly, initially, in how existing data are analyzed, interpreted and reported to the public. This point was highlighted in the Sustainable America report with the following statement:

"Government already has collected an abundance of information, but often it is not available to policymakers or the public in a form they can use."
Monitoring professionals will be asked to create more publicly relevant information from existing water quality data. This will require monitoring professionals assuming a lead role in adding data analysis, interpretation and reporting, for public consumption, to their existing monitoring system operations. The resources (computer systems and data analysis software) and skills (statistical, public relations and GIS) required to do this will have to be increased in many monitoring systems operations where the historical focus has been on simply collecting data.

Beyond "adding public value" to their existing data, monitoring systems must also make existing data itself readily available to people and organizations outside the immediate agency. Again, using the words of the Sustainable America, report, it was recommended that environmental management organizations (and, by implication, their monitoring programs):

"Adopt open information policies and practices, recognizing that disclosure and active dissemination of information should be the rule, not the exception. Adopt policies that increase access to public information for all segments of society and encourage the development of the national Information Infrastructure by the private sector in ways that improve access for all."

Providing water quality data and information to the public, and other agencies, permits water quality data to be used by people outside water quality management to develop pictures of sustainable development. These efforts should not be stymied by the inability to access water quality data. Such efforts, in the past, have not been viewed positively by many agencies because of the opportunity of the public or other agencies to misuse water quality data. Unfortunately, such an attitude tends to place a barrier to adding value to an agency's water quality monitoring effort. If the monitoring program's results cannot be readily accessed, the monitoring effort's value is hidden from the public. It appears the monitoring programs that will continue to exist into the future are those that not only collect and use data to accomplish their legally and administratively assigned missions, but also make their data readily accessible to anyone. The U.S. Geological Survey is an example of an agency that is making its data easily accessible via the internet.

Water quality data, due to the many constituents and resulting large volumes of numbers, can be overwhelming to analyze, interpret and understand. The complexity of water quality data can make the evaluation of progress toward national water quality goals difficult to assess. There is a rapidly evolving need to identify "indicators" of water quality in order to explain water quality and related ecosystem and economic characteristics to the public and their elected representatives. The Sustainable America report recommended that Federal government should:

"Develop indicators of progress toward national sustainable development goals and regularly report on these indicators to the public."

Indicators can take on many forms, from an index of water quality variables to a separate measurement, such as a highly representative fish species whose presence or absence indicates the overall health of an ecosystem. Indicators have been used for many years in water quality monitoring. Perhaps the most widely recognized is the organism Escherichia.coli, which has been used for many years as an indicator of fecal contamination of water.

There are no widely accepted indicators of water quality conditions that are regularly produced and disseminated by water quality monitoring programs. This lack of agreement among water quality monitoring professionals tends to leave the development of water quality indicators to broad-based environmental managers and public relations specialists. Will the absence of broad agreement, by water quality monitoring professionals, as to what constitutes a water quality indicator, permit unreasonable information expectations to be demanded by those who do develop environmental indicators for sustainable development accountability? This is not a new problem. A paper presented at the 1970 National Symposium on Data and Instrumentation for Water Quality Management, held in Madison, Wisconsin, was titled: "A Water Quality Index - Do We Dare?" (Brown et al., 1970). This phrase captures the historically timid nature of many water quality monitoring professionals regarding water quality indicators. The recent indicator report from the U.S. EPA (1996) suggests renewed efforts to address this critical topic in the U.S.

When water quality indicators are developed and data are made readily available, how can the data be analyzed, interpreted and reported in a manner readily understood by the public? At present, there is no "Bureau of Environmental Statistics" available to do this. There are no agreed upon data analysis protocols available. There are no widely recognized environmental indicators to inform the public about the general health of their environment like the Dow Jones Index informs people about the general health of the U.S. stock market. There is not even, in many monitoring programs, a recognition of the need to staff and fund data handling, data analysis, interpretation and reporting as an ongoing function of the monitoring program (Ward, 1995b).

Just as standardization of economic data handling, analysis, and reporting (via indices) facilitates public understanding of the behavior of our national and world economies, the standardization of water quality data handling, data analysis, interpretation of the findings and reporting (via indices) are needed to move the "state-of-the-art" of water quality monitoring into the forefront of public understanding. This will be particularly critical to water quality taking its proper place in assessing the environmental status and trends associated with sustainable development. Standardization of water quality data analysis, interpretation and reporting will happen, however, only when enough water quality monitoring professionals recognize its value in the operation and accountability of their programs. Such was the case when sample collection and laboratory methods standardization was recognized and strong efforts made to coordinate use of common methods. The Monitoring Tailor-made workshops increase recognition of the need for monitoring professionals to go beyond standardization of just sample collection and laboratory methods to coordination of the entire information acquisition effort.

The increasing focus on data analysis, interpretation and reporting methodologies will, in time, force water quality monitoring managers to carefully examine their distribution of staff and equipment between data collection and data analysis. The future water quality monitoring organization may see as much as 30 to 40 percent of its human and financial resources being devoted to accessing, generating and reporting information from data it, and others, collect.

With limited amounts of data analysis and reporting expertise in monitoring today, there is some concern that water quality monitoring professionals are not in a position, currently, to be active players in the move toward more State of the Environment and Sustainable Development reporting. Water quality monitoring professionals seem to be inhibited by the scientific method and/or the political nature of water quality information when it comes time to make publicly relevant statements about the quality of our water. Many monitoring professionals, like scientists, have a way of writing that tends to hold them back from making conclusive statements about the status and trends in the environment. The volunteer monitoring efforts rapidly developing around the world indicate the need to put the findings of monitoring into more publicly oriented formats and statements (The Volunteer Monitor, 1995).

CONCLUSIONS

New environmental management goals (i.e. sustainable development) will require water quality monitoring programs to produce more publicly relevant information, share data, and provide indicators of environmental status and trends. To undertake such activities will require an expansion of the personnel and financial resources devoted to data analysis, interpretation and reporting. To provide the data to drive this growing desire for information regarding sustainable development will require water quality monitoring programs to cooperate more closely in the acquisition and sharing of data.

Water quality monitoring has evolved rapidly over the past thirty years and now faces even more change as environmental management goals expand to integrate economic and ecological goals. As a result of the growing demand for more information about sustainable development, water quality monitoring, if it can adapt to the changing information expectations being placed upon it, has a critical role to play in the future of environmental management. Are the professionals managing today's water quality monitoring programs able to see this future role and are they willing to undertake the challenge?

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MATCHING WATER QUALITY PROGRAMS TO MANAGEMENT NEEDS IN DEVELOPING COUNTRIES: THE CHALLENGE OF PROGRAM MODERNIZATION

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ABSTRACT

Developing countries face an array of traditional and modern water quality problems - from faecal contamination to toxic chemicals. Moreover, they do so in an economic environment that is severely restricted, an institutional environment which is often poorly structured, and for which the modern scientific knowledge base is frequently poorly understood and applied. Agencies in many developing countries recognise this as a major impediment to sustainable development, especially as water quality has become one of the leading economic issues for the purposes of development and investment (Matthews, 1994). Water quality programs tend to suffer from traditional approaches, both of methodology and legal/administrative. The consequence is that many water quality programs are grossly inefficient, produce often unreliable data and which in any case are not generally useful for making management or investment decisions, and face decreasing economic and political support. Program modernization is essential to achieve the twin goals of greater efficiency and greater relevance in meeting data needs for contemporary water quality management purposes. Modernization reduces costs, may reduce the amount of equipment and infrastructure required, often reduces the amount of data collected, and more closely matches the abilities of developing countries where, for example, knowledge of advanced environmental chemistry may be limited, but where knowledge of biological systems is strong.

INTRODUCTION

The developed world has had the luxury of facing major problems of environmental degradation sequentially in time - faecal contamination and water supply, eutrophication, acid rain, toxic chemicals, ecosystem dysfunction, etc. In contrast, developing countries are facing these simultaneously. Moreover, they do so in an economic environment that is severely restricted, an institutional environment which is often poorly structured and for which the modern scientific knowledge base is frequently poorly understood and applied. Agencies in many developing countries recognise this as a major impediment to sustainable development, especially as water quality has become one of the leading economic issues for the purposes of development and investment.

Typically, the water quality data programs of developing countries fall into one of two categories: (a) programs that focus on "traditional" parameters such as major ions, nutrients and microbiology; and (b) those that attempt to include some toxic chemicals (metals and some organic contaminants such as organochlorine (OC) pesticides and PCB). Additionally, those rapidly developing countries with concerns for toxic organic and inorganic chemistry almost always adopt the "chemical list" approach to monitoring. This is equivalent to the 1970s "Priority Pollutant" approach of the US-EPA which has been demonstrated to be inefficient and costly in the USA and which is now producing inefficiency and rigidity in developing countries just at the time when the USA and other western countries are adopting much cheaper, more flexible, and more effective alternatives.

A common observation of water quality programs is that they tend to be inefficient, the data are of uncertain reliability, program objectives are poorly linked to management needs for data, the analytical technology is often old and inefficient, the focus is on water chemistry even though water is known to be a poor monitoring medium for many toxic chemicals, and databases are incapable of mobilization for management purposes (Ongley, 1993). The concept of program efficiency includes consideration of all these factors, ranging from appropriate selection of parameters and sampling medium, to institutional inefficiency. It has legal and regulatory implications, especially where the regulatory framework imposes rigidity and prevents the use of more cost-effective field and analytical methods. However, the greatest inefficiency tends to lie in the assumption that conventional water quality programs produce data that can be used to make managerial decisions on pollution control, water resources planning, and related investment decisions. The fact is that such programs are designed mainly for descriptive rather than prescriptive purposes, with the result that nations tend to spend much money producing data that are not closely linked to decision-making and, not infrequently, not used at all!

DATA FOR DECISION-MAKING

In developing a monitoring program it is necessary to identify the principal reasons for which monitoring data (chemical and biological) are required. Generally, these are as follows:

a. Description of water quality at the regional or national scale, including determination of trends in time and space.
b. Determination of whether or not water quality meets previously defined water quality or regulatory objectives for designated uses, including public health.
c. Managerial resolution of specific pollution management issues, including post-audit functions.
d. Decisions for investment options based on potential benefits from proposed or alternative remediation options.
e. Comprehensive assessment of river basins, especially to determine the relative importance of point versus nonpoint source pollution sources.
f. Regional or river-basin planning, including the development and audit of national/regional policies on land use relative to polluting land use activities.
g. Reporting of compliance to international standards or action plans. An example is the forthcoming Global Plan of Action for Prevention of Pollution of the Marine Environment from Land Based Sources of Pollution which is likely to focus upon inputs of nutrients, organic micropollutants (=Persistent Organic Pollutants [POPs]) and, possibly, environmental estrogens at the global level, plus metals at local and/or regional levels.

The inability of most national monitoring programs in developing countries to meet many of these objectives requires rethinking of the role and practice of water quality monitoring. There is *no single* type of program that meets all objectives. Therefore, the modernization of water data programs requires a careful evaluation of objectives, a knowledge of alternative methods of achieving the objectives, and the ability to implement change that will enhance efficiency throughout the program (data collection, laboratory operations, data assessment, and mobilization/use for the client).

MODERNIZATION

This paper focuses on the types of considerations that program managers must make to implement program modernization. It is necessarily selective. The background for this material lies mainly in the involvement by the author in program redesign and training in a number of countries - both developed and developing.

A. LEGAL AND INSTITUTIONAL CONSIDERATIONS

Role of Government in Water Data Programs: The role of government in monitoring is being reevaluated in many countries. The old model — government does everything and pays for everything, is being replaced by market forces in many countries. This means that some governments, especially in developed countries, are reducing their direct participation in monitoring and enforcement, and are concerned primarily with setting and enforcing rules and standards. There is no reason why developing countries cannot make the same change. Market forces will produce more efficient laboratories and decrease the need for government expenditures. Under market economic conditions, there are greater efficiency and less cost to government if industrial monitoring is carried out by industry with data reported to government agencies. This mechanism is used by USA, Canada and many European countries. A critical role of government is, however, in developing and enforcing national data standards through programs of quality assurance and laboratory accreditation. National standards cannot be achieved simply by regulating analytical methodology — variation in methods, matrix problems/recovery efficiencies as well as deliberate cost-cutting, operator error, corruption of data files, and outright fraud are part of the realities of the laboratory business. National standards impose a strict regimen on all aspects of laboratory operations and, in some cases, field operations, so to ensure the reliability of the final data products. The present situation in some countries where there is devolution of decision-making authority to local levels, has been the loss of national data standards, which is catastrophic for developing countries that require reliable data for evaluating and deciding upon investment alternatives for remediation and/or development.

Commitment by Management to Change: The rational and comprehensive management of change is very difficult in large organizations. Senior management often resort to budget solutions (reduced budget = reduced number of stations) rather than to a serious examination of how modern monitoring technologies, regulations and institutional structures can introduce greater efficiencies. It is essential that commitment be demonstrated at the most senior levels of an organization. An example of commitment is in the realm of training.

Changing Legal and Regulatory Standards for Greater Efficiency: The use by many countries of rigid legal standards both for the parameters used for regulation and for the types of analyses permitted, is inflexible and inefficient. The US-EPA has now recognized that the rigid codification of analytical standards has not been as effective as once thought, is expensive both for government and industry, and does not provide the flexibility to permit new and more efficient regulatory methods. The alternative, as used in Canada, is "performance based" techniques, which offers simpler and more cost-effective bases for attaining program goals. In "performance based" techniques the method is not rigidly prescribed. However, the outcome must meet predetermined requirements of accuracy and precision. An example is the requirement in some countries to use atomic absorption spectrophotometry (AAS) for metals, whereas new techniques using emission spectroscopy reduces costs by one to two orders of magnitude by its ability to perform multiple simultaneous analyses on a sample.

Maintenance of rigid legal standards ensures that countries will repeat the costly mistakes of many North American and European water quality programs as it will not permit use of new and more efficient methods such as screening methods, and use of toxicity as a regulatory mechanism. In some cases, legal restrictions extend to the requirement to use outdated field methods, which preclude cheaper methods such as portable dissolved oxygen (DO) meters and proxy data such as turbidity measurements instead of direct measurement of suspended sediment concentration. In China, for example, monitoring is restricted to standards and types of analyses identified by Chinese law. Therefore, more efficient and modern methods, especially those of biological screening are difficult or impossible to use for most purposes. The consequence is that, without greater flexibility, monitoring in China is unlikely to be able to modernize significantly (except to build bigger and more expensive laboratories).

B. TECHNICAL

Laboratory Programs: Recognising that there are economic and cultural limitations to issues of efficiency, one of the greatest barriers to modernization is the inefficiency and, in some instances, duplication of government laboratories within small geographical areas. Sample volumes are small and therefore preclude economies of scale that are possible with modern, automated instrumentation. Many laboratories are equipped with sophisticated AAs and GCs, but do not have autosamplers without which volume analysis is impossible and data accuracy jeopardized. In some countries, laboratories having approximately thirty staff handle some two thousand samples (mainly of "low-end" analyses) per year - about what a modern laboratory would process in 2-3 weeks. Gains in efficiency of a factor of x10 are possible and achievable. Gross inefficiency also produces waste in training, equipment procurement and infrastructure costs because of the need to train more operators than are necessary, to purchase equipment that is not efficiently used, and to build and maintain redundant facilities.

The objective of laboratory optimization is faster turnaround time at lower costs per sample of water, sediment and tissue, and for any toxicological tests that may be included in the analytical program. Modernization may involve some or all of the following considerations:

- Sample handling, with special consideration for samples that may be used for legal purposes.
- Turn-around time required
- Reporting requirements
- Sample tracking - level of automation
- Client access to databases
- Types of analyses that are meaningful - totals/filtered for water; sediment and tissue analyses; toxicological testing.
- Detection levels required and appropriate instrumentation.
- Level of QC required for different types of clients
- Type of QA required for lab operations and interlab comparison
- Availability of certified pure reagents
- Personnel available and/or needed, and training required.
- Operating budget and cost-recovery
- Ambient air quality (site limitations)
- Facility infrastructure, including ambient laboratory temperatures and stable electric power supply
- Workplace safety and health

Multiple Techniques Within Monitoring Programs: Most monitoring programs tend to be relatively inflexible and are unable to accommodate many of the new approaches to monitoring. In part this tends to reflect old-fashioned legislation on water quality and, in part, a reluctance on the part of managers to change. Nowadays, monitoring is a comprehensive activity that includes many different approaches to problem solving including fixed site monitoring, field surveys, emergency mobile monitoring, toxicity assessment, and environmental effects monitoring which focuses on in stream biological response to control measures. A useful summary of some of the new procedures is provided in a recent Dutch report by the Ministry of Housing (1995). Biological assessment and toxicity assessment are increasingly being used as part of modern monitoring and enforcement programs and, in some countries, are dramatically reducing the need for expensive chemistry. The interpretation of toxicological tests is now well understood and has at least as much, and usually more, relevance to decision-making as chemical data. Toxicity is used by several Canadian provinces, US states and a number of European jurisdictions for industrial effluent testing as a cheap and effective screening method.

New Screening and Diagnostic "Tools": There now exist many new and cost-effective techniques for water quality monitoring which are more cost-effective, produce more useful information that is linked directly to decision-making, which save time, and have reduced instrumentation needs. These methods follow from the recognition that 1970s type chemical monitoring is not very useful for **managing** rivers and lakes. These "tools" include a wide range of new biological, biochemical, chemical and toxicological techniques. Selection among these many new techniques depends very much on the objectives of the program and the ability to integrate these techniques into the water program. These new tools produce useful information quickly and cheaply, and can be used to define which parameters need further expensive and detailed analysis. They also include methods that are outright substitutes for conventional data programs or which are used to determine whether there is reason to continue with a more detailed and expensive analysis.

As examples, simple inexpensive field kits are available for screening samples for toxicity. Toxicity Evaluation Identification (TIE) is a procedure which permits accurate determination of those chemicals that cause toxicity to selected organisms and which should be added to the monitoring program. Indeed, "TIE" is usually sufficiently precise that it allows identification of a particular industry group that needs to be studied or regulated first. One could completely replace traditional chemical monitoring of large and polluted rivers with a combination of benthic surveying (for DO and nutrient conditions), use of sentinel organisms (for target toxic chemicals), and cheap toxicity testing. The data produced by such a program are inexpensive, and more relevant for **managing** the river, especially as chemical data are not directly related either to human health or to the ecological "health" of the river. Also, it is now widely accepted that toxic chemicals of concern are frequently none of those routinely included in chemical analytical protocols and in many countries and exist at concentrations known to be of concern, but which are below the detection levels used in most laboratories.

Data Quality Objectives: While much of the monitoring in many countries appears to be established according to legal requirements, monitoring stations should begin the use of Data Quality Objectives (DQOs) to ensure greater communication between the monitoring program and those that use the data. DQO's are a process through which the laboratory and the client mutually examine the type of data needed for the management issue at hand, the level of reliability required, reporting requirements, etc. The objective is to ensure that the client understands the costs, the limitations, and the uncertainty in the information produced by the monitoring program. This process is essential when monitoring programs achieve greater flexibility in choice of methods, etc. In a market economy, the DQO process, either formal or informal, becomes the basis for a contract between the monitoring station and the paying client.

Optimization of the National Network: Optimization is a complex and unfamiliar task to many agencies. While optimization may mean reduction in the size of the network and rationalization of the types of parameters, it can also include change in monitoring sites and the addition of more relevant parameters or in limited situations, the expansion of the network to include important unmonitored rivers and lakes. Optimization also includes the detailed analysis of historical data sets in order to identify and eliminate data that do not change or that change in a very predictable way (e.g. annual cycles) or that consistently report NDs (not-detectable). Optimization includes reconsideration of types of data, including use of biological survey methods and measures of toxicity, plus use of sentinel organisms for determining the presence of important industrial parameters.

Information Systems: Database and information systems in most monitoring agencies in developing countries are not efficient and are not effectively used for information processing, analysis and visualization of data, and decision-support functions. This has two types of implications - one is that data are not easily accessible for management purposes. The second is that water quality programs remain largely invisible because of the lack of highly visible data products; the result is often that such programs fail to win managerial and political support.
Modern information systems include: database and data archiving, GIS (Geographical Information Systems), analytical tools (statistics, graphics, etc.), decision-support capabilities, and visualization (output display) capabilities. They should be capable of being operated by nonspecialists with a minimum of training. One such example is Canada's RAISON system (accessed via e-mail <nwri.software@cciw.ca>) which has been adopted by the United Nations GEMS/Water Programme as well as by many water agencies. Great care should be taken to match real GIS requirements with the type of GIS system purchased, especially as GIS is only part of a full information system. Indeed, the GIS requirements of most water agencies are, in fact, quite small (usually limited to handling georeferenced site information, spatial mapping, and limited map overlaying) and which can be met by many inexpensive GIS software packages. Large commercial GIS systems should only be used for specialized GIS activities because the learning requirements are substantial, the hardware and software costs are high, and only specialists can efficiently use such systems. Contrary to GIS vendors, GIS is only one part of a complete information system.
In Mexico, for example, the Mexican Water Commission (Comision Nacional del Agua - CNA) is a partner in the development process of suitable inexpensive software based on the RAISON software platform. The main reasons are the high cost of providing commercial packages (in Spanish) to all the regional offices, and the need to develop simple applications for routine tasks that can be operated by relatively unskilled operators. These tasks include source inventories, data interpretation, and standard report writing. This approach will integrate with most GIS systems in the Mexican government and link directly to commercial databases.

Quality Control / Quality Assurance / Accreditation / GLP: Data quality in many developing countries is a serious problem. Increasingly QA/QC and GLP (Good Laboratory Practices) are a major part of bilateral assistance programs in developing countries. The problem of reliable data within a laboratory has too many facets to discuss here. These include not only the normal quality control steps during analysis, but also the difficulty in many countries of obtaining pure reagents or of ensuring that so-called certified reagents are, in fact, pure. Site conditions in many developing countries are a major problem, especially the location of laboratories in highly polluted airsheds of major cities. Rarely do laboratories have proper air handling systems that can deal with such problems. This is particularly acute for metals and, given the recent experience with contamination of samples in very well operated North American metals programs, suggests that metals data are especially unreliable in developing countries.
A major advancement and absolute necessity, especially in large countries with many laboratories, is an accreditation program that establishes common performance criteria for all labs responsible for water quality data used by government. Modernization requires rigorous application of these principles both in government laboratories and in those private laboratories that serve government needs.

Reporting: For descriptive purposes at the national scale, monitoring should increasingly focus and report upon biological conditions and measures of toxicity including the use of sentinel organisms for determining the presence of important industrial parameters. This follows from the recognition that chemical data are difficult to interpret or to relate to the actual ecosystemic state of the water environment. Chemical parameters should include general indicators such as adsorbable organic halogens (AOX) as an indicator of total chlorinated compounds. A limited number of key parameters (especially for parameters that are controlled or prohibited) could be included. Data should be integrated into several useful indices that describe the water according to water use. Surface water sites should be reported separately from a national effluent monitoring program. If there are a large number of river sites, the sites should be integrated (for reporting purposes) into river reaches.

C. CAPACITY BUILDING

Training: Contemporary needs of water managers for water quality data and for informed analysis and interpretation, is often beyond the technical ability of staff in many developing countries. Managers of monitoring programs are often not committed to sufficient training, especially for junior staff. Investment in infrastructure, equipment, etc., is not effective or efficient without a commensurate investment in training. In the 1995 redesign of the Mexican national water quality program, a significant percentage of the program resources are dedicated to training and professional development. This training includes activity-oriented training for the bulk of field and laboratory staff, and advanced training in Mexico and abroad for key staff who will be the leaders of the program in the future. In contrast, certain other countries are reluctant to invest in training as a matter of program strategy and tend to rely on training that can be obtained at low or no cost from donors. Unfortunately, experience suggests that while such courses may be individually of merit, collectively they do not comprise a well-structured and balanced program that meets the needs of the agency. Also, there is often little if any follow-up. A strategic training program also needs to include consideration of the types, numbers and length of foreign "training" missions undertaken by domestic staff.

There is much use by many countries of "short courses" by foreign professionals. Experience indicates that short courses are generally valuable only in two circumstances - one is for the raising of awareness of new types of monitoring and assessment methodologies; the second is the training in very specific topical areas having limited scope (e.g. training in a particular methodology). Training programs overseas that involve extensive visits to labs etc., especially for junior staff, are not particularly cost-effective. Foreign training should be restricted to highly specialized training (e.g. specialized analysis, computer training, etc.), for extended educational purposes (advanced degrees) and for senior staff who need familiarization with foreign environmental management methods, policies and regulations.

A national strategic training program should establish training goals of "x" number of PhD and MSc degrees within the next "n" years to develop a core group of professionals around which the next generation of monitoring will be built. The principal mechanisms for advancement of monitoring programs in developing countries include:

- On a competitive basis, identify the best persons for foreign education in environmental chemistry and toxicology, environmental assessment, etc.
- Increase the number of persons with advanced degrees within monitoring programs.
- Promote persons with advanced foreign education and suitable experience to positions of management responsibility.

Other mechanisms include the development of long-term relationships with foreign companies and agencies for importation of appropriate methodologies. This may be done through international aid programs and, in some cases, with specialized foreign agencies (e.g. US-EPA). However, the relationship with foreign agencies and companies can increasingly be expected to be of a commercial nature, due to the economic change occurring in most western countries. Also, for many developing countries, commercial relationships are consistent with developing market economies and offer a fast-track solution to training, program efficiency and financial sustainability of the program by the development of a market-based program in which the government is one of the clients.

Institutional Development: Institutional development is necessary to deal with all aspects of –program management, legal change, etc. Below are two very specific types of institutional development that are essential if modernization of data programs is to be successful.

a) Client-Oriented Programs: It is essential that managers of monitoring programs regard their work as a service for clients. Clients in most cases are the different levels of government. However, as the move to a market economy increases, there may develop the need to work with other types of clients. Use of Data Quality Objectives (DQOs) ensures greater communication between the monitoring program and clients that use the data. The objective is to ensure that the client understands the costs, the limitations and the uncertainty in the information produced by the monitoring program.
b) Revenue Generation: Data programs can be put on a revenue-generating basis, especially if the government is considered to be a client. A revenue basis is essential for financial sustainability of program; it is also a major tool in creating the appropriate business approach to data programs. A business approach ensures that data programs are relevant and efficient, well connected to users and are operated according to good scientific and business practices. There are many ways to accomplish this. However, the major message here is that government managers of water programs usually find the concept of revenue very difficult to accept. Therefore, institutional change requires education about alternative approaches to the business of water data programs. Experience shows that modernization creates opportunities for revenue generation for government laboratories and opportunities for creative partnerships between government agencies and the private sector. In many developing countries, a revenue base may provide the funds to adequately compensate staff to achieve personnel stability.

CONCLUSIONS

Program modernization is essential to achieve the twin goals of greater efficiency and greater relevance in meeting data needs for contemporary water quality management purposes. Modernization reduces costs, may reduce the amount of equipment and infrastructure required, often reduces the amount of data collected and more closely matches the abilities of developing countries where, for example, knowledge of advanced environmental chemistry and toxicology may be limited. Modernization is a complex and comprehensive activity that includes legal and institutional considerations, technical issues and a strategic program of capacity building.

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A MANUAL OF BEST PRACTICE FOR WATER MONITORING: UK-ITALY COLLABORATIVE PROGRAMME

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ABSTRACT

A collaboration programme involving the Environment Agency (formerly the National Rivers Authority) in the UK and the Po River Basin Authority in Italy has been established, with the objective of exchanging experiences on water quality monitoring and developing a Manual of Best Practice for surface water monitoring. The purpose of the Manual is to optimise the cost effectiveness of monitoring, through improved approaches to programme design and implementation. It is envisaged that this collaboration programme will form an important input into the European Environment Agency's (EEA) desire to promote best practice throughout Member States as part of the establishment of a pan-European monitoring network.

INTRODUCTION

Over recent years there has been growing awareness and concern about environmental quality issues, at both national and European levels. Inevitably, this has resulted in increasing environmental information requirements, in terms of both quantity and quality of information; a trend which is likely to continue. This can be seen, for example, in the demanding monitoring requirements resulting from the implementation of new European Union (EU) Directives. The net effect of this trend towards a greater need for environmental information has been a large and costly monitoring burden for Member States. At the same time, this has focused attention on the need to optimise the efficiency of monitoring programmes, and to ensure that monitoring is carried out in a consistent manner across Member States.

The European Environment Agency (EEA) is playing a key role in pursuing the optimisation and harmonisation of monitoring. Recently the EEA has issued a proposal for a pan-European monitoring network for surface waters and groundwaters (ETC/IW 1996). This provides the framework for a harmonised, transnational monitoring network but to support such a programme there is the need for sound underlying methodologies for the design and implementation of monitoring programmes.

There are no precise figures on the costs of water quality monitoring across Europe but some estimates put the current total expenditure for the EU somewhere in the order of 500 MECU annually (Groot and Villars, 1995), and this can be expected to increase in the light of future requirements. It is essential therefore that the efficiency of monitoring programmes are optimised as far as possible, and that opportunities for cost savings are fully exploited.

Against this background a collaboration programme has been established between the Environment Agency (formerly the National Rivers Authority) in the UK and the Po River Basin Authority in Italy (Figure 1) to produce a Manual of Best Practice for surface water monitoring. The Manual aims to promote improved approaches to the design of monitoring programmes, and provides a common framework within which the optimisation of monitoring programmes can be achieved.

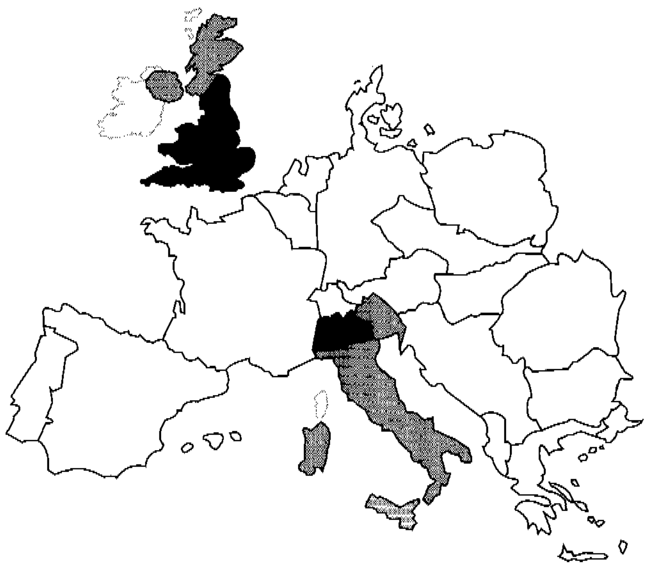

Figure 1: Environment Agency in UK and Po River Authority in Italy.

In the UK, the Scottish Environment Protection Agency and the Northern Ireland Environment and Heritage Service are associated with the work programme.
This paper briefly reviews the legislative and institutional arrangements in England and Wales, and Italy with regard to monitoring and provides an overview of current monitoring practice. The development and evaluation of a Manual of Best Practice is presented and its role in a wider European context is described.

THE ENVIRONMENT AGENCIES IN THE UK

In April 1995 three new Environment Agencies came into being in the UK; the Environment Agency, for England and Wales, the Scottish Environment Protection Agency in Scotland and the Northern Ireland Environment and Heritage Service.

The Environment Agency brings together the former Inspectorate of Pollution, the National Rivers Authority, local waste regulation authorities and certain parts of the Department of Environment into one regulatory organisation whose prime role is the protection and enhancement of the environment as a whole, including surface waters, groundwater land and air. The Agencies in Scotland and Northern Ireland have similar duties.

The Environment Agency operates on a regional basis, with eight Regions based largely on the major river catchments of England and Wales.

THE PO RIVER BASIN AUTHORITY IN ITALY

The Po River basin Authority was established in 1989 with the responsibility of preparing and implementing the Master Plan for the whole of the Po River Basin. The Po Basin is the largest catchment in Italy covering an area of some 72,000 km^2 (which represents about 24% of the area of Italy) and is made up of six administrative regions, Lombardia, Piemonte, Liguria, Emilia Romagna, Veneto, Valle d'Aosta and the autonomous province of Trento. The Master Plan specifies the policies and actions required for effective environmental management in the Po Basin and covers such important aspects as surface and groundwater pollution control, drinking water contamination, flood control, landslides and soil erosion. More detail can be found in Pagnotta et al (1995).

PURPOSES OF WATER QUALITY MONITORING

INTRODUCTION

Water quality monitoring is undertaken primarily to provide information for the following three needs:

1. assessment of compliance with targets, authorisations, environmental quality standards or consents;
2. surveillance, for the detection of long-term trends and changes in quality; and
3. investigations, to provide answers to specific questions.

There are a number of different international, national and local requirements behind these needs for monitoring , amongst which are the following principal categories:

- implementation of EU legislation and international agreements;
- implementation of national legislation;
- classification schemes;
- local requirements, including support of local water management activities; and
- determining the effectiveness, or otherwise, of policies.

The first two categories usually comprise statutory requirements for monitoring, with many aspects of the monitoring programme design pre-defined. Classification schemes, or routine monitoring networks, are generally implemented at a national level for the categorisation and surveillance of water quality status and trends and to support water quality planning and pollution control management. Local water management activities often encompass a more varied and dynamic type of water quality monitoring, generally designed at the local level and including short-term and investigative programmes.

Determining the effectiveness, or otherwise, of policies at all levels is increasingly being seen as an important aim of monitoring programmes.

IMPLEMENTATION OF EU LEGISLATION AND INTERNATIONAL AGREEMENTS

A number of EU Directives and Decisions have been adopted that require monitoring of surface waters as part of their implementation. These include:

- Surface Water for Abstraction Directive (75/440/EEC);
- Dangerous Substances Directive(76/464/EEC), and Daughter Directives;
- Decisions on the Exchange of Information on the Quality of Surface Freshwaters (77/795/EEC and 86/574/EEC);

- Freshwater Fisheries Directive (78/659/EEC);
- Urban Waste Water Treatment Directive (91/271/EEC); and
- Nitrate from Agricultural Sources Directive (91/676/EEC).

These Directives and Decisions require monitoring for a variety of reasons, including: monitoring compliance with environmental quality standards, monitoring trends in surface water quality, and identifying areas susceptible to pollution. The extent to which the Directives are implemented varies between Member States depending on the characteristics and uses of the surface waters and the nature of the inputs to these waters.

As well as these requirements from EU legislation, there are a number of international agreements and conventions that have associated monitoring requirements, for example the Oslo and Paris Commissions. These agreements often relate to the limiting of pollution loads, as well as concentrations.

NATIONAL REQUIREMENTS

In addition to the monitoring requirements arising from the implementation of EU legislation, monitoring is undertaken to satisfy national requirements. These national requirements result from the implementation of specific legislation or for the purposes of supporting more general water management activities at the operational level.

Detailed accounts of national monitoring programmes were given by the authors in Monitoring Tailor-Made I (Pognotta et al., 1995; Seager, 1995). Brief summaries are given below.

Italy

The basic legal framework for national water quality protection in Italy, including the elements relating to water quality monitoring, is set out in a number of laws. These define monitoring requirements, including the tasks to be undertaken for the identification of the water bodies to be monitored, sampling location, methods and sampling frequency.

The national Government is empowered with supervision, and delegates to the Regions the operational design and establishment of monitoring activities in water bodies.

A recent law (61/94) brought about the re-organisation of environmental controls and instituted the National Agency for Environmental Protection (ANPA) with responsibilities for: collecting and disseminating through public and educational programmes all environmental information, setting limits for polluting agents, air soil and water quality standards, sampling and survey rules, and establishing monitoring methods for the determination of the state of the environment and pollution control.

England and Wales

In England and Wales the legal framework for river monitoring is contained in the Water Resources Act 1991. Various Sections and Schedules of the Act require the Environment Agency (formerly National Rivers Authority) to undertake river monitoring for different purposes.

Prior to the formation of the National Rivers Authority in 1989, river monitoring for the purposes of water quality classification was undertaken by the ten former public Water Authorities.

Although this monitoring was undertaken within the framework of the former National Water Council classification scheme, problems arose with the comparability of data between the ten Authorities, since their individual approaches to monitoring differed.

With the adoption of the Water Act 1989 (subsequently consolidated into the Water Resources Act 1991), the use-related and classification elements were separated into two new systems; statutory Water Quality Objectives (yet to be fully implemented) incorporating the use-related elements, and a General Quality Assessment (GQA) scheme for classification purposes.

The GQA scheme will be implemented in a consistent manner by the eight Regions of the Environment Agency, enabling objective comparisons to be made of water quality across the country.

LOCAL REQUIREMENTS

Varying amounts of monitoring may be undertaken to meet local operational needs, as opposed to statutory, international or national requirements. This type of monitoring is usually locally driven, designed and implemented, although there may be regional or national guidelines. Local monitoring may be undertaken for a number of purposes, including the following:

- discharge impact assessment and pre-consenting studies;
- development impact assessment;
- diffuse source impact assessment;
- detection of trends and general water quality characterisation;
- Research and Development;
- post pollution incidents;
- real time water quality management;
- model development and validation

The diversity of local monitoring requirements require flexible approaches to the design of monitoring programmes and often very varied individual programmes. There are often diverse organisations undertaking local monitoring within a country, which can result in widely differing approaches to the design of monitoring programmes to address essentially similar issues.

THE ROLE OF THE EUROPEAN ENVIRONMENT AGENCY

The prime objective of the EEA is to achieve the aims of environmental protection and improvement laid down in the Treaty of Maastricht and by successive Community action programmes on the environment, by providing the Community and Member States with objective, reliable and comparable information at the European level. This will allow governments to take the requisite measures to protect the environment, to assess the results of such measures and to ensure that the public is properly informed about the state of the environment. The European Topic Centre on Inland Waters (ETC/IW), working under contract to the EEA, has recently published a review of the international legislative requirements for monitoring all surface waters. This has identified a number of barriers to the harmonisation of monitoring arising at the sampling, analysis and reporting stages because of conflicts between Directives or because of imprecise specification within Directives. A widely accepted and unified approach to surface water quality monitoring, based on best practice, would clearly be beneficial in eliminating many of these barriers. The ETC/IW is, therefore, seeking out, developing and promoting examples of best practice that could be deployed throughout the EEA area.

The work programme of ETC/IW is ultimately leading to the design and establishment of a freshwater monitoring network for the EEA area. The EEA monitoring network will be based, where possible, on existing national and international networks, using existing sources of monitoring information. The EEA has recently published a proposal for the establishment of such a network (ETC/IW, 1996).

The objectives of the freshwater monitoring network are to obtain timely, quantitative and comparable information for all EEA member states, so that valid temporal and spatial comparisons can be made and key environmental problems associated with Europe's inland waters can be defined, quantified and monitored. The EEA has recently reached agreement to co-ordinate the freshwater monitoring network with the monitoring networks of the Eastern and Central European countries to create a larger, pan-European monitoring network.

Underlying the philosophy of the EEA's mission is the recognition that information should be relevant and reliable and that the outcome of decisions and policy must be measurable - in other words that real environmental improvements can be shown to have occurred. The EEA recognises that there is already an abundance of data and information that is not used by decision makers because it is either irrelevant or not sufficiently reliable.

COLLABORATION PROGRAMME

INTRODUCTION

Against this background of a need for greater harmonisation of monitoring across Europe and for the optimisation of the efficiency of monitoring, a major collaboration programme to develop a Manual of Best Practice for surface water monitoring is currently being undertaken by the Environment Agency in the UK and the Po River Basin Authority in Italy. This collaboration arose from the recognition of a common need to optimise surface water monitoring and represents a unique pooling of expertise by two of the major regulatory organisations in Europe. The collaboration programme recognises the need for the development of efficient monitoring programmes which will meet the requirements of EU and national legislation, as well as local operational needs, but will at the same time minimise expenditure on monitoring through the use of cost-effective approaches.

REVIEW OF CURRENT PRACTICE

A key component of the collaboration programme was an initial review of current monitoring practice undertaken within each organisation. The review examined the requirements of those involved in the design and interpretation of monitoring programmes and identified the range of tools and procedures currently used. This information has led to the identification of the most appropriate procedures to be adopted for particular monitoring requirements, and to meet particular monitoring aims. These procedures have been drawn together into a draft Manual of Best Practice. In addition, the review exercise allowed the identification of areas where there is a need for improved methodologies to be made readily available to practitioners.

SCOPE OF THE MANUAL OF BEST PRACTICE

The Manual gives step-by-step guidance through all the stages of the design and evaluation of a monitoring programme (Figure 2). For a given monitoring aim, the user is guided through the processes of choosing an appropriate monitoring strategy: deciding what to sample and how and when to sample it, as well as how to analyse the resulting data and generate management information. The Manual does not cover the choice of analytical techniques. The guidance covers the use of both chemical and biological monitoring methods, for rivers, estuaries and coastal waters. The user will therefore be able to design a monitoring programme that will be most appropriate for the set of problems and circumstances encountered.

Importantly, the process is not prescriptive but guides the user to the optimum design to meet their particular situation and needs. The Manual recognises that resources are usually a limiting

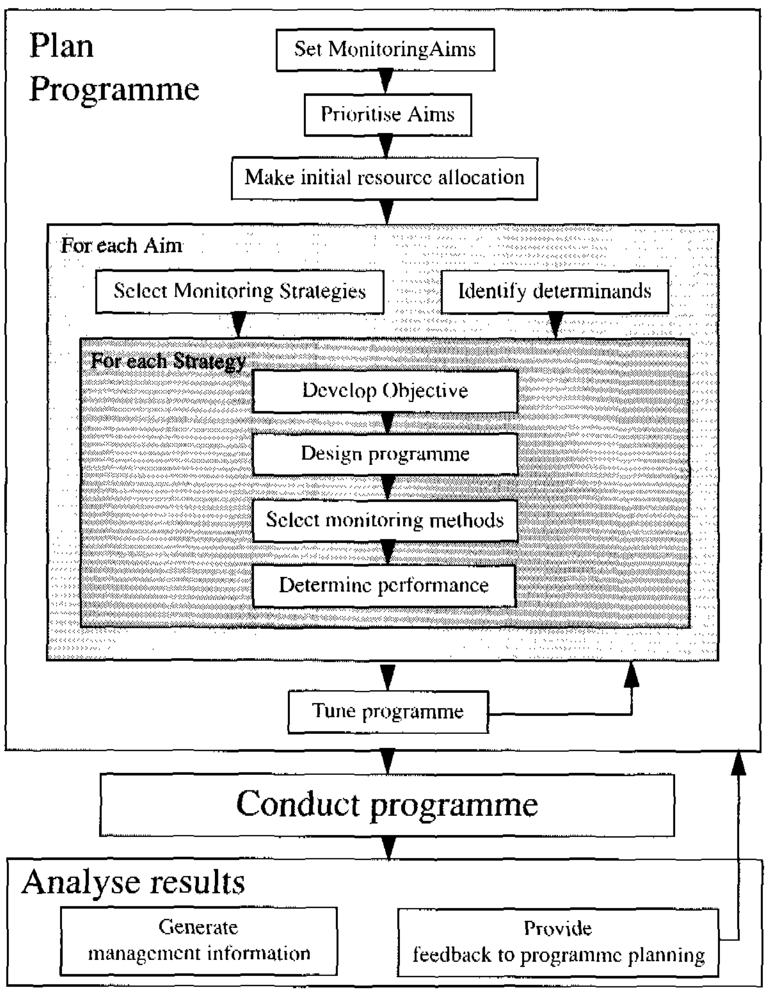

Figure 2: Structure of Manual of Best Practice.

factor for monitoring programmes and therefore allows monitoring programmes to be made more effective within the constraints of a fixed resource. Another important aspect of the Manual is maximising the use of existing data, to address current needs and in the design of future programmes, as well as maximising the potential reusability of new data.

Under the collaboration programme, a range of software tools are being developed which will complement the Manual and facilitate the monitoring programme design process. The tools will encompass a number of features to aid the achievement of cost-effective designs. All the software tools will operate as stand-alone packages running under Microsoft WINDOWS.

There are a number of existing manuals and other publications on monitoring programme design, e.g. Ellis (1989) and Adriaanse et al. (1995). These tend either to be statistically detailed, which can discourage users, or to be fairly general in approach. The current Manual aims to be sufficiently detailed to allow precise planning of a programme while keeping the more complex statistics hidden in the software tools, and therefore easy to use.

Some key areas where the Manual, and associated tools, are expected to help the user optimise monitoring programmes are outlined in the following sections.

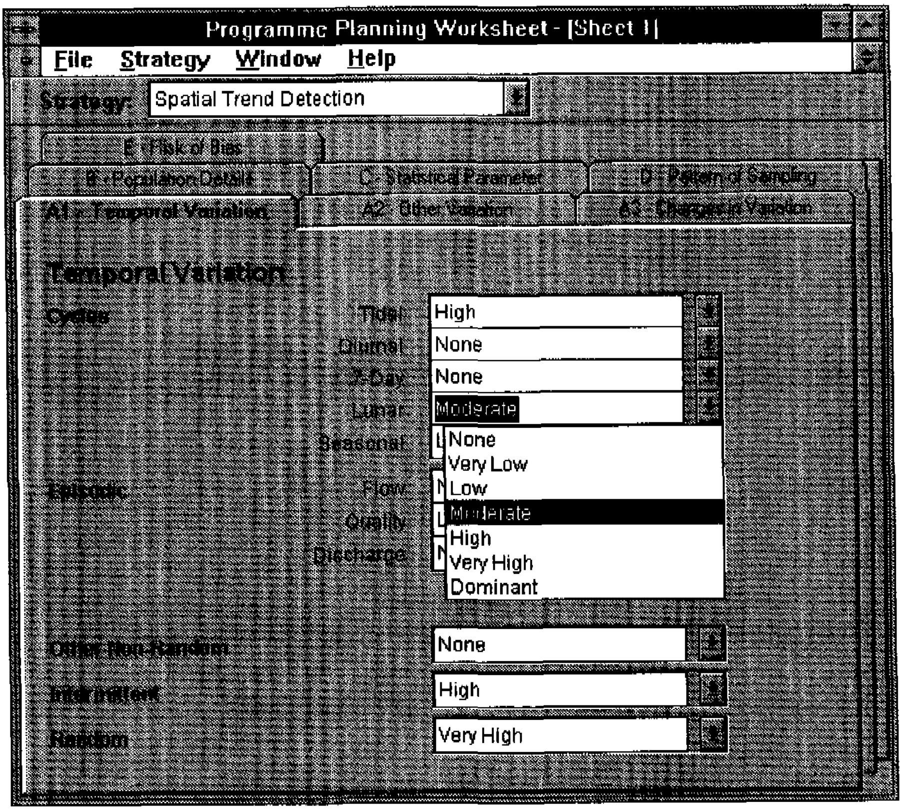

Figure 3: Example of planning software.

EXPLOITING KNOWLEDGE OF VARIABILITY

The variability of data arising from samples taken will, in part, be determined by the underlying variability of the environment. Much of this variability is predictable, for example diurnal and seasonal patterns. The Manual and associated tools allow the user to gain an understanding of this variability and to use this information to target the sampling effort in such a way as to minimise the variability of samples taken. This will allow achievement of a specific monitoring objective with fewer samples. To take a simple example, if there is a strong diurnal pattern, taking samples at the same time of day will allow the detection of an underlying difference in quality between two sites with fewer samples than if the samples are taken randomly through the day. Figure 3 shows an example of prototype planning software which helps the user to understand and exploit variability when designing a monitoring programme.

RE-USABILITY OF DATA

Clearly, the various purposes of monitoring described earlier have very different requirements in terms of monitoring programme design. However, there is often considerable overlap and the first step in maximising the cost efficiency of monitoring must be the re-use of data for multiple purposes wherever practicable. For example, data already being collected for a statutory EU requirement might be re-used in a national surveillance monitoring network. (The EEA recognises the importance of the re-use of data and its pan-European freshwater monitoring network will use existing national and international networks as far as possible.)

At the programme design stage, the Manual prompts the user to consider the potential re-usability of the data that will be collected, primarily in terms of how restricted the pattern of sampling is (analytical methods are not considered). At the same time, a check is made that the best use of existing data is made in the planning of a new monitoring programme, and in achieving its objectives if appropriate.

ASSESSING EFFECTIVENESS IN ADVANCE

The Manual allows an assessment of the likely effectiveness of the monitoring programme at the design stage. The user can therefore ensure that the expected accuracy, precision and quality of information will be in accordance with the perceived importance of the monitoring aim and sufficient for any decision-making requirements.

One of the software tools to be developed will allow easy estimation of the precision and confidence likely to be achieved by the planned programme. The user will be able to input their chosen amount of sampling effort, together with estimates for the major components of variability, and will be presented with a range of options of achievable precision and confidence. If these are unacceptably poor, the user can consider the effect of augmenting the sampling effort by differing degrees. Alternatively, if the precision and confidence are better than required, the scope for reducing the sampling effort can be investigated. In this way, the overall resource available can be distributed between elements of the monitoring programme to maximise cost-effectiveness.

MONITORING PROGRAMME SIMULATOR

A computer simulator of monitoring programmes is being developed. This will allow the user to visualise the degree of uncertainty and bias inherent in a chosen sampling programme. The simulator will run in three stages:

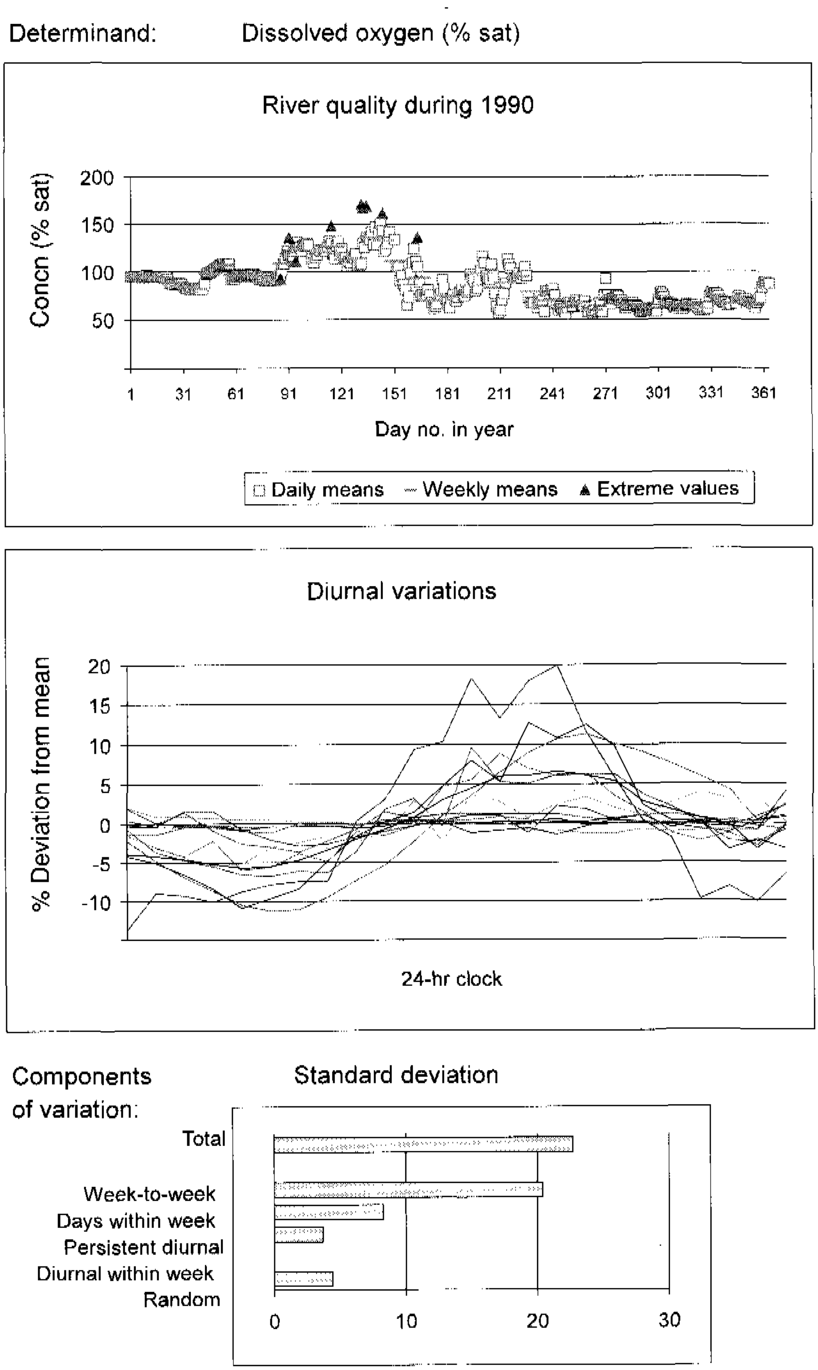

Figure 4: Example output from prototype software for understanding continuous monitoring data.

1. the user sets any desired description of actual quality variations over time (i.e. defines the 'statistical population' to be sampled);
2. then the user specifies the number of samples and the sampling pattern through space and time; and
3. the simulator randomly generates many repeat sets of samples according to the prescribed pattern, and displays the relevant 'answer' e.g. mean upstream-downstream difference, 90%ile, slope of trend etc.

A graphical display showing the levels of uncertainty and bias inherent in a chosen monitoring programme will provide a powerful illustration of the degree of effectiveness of what is planned.

APPROPRIATE DATA ANALYSIS METHODS

As well as optimising the planning of a monitoring programme, it is important to ensure that the data collected is analysed appropriately. Use of grossly inappropriate data analysis may of course lead to false conclusions being drawn. More subtly, the use of valid but sub-optimal statistical approaches to data analysis will reduce the power of the analysis, in other word the power to detect trends and changes in quality. This will also increase the risk of arriving at wrong conclusions (or will require more sampling effort to arrive at the same conclusion).

Under the collaboration programme, a number of custom software packages will be developed for specific water quality data analysis applications, for example for making upstream-downstream comparisons. These will enable the non-statistician to be confident that they are performing the most appropriate statistical analysis, and will display the conclusions in a straightforward manner, without calling for interpretation of complex statistical outputs, and with visual support through graphics where appropriate.

EXTRACTING INFORMATION FROM CONTINUOUS DATA

One specific area where monitoring programmes can be made more effective is in extracting the maximum possible useful information from continuous monitoring data. Data of this sort may be collected for general water quality monitoring purposes, or for alarms and warning systems. Whatever the reason, the huge amount of useful information contained within a long series of continuous monitoring data is all too often under-utilised. One of the principal reasons for this is the lack of appropriate tools for extracting the information. To this end, a software tool is under development which will allow the extraction of information on trends, cycles and random variation and to produce succinct summaries of this information. An example of one of the outputs from this software is shown in Figure 4.

UNDERSTANDING BIOLOGICAL DATA

The majority of existing tools for water quality data analysis are aimed specifically at chemical quality data, while the interpretation of biological data is often left to more subjective assessments. Another of the tools under development in the collaborative programme is aimed at detecting statistically significant spatial trends in biological quality along the length of a river. An example one of the outputs from this software is shown in Figure 5.

EVALUATION PROGRAMME

The draft Manual of Best Practice, and associated tools, are being evaluated in selected catchments in the UK and Italy, as part of the ongoing work programme. In the UK, the test catch-

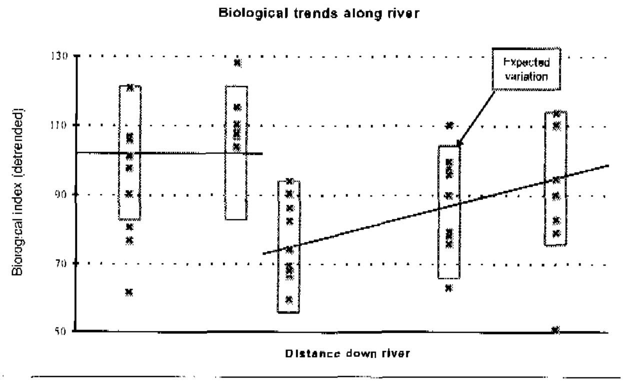

Figure 5: Example output from prototype biological data analysis software.

ments are the River Mole and the River Weaver in England, and the River Carron in Scotland. In Italy, the test catchments are the Sesia and Enza catchments, in the Po River Basin. During the evaluation programme, the Manual will be used by operational staff in the test catchments to provide a realistic assessment of its performance and ease of use.

Case studies from these trials will be incorporated into the final version of the Manual as illustrative examples. In particular, examples will be used to illustrate how improvements in design can critically effect the outcome of monitoring programmes and subsequent decision making.

The final version of the Manual is due to be published in mid-1997. It will be implemented initially in the UK and Italy.

CONCLUSIONS

The development of the Manual of Best Practice, which will become available during 1997, is seen as an important contribution to the needs of EU Member States and in support of the role of the European Environment Agency in promoting best monitoring practice. Wide implementation of the Manual will enable practitioners in Member States to maximise the cost-effectiveness of monitoring programmes and ensure that data produced is relevant to information requirements and decision making. At the same time, the promotion of a common framework will help to reduce some of the current barriers to the harmonisation of monitoring that arise from differences in approach.

With the growing monitoring demands on Member States and the associated cost burden, it is becoming increasingly important that there are effective feedback mechanisms that can demonstrate that the outcome of EU legislation, and the investment in resulting monitoring programmes, was indeed effective.

Use of the Manual will ensure that the precise objectives of a monitoring programme are specified in advance and that its expected outcome in terms of information generation is also known. The approach therefore allows the opportunity to retrospectively assess the achievement of the monitoring programme directly against what was intended. Assessment of effectiveness will therefore be possible, as well as feedback into the design process to further optimise future programmes.

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COMPARING MONITORING OF SURFACE AND GROUND WATER SYSTEMS

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ABSTRACT

Monitoring can be seen as part of a complex cybernetic circuit in which many actors and system characteristics are involved. Monitoring and assessment are essentially transmitting 'differences' of characteristics of a water system relevant to develop a water system to an ideal or a (socially or economically) preferred situation. Monitoring systems for ground water and surface water are seldom integrated into one system. Mostly only one or a few purposes (e.g. ground water quality or ground water level) are taken into account per monitoring system (single purpose monitoring).

Integrated water management concerns both ground water and surface water systems. The way of monitoring and the purpose of monitoring in these water systems for water management reasons can be quite different, due to differences in properties and human interest/use of both systems. There are a number of similarities between monitoring and assessment of ground and surface waters. Especially procedures for optimizing monitoring systems and tuning monitoring information in to information needs are similar. The monitoring cycle can be helpful for tailor-made monitoring in both types of systems.

Various types of monitoring systems are compared from different points of view. Also, the linking up of both monitoring types is discussed. Two examples in The Netherlands are presented: monitoring of desiccation of aquatic and terrestrial ecosystems and surface and ground water monitoring of a lake/ground water system. The need to integrate monitoring of ground water and surface water depends on how far processes and variables in ground water and surface water are interrelated. It is often sufficient to gear the information needs for both systems to one another and to integrate the assessment. Physical integration is not recommended, for practical reasons.

For the functions 'transport of water and solids/solutes', 'ecosystems' and 'agriculture', joint assessment of information from both systems can be worthwhile. Especially if the issues are: desiccation, salinization and acidification of ecosystems; macro- and micro-pollution, and drought damage to agriculture.

INTRODUCTION

Before the differences and similarities of monitoring between ground water and surface water are discussed in the third paragraph, some general principles of monitoring will be introduced. This introduction will present my point of view about monitoring of water systems. My vision about the position of monitoring in the framework of water management will be presented in the second paragraph. However, before that it is illuminating to look at monitoring from a cybernetic perspective. So in the first paragraph the similarity between monitoring water systems and cybernetic circuits like the steam engine is clarified. The general introduction is concluded with definitions of monitoring for integrated water management and remarks about the developments in monitoring of water systems and the importance of the monitoring cycle.

DIFFERENCES THAT MATTER

Gregory Bateson, a biologist, anthropologist and philosopher, was an inspiring person. He developed the cybernetics and information theory, together with Norbert Wiener and others (Gregory Bateson, 1984). For many years he was thinking about the concept of mental processes, trying to find out what "the environment" of organisms, what "an organism" and especially what kind of thing "a mind" is. He discovered that all kinds of causal circuits with complex energy relations and feedback loops have similar "mental" like characteristics. These systems compare situations or states and are responsive to differences, in addition to being affected by ordinary physical causes such as impact or forces. They process information and will inevitably be self-corrective, either towards homeostatic optima or towards the maximization of certain variables (Gregory Bateson, 1984). A 'bit' of information can be defined as 'a difference which makes a real difference' in the state of a system. Bateson often used as an example the simple self-corrective system of a steam engine. In a steam engine a 'governor' is an essential sense organ or transducer, which receives a "transform" of the difference between the actual running speed of the engine and some ideal or preferred speed. This organ transforms these differences into operational messages, for example to supply fuel or to brake. The behaviour of the governor is determined by the behaviour of other parts of the system, and indirectly by its own behaviour at a previous time. The latter clearly demonstrates the mental-like character of the system. Message material, i.e. successive transforms of difference, must pass around the total circuit, and the time required for the message material to return to its starting point is a basic characteristic of the total system. Thus, there is a kind of determinative memory in even the simplest cybernetic circuit.

So, all parts of networks of closed information and causal circuits, through which differences are transformed, besides the governor, are important. We can use this 'steam engine metaphor' for monitoring systems.

Monitoring systems are tools for the manager, 'who governs' the management of a water system. The water system, water users, the manager and his governing tools can be seen as one large complex mental system. In a broad sense the manager is like 'the governor' and the monitoring system is a part of his information system. Besides monitoring, the manager also uses other information about the water system such as historical data (memory), models, and operational devices, such as sluices or legislative tools, etc. Monitoring is essentially transmitting 'differences' of qualities of a water system in time and space, which are relevant for developing or maintaining an ideal or a socially or economically preferred situation. As in a steam engine, the differences detected, as 'bits', must go through the whole water management circuit. A 'bit' goes from water system to water manager, maybe to water users, public and science. For this, we have our own monitoring cycle, linking information needs of the water management (policy) via the monitoring system to information ready to use in water management. As a result, this leads to correcting water system valves and steering variables, changing the situation to a target state. When all steps of the circuit have been passed, the state will be checked in a following cycle. Thus, monitoring can be seen as part of a complex cybernetic circuit in which many actors and qualities are involved, such as water system goals and targets, water managers, public interest, commercial interests, nature conservation interests, etc. The similarity between monitoring and information theory forms the basis of this paper.

MONITORING AND INTEGRATED WATER MANAGEMENT

Water management can be defined as research, policy and operational activities, pursued to maintain, to control, to manage and to develop functions and (potential) uses of the water system involved. A water system is defined as: the water, her banks and underwater beds, including relevant ground water systems, also including the existing ecosystems, and the technical infrastructure.

Integrated water management (IWM) concerns surface water and ground water systems. It is concerned in the integration of all different aspects of a water system: water quality, water quan-

tity, ecology, but also in the sociological and economical aspects of water management.
Water systems do not always have strict and permanent boundaries.
They are limited in space and time by morphological, ecological and functional characteristics.
Within one system there are specific goals and/or problems related to specific characteristics.
The goals and problems should determine the schematization of a water system and its relevant environment.
IWM also means managing the water systems coherently, in one or more control systems, by *gearing the different policy and management levels to one another as good as possible*. During the past decade IWM has become the general management policy for water systems in The Netherlands.

Definition of monitoring

In ancient times the monitor was a man at the front of a (war)ship, sitting in the crow's nest at or near the top of the central mast, or on the bowsprit in front of the ship, watching and warning for land, reefs, floating mines, whales, friends and foes. The monitor informed the captain and crew. The system to be managed comprised the sea, the weather, the boat, the cargo, the crew and the captain sailing with a specific mission. This metaphor can help to see the use of monitoring and the need for information to get a goal. In fact, the monitor searches/warns for differences that matter to the captain, or, in our case, the differences that matter to the water manager.
Monitoring deals with differences in characteristics of the system, which are relevant for the operating or the performance of the system. The differences in the characteristics of the system will matter if they urge the manager to take measures to ensure that he will reach a goal or target in time or to change his goal. If at some moment there are no substantial differences, this means that the systems' performance is all right.
The general definition of monitoring according to Webster's Dictionary (1980) is: any or various *devices for checking or regulating the performance of a machine, aircraft, guided missile, etc.* but also of water systems. Figure 1 shows the general information system for water management. As mentioned before, this information can be produced in different ways. Monitoring is only one of the tools for management. Information can also be obtained from surveys (relatively short-time field inventories) or from rules of thumb, expert judgement, from models, decision support systems or from other system knowledge, such as historical databases. Of course, also combinations of these are possible.
Monitoring water systems provides systematic, permanently available information about the performance of the water system, with respect to its societal and ecological functions. The monitoring system provides relevant information about the water system. This regards the functions,

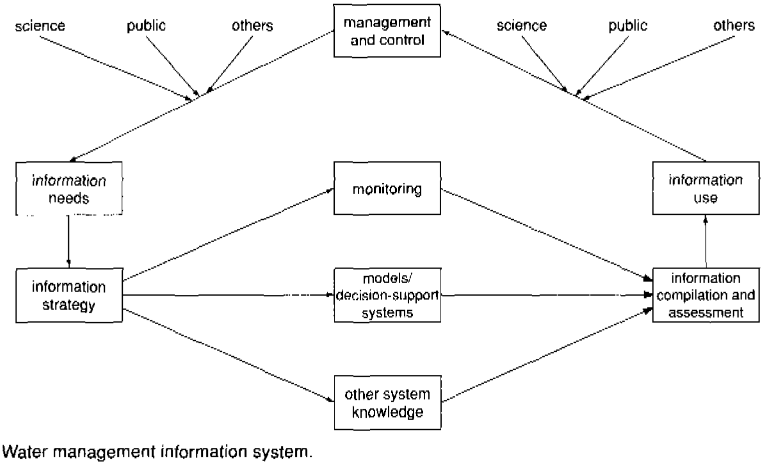

Figure 1: Water management information system.

the policy goals and targets, such as correct functioning and good health of the ecosystems, and the state and qualities of the water system resources.

In the past monitoring networks were constructed for different reasons:

1. For operating water management systems and for devices, such as manipulating sluice gates, inlet works for water supply, operation of hydraulic pumping stations and electric power plants (cooling systems), for navigation, reservoir management, water level control, etc. Initially, these were often single purpose operating systems.
2. To amass general knowledge about the water systems and water resources. Mostly, a governmental or scientific organisation was responsible for this. Every ministry or administration unit, dealing with one or more aspects of water management, built its own monitoring network. As a result of (1) and (2) separate monitoring networks were developed at different geographical scales, for different scientific disciplines and for different sectors of water users.
3. In the past, the information need for design was based on the idea that to measure the system was to know the system. Every aspect of the hydrology of an area would bring more accurate information. In fact, monitoring in this way tends to suffer from a chronic failure to establish meaningful programme objectives (Adriaanse and Lindgaard-Jørgensen, 1996). A clear view on the information product and on the cost-effectiveness of monitoring is missing (Ongley, 1994).

Today, especially in a highly industrialized and densely populated country such as The Netherlands, awareness has grown that integration of water management and policies are required to cope with water problems. Therefore, integration of monitoring systems is needed as well. Besides, there is an increasing consensus that monitoring systems must deliver information required for action, for decision-making. The information needs for management and control must be well specified and quantified. They must be so specified that design criteria for the various elements of the information system can be derived. Specified relevant margins are helpful for network design. With these, sampling frequencies and density of the network can be optimized, when reliable time-series of measurements are available.
Figure 2 shows how (optimization of) the monitoring- and information system can be schema-

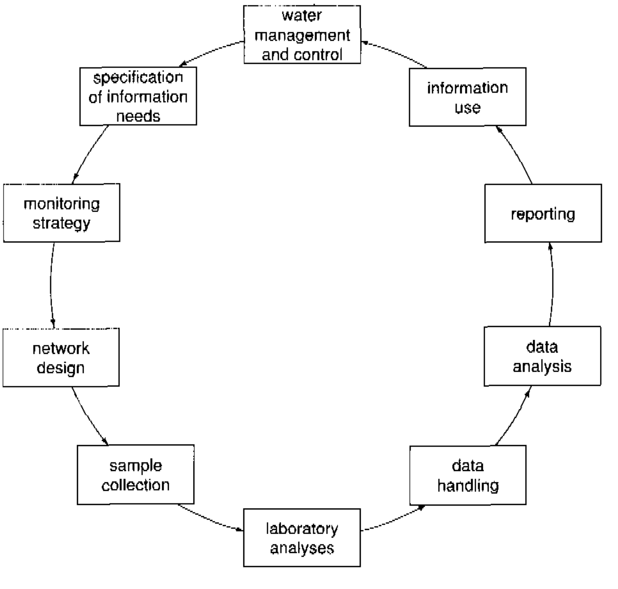

Figure 2: Chain of activities in the monitoring cycle.

tized as a chain of activities. In essence, the chain is closed when the resulting information meets the information needs of water management and control. Schemes from the past may show a more top-down sequence of a restricted number of activities, starting from a rather arbitrarily chosen network and having an open end concerning the production of data. Building an accountable information system implies that the activities in the chain are sequentially designed from the specified information needs (Adriaanse and Lindgaard-Jørgensen, 1996). Dynamic information needs require a regular rethinking of the information system. It is essential to add, to cancel, to revise and to update the concept. Feedback is accomplished if the results of the first cycle of activities are used as a starting point for the next cycle.

After this general introduction to the monitoring framework, the investigation of the similarities and differences between ground water and surface water monitoring systems can start. Besides, the question when it is desirable to integrate the monitoring activities for both water systems and when not, will be answered. In the framework of IWM, one can question the need of integration of different monitoring systems for surface water and ground water.

SIMILARITIES AND DIFFERENCES BETWEEN MONITORING GROUND WATER AND SURFACE WATER.

Ground water and surface water are both parts of one and the same hydrological cycle. Physically, they are connected to each other. Mostly, ground water, fed by net precipitation, flows after passage through the subsoil again upwards into surface waters. So ground water is generally, within the hydrological cycle, located upstream from surface waters. However, in the lower parts of streams and rivers, in (artificial) lakes and reservoirs and in polders, infiltration downwards flowing from surface water to ground water can also occur.

DIFFERENCES IN THE CHARACTERISTICS OF THE 'STATE' OF THE SYSTEMS

Monitoring systems can differ due to differences in characteristics (shape, physics, chemistry, biology, functions, etc.) of ground water and surface water. The major differences between these water system types, regarding monitoring, are:

1. Physical differences.
   Differences in time scale and spatial scale have consequences for monitoring. For instance, surface water runs fast (m/s) and the water quality changes fast; ground water runs slowly (m/year) and the water quality changes very slowly. Thus, the time required for message material (see the introduction) to pass around the total information circuit and the time required for message material to return to the starting point differ considerably between surface and ground water monitoring systems. The spatial scale of surface water bodies is normally smaller than that of ground water bodies. The flow of surface water is easier to trace than that of ground water. The water body of surface waters is limited in depth and horizontal extension. However, ground water reservoirs have no sharp boundaries except for the phreatic surface at the top. The basis of the ground water systems is hard to define. Horizontally, ground water has no boundaries! The ground water flows from watershed to watershed are therefore hard to define, both in time and in space. Nevertheless, especially in hilly areas, the geomorphology can be helpful to understand the ground water flow direction. So, often, surface water watersheds (e.g. hill tops and rivers) are also ground water watersheds in the subsurface.
2. Differences in water system functions.
   Table 1 presents the most common functions of and the issues related with surface water and ground water systems or both. Ground water serves as a reservoir for water supply (drinking, industrial, cooling, irrigation water). It is also very important to the water supply of vegetation (crop and natural vegetation), therefore to agriculture and nature reserves. Surface water

has, besides the same water supply and reservoir function, many other functions. It is used for all kinds of activities: recreation (swimming, skating, yachting, fishing), commercial fishery and navigation.

3. Surface waters are more susceptible to accidental pollution than ground water systems. The effect may be much faster in surface waters, but the gravity does not have to be. The risk of long term pollution is much higher for ground water than for surface water. Indeed, ground water pollution can be combatted, e.g. by polluted ground water extraction. However, this is expensive.

Due to the different features and characteristics mentioned above, the monitoring variables, the layout and sampling of/by the monitoring networks of both water system types will differ considerably (see Table 1).

Examples of monitoring variables only used in surface water are: Ammoniacontent (macro-invertebrates and fish); biological alarm system; phyto/zoo-plankton; ice cover/thickness; transparency; birds, animals, etc.; sediment load; silt content. Monitoring variables exclusively used in ground water are hard to be found. Monitoring of the ecological quality is mostly limited to surface waters. However, incidentally biological monitoring in ground water takes place for bacteria, crustaceans, etc. (Danielopol and Pospisil, 1991).

SIMILARITIES IN MONITORING SURFACE AND GROUND WATER:

1. Until recently, in The Netherlands, but also in many other countries, the integrated water management concept had not yet been introduced and water management was subdivided into many sectors of interest and over many organisations (Van Engelen and Kloosterman, 1996). Therefore, many monitoring systems of both ground water and surface water deal only with that part of water management for which the organisation was responsible. Water quantity, quality and biological qualities, were monitored separately (Van Geer et al., 1996).
2. In the past, in both kinds of water systems, monitoring networks were developed without a clear identification of information needs based on political or water management objectives and targets. In recent years, a large part of these networks has been evaluated, optimized, information needs have been specified and quantified. So, both types of monitoring systems can use the monitoring cycle, to tailor the monitoring systems by:

   - specifying the information needs, namely the goals and targets related to actual socio-economical needs and demands of a water system, with help of state of the art knowledge of the water system through former monitoring analysis and specific project research (surveys of the water system);
   - the development of strategies for monitoring (what to monitor, parameters, relevant margins (quantified need));
   - network design (how, where, when to monitor? ; layout of the network, way of sampling, sampling intensity in time and space, costs);
   - sampling, laboratory analysis (data acquisition);
   - assessment (transforming the data to information by formatting/linking with the information needs);
   - reporting and information use (for water system research, policy and control/management);
   - specification of new information needs (next phase of policy or after some years).

3. Both types of systems can have the function of water supply (drinking, industrial, agricultural water supply), though design and way of monitoring can differ, due to the differences in water system qualities.
4. Monitoring can take place in ground water as well as in surface water systems for specific issues as: pollution by hazardous substances, organic and inorganic micropollution and for *desiccation of ecosystems and droughts damage to agriculture.*

WHEN TO INTEGRATE GROUND WATER AND SURFACE WATER MONITORING?

The need to integrate monitoring of ground water and surface water depends on the extent processes and variables in ground water and surface water are interrelated. If we take water management measures in the ground water system and these measures lead to substantial changes in the surface water system, connected to this ground water system, and if these changes have an impact on specific functions or issues, then integration of monitoring and assessment of both systems can be useful.
Table 1 summarizes the most common functions (columns) and issues (rows) of surface- and/or ground water systems in The Netherlands. The issues are here synonyms for some problems or needs of the different interests in a water system. The table shows for what functions and issues both types of water systems can be monitored (g+s) and for what functions and issues data of only one water system type are needed (g or s).
Thus, for the functions 'transport of water and solids/solutes', 'ecosystems' and 'agriculture', joint information from both water system types can be useful. This is also the case for the issues desiccation, acidification of ecosystems, water pollution, salinization and drought damage to agriculture.

EXAMPLE 1. DESICCATION OF AQUATIC AND WET TERRESTRIAL ECOSYSTEMS.

Speaking in terms of this paper 'desiccation' can be defined as: 'differences that matter' between actual and target values of relevant hydrologic site variables. For monitoring desiccation it is necessary to monitor abiotic site conditions of ground water dependent ecosystems (wetlands). The main hydrological site factors (indicators) of this valuable semi-terrestrial vegetation for the Dutch situation are: lowering of the average surface water level, lowering of the *average ground water table in spring, decrease in seepage to rootzone and the increase of inlet* of nutrient-rich river water (Gieske et al., 1994).

Issue/ function	**drinking water**	**industrial water**	**transport water/ solids**	**eco-systems**	**irrigation agriculture**	**fishery**	**swim-ming**	**shipping**	**safety**	**hydro power**
water level control	s	g+s	g+s	s	s	s	s	s		
desiccation	g	g	g+s	g+s	g+s					
eutrophication	s	s	s	s	s	s	s			
org. pollution	s+g	s+g		s	s	s				
salinization	g	s	g+s	s	s					
acidification				g+s		s				
pollution with hazardous substances	s+g			s	s	s	s			
erosion/ sedimentation	s		s	s				s	s	s
Landsubsidence	g	g							g	
flooding/drought			s	g+s				s	s	s

Table 1: Functions and issues typical for ground water (g) or surface water systems (s) or of both (g+s).

Consequently, for monitoring hydrological and ecological impact of measures in water management in areas suffering desiccation, one needs information about both the ground water and the surface water system. The relevant margins of these parameters must be determined, before a monitoring strategy is set up. These margins are estimated as follows:

1. Lowering of the average surface water level: changes > 5 cm; with help of this margin it is possible to derive frequency and distance of sampling sites [8]: 24 x per year; geohydrological units/site unit. The unit size depends on uniformity of soil and water management characteristics at a site.
2. Lowering of the average ground water table in spring: changes > 5 cm; frequency, etc.: 365 x per year (> 8 years[1]); local surface water level units/site unit.
3. Decrease in seepage to the rootzone (average annual seepage: change > 0.1 mm/day; change between - 1 and + 1 mm/day) is essential if the ground water table still reaches the rootzone; frequency, etc.: 24 x per year; geohydrological units with seepage/site unit.
4. Change of nutrient and chloride content due to inlet of river water; relevant margins: > 0.1 mg/l changes for nutrient and 10 mg/l for chloride); frequency etc.: 365 x per year; site unit with surface water inlet.

Together with information about the reliability of the sampling, the accuracy of the sampling can be derived and the representative sample area, as a function of the geographical scale of the ecosystem units, which is less than one or several hectares. The choice of number and sites of sample locations depends on the monitoring scale (national, regional or local level). The differences in frequency are due to differences in the dynamics of the system types: monitoring ground water parameters 24x per year versus 365x per year for surface water parameters. By means of time series analysis it is possible to correct the ground water table and seepage values for meteorological influences. Thus it is possible to get an estimate within some years and not only after the eight years of the average change in these variables. Also, scores about the rate of desiccation in some area can be determined annually, by correcting the value of hydrological indicators by means of hydrological time series. If all four variables at one site change simultaneously, the scores are combined, using weighing factors. To be able to design desiccation monitoring strategies and networks, these specifications of information needs are necessary. Today, in fact, this strategy and design are in preparation in The Netherlands. This example shows different aspects of monitoring water systems: conjunctive monitoring in ground- and surface water, usage of the information cycle and the use of models to get short term information, which is distorted by stochastic meteorological processes.

EXAMPLE 2. SEPARATE MONITORING SYSTEMS OF LAKE VELUWE AND ITS GROUND WATER SYSTEM.

Forty years ago an artificial lake, "Lake Veluwe" (Dutch: Veluwemeer) in the central part of The Netherlands, was constructed for geohydrological reasons. Lake Veluwe is situated between sandy hills (up to 100 m above mean sea level) in the south and a newly reclaimed polder (5 m below mean sea level). One of the main water management functions of this lake is to operate as a hydrological buffer against ground water level decline, due to the reclamation of the deep polder Flevoland. The aim was to conserve existing high ground water tables in the shore area south of the lake. Other functions of the lake are navigation, fishery, water supply for agriculture, habitats for aquatic ecosystem and recreation. Eutrophication is a large threat to the functions recreation and ecosystems and is a big issue for the water management.
Table 2 presents the water balance of the lake. The table shows the input and output of water, nutrients and chloride to and from this lake between former coast and empoldered land.

[1] Eight years is the minimum measure time in The Netherlands to determine the standard average yearly ground water depth range [9].

Although for water quantity ground water in- and outflow is very important, for nutrients and chloride ground water plays a minor role. For more than forty years monitoring of the water quantity of surface- and ground water has taken place in and around the lake. Although the monitoring systems had been constructed by one governmental organisation (Rijkswaterstaat), monitoring was not integrated in one operating system. After the introduction of integrated water management monitoring has not been integrated, either.
The surface water quantity monitoring system forms a part of the larger regional network Lake IJssel (Dutch: IJsselmeer). Every day, water levels and discharges are measured from and to the lake. The ground water network is also a part of the Lake IJssel network for water quantity (piezometric head: frequency of measuring: 24x per year) and quality (sampling every five years). There is a regional water quality and biological monitoring network (frequency: 24x per year: chemistry of water and suspended material (incl. phytoplankton). The biological sampling (birds, fish, water plants, zooplankton) is carried out every year extensively, every fourth year intensively. The biological monitoring was started in 1990.
The information from these different monitoring programmes is used for data handling, assessment and model simulation, regarding the functions and management problems (issues) of Lake Veluwe. However, information needs and objectives of the different monitoring systems are not always sufficiently coherent for understanding the issues into detail, for instance eutrophication of the lake. Especially, ground water monitoring was not set up for integrated problems. However, with the help of existing information about the yearly influx of surface and ground water (and nutrient content) in combination with ground water modelling, it was possible to assess the basics of the process. So integration of surface and ground water monitoring of this lake is not necessary.
Generally, monitoring is restricted to one of the system types. Monitoring systems are often organised separately, also when information from both types is useful or required. This is due to the large differences in quality between surface- and ground water systems. Another reason is that the organisation of the ground water management is often separated from the surface water management. However, it can be very useful to integrate information from both types of monitoring systems in the assessment phase (see Figure 2). Then, during the optimization phase of both monitoring systems, the information needs, related to interactions between surface and ground water, must be identified and quantified and included in the respective monitoring systems. For this case for instance it was recommended to monitor the nutrient content of the ground water especially in the seepage areas near the lake and the infiltration areas below the lake, for the benefit of a better nutrient balance of Lake Veluwe.
However, till now this tuning of different monitoring programmes of different water systems with the integrated information needs is not at all standard.

	water balance in million m³/year		**phosphorus tons/year**		**nitrogen tons/year**	
	in	**out**	**in**	**out**	**in**	**out**
high land	51	0	11		150	
old polders	39	10	9		130	
new polders	205	0	18		1023	
precipitation	34	0	3		107	
evaporation	—	25			0	?
ground water	**30**	**91**	**2**	**?**	**21**	**?**
other lakes		250		40		820
sewage treatment plant.	17	0	15		269	
total	376	376	58		1700	

Table 2: Water and nutrient balance of Lake Veluwe.

CONCLUSIONS

- Similarities between monitoring surface water and ground waters are:
  - Surface water and ground water are parts of the same hydrological cycle.
  - Often both monitoring systems deal only with a part of the water management (single or few purpose(s) monitoring, instead of integrated monitoring). Today there is a tendency for more integration (within the framework of integrated water management).
  - In the past no clear information needs were specified. In recent years optimization has been started for both systems.
  - Both systems can use the monitoring cycle to tailor the monitoring systems.
  - Surface water and ground water have only one function, water supply, and only some issues in common: pollution and drought threats to ecosystems and agriculture.

- Differences between monitoring of surface and ground waters:
  - Physical characteristics of the subsystems are quite different:flow velocities and reaction time in surface water systems are much faster, occurrence and shape in surface water are more discrete, surface water is visible, surface waters comprise most of the aquatic ecosystems, biology of ground water is of minor importance.
  - Surface waters have many different functions, ground water has only a few.
  - Water systems with the same functions can have considerably different measuring devices, layout of measuring locations, the same measuring variables and monitoring frequencies.

- Monitoring systems in ground water and surface water are seldom integrated in one system. Mostly, only one or few aspects of this water system type are taken into account (single purpose monitoring).

- The need to integrate monitoring in ground water and surface waters depends on how far processes and variables in ground water and surface water are interrelated. However, even then it is often sufficient to gear the information needs for both systems to one another and to integrate the assessment. Physical integration, also for practical reasons, is not recommended.

- For the functions 'transport of water and solids/solutes', 'ecosystems' and 'agriculture', joint assessment of information from both system types can be worthwhile. Especially if the issues are: desiccation, salinization, eutrophication, acidification of ecosystems, and macro- and micro-pollution, and drought damage to agriculture.

- Monitoring and assessment are essentially transmitting 'differences' of characteristics of a water system, relevant to develop a water system to an ideal or a (socially or economically) preferred situation.

- Monitoring can be seen as part of a complex cybernetic circuit in which many actors and characteristics are involved: water system goals and targets, water manager, public interest, commercial interests, nature conservation interests, etc.; see Figure 1.

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THE VALUE OF WATER QUALITY INFORMATION FOR NATIONAL ENVIRONMENTAL POLICY

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ABSTRACT

Scientific studies of water quality can provide answers to four questions essential for managing and protecting water resources in a cost-effective manner:

1. *WHERE are water quality problems most likely, and most severe?*
2. *WHEN are water quality problems most likely, and most severe?*
3. *WHY does water quality differ between areas and times? and*
4. *HOW can monitoring programs be tailored to be most efficient?*

Answers to these questions emerging from the U.S. Geological Survey's National Water Quality Assessment (NAWQA) Program provide examples of the value of water- quality information to national environmental policy. Knowing which factors control differences in concentration is the key to understanding how to tailor future monitoring, and to addressing policy-relevant issues.

BACKGROUND

Environmental laws such as the Clean Water Act and the Safe Drinking Water Act in the United States have been implemented over the last 20 years in order to provide water to citizens that is safe for drinking, swimming, and fishing. Some of the protective measures which have been utilized for this purpose — nationally consistent regulatory limits for a suite of possible contaminants, in particular — are now being called into question.

A first question is whether the suite of contaminants being investigated can be shortened without sacrificing protection of human health. Consumers are not willing to pay for protection from unrealistically small risks of contamination. However, they would often choose to increase payments for increased protection from known threats. Natural contaminants such as arsenic are present in large concentrations in some locations, but not in others. Anthropogenic contaminants also vary geographically, due to differing historical patterns of release or use. So the suite of contaminants posing a risk to health will not be the same everywhere. Determining which contaminants are likely to be present in large amounts, and focusing on those in a monitoring program, provides greater protection for less cost than measuring "everything everywhere".

A second, related question is whether different levels of protection can be adopted based on the geographic variation in susceptibility to contamination. Some areas have soils, rock types, or other conditions that make them more susceptible to contamination than others. These areas are at greater risk should contaminants be introduced. Accurately understanding and then mapping changes in susceptibility allows areas of greater risk to receive greater protection.

In short, it is conceivable to tailor protection strategies so that attainment goals are not sacrificed, by accounting for the geographic and temporal variations in both contaminant patterns and susceptibility of the environment. However, this requires a substantial amount of information in order for tailoring to occur. The understanding provided by properly-designed scientific programs such as the U.S. Geological Survey's National Water Quality Assessment (NAWQA) Program can form the basis for this tailoring. The goals of the NAWQA Program are to:

1. Describe current water-quality conditions for a large part of the freshwater streams, rivers, and ground-water aquifers of the United States. ***Where?***
2. Describe how water quality is changing over time. ***When?***
3. Improve understanding of the primary natural and human factors that affect water-quality conditions. ***Why?***
4. Determine how these results can improve the management of water resources. ***How?***

Monitoring itself is usually not designed to address questions of spatial and temporal variation, but instead focuses on determining how frequently standards are violated, or on other statistics of contaminant concentrations for a geographic region. A scientific assessment provides much more. By addressing the "where, when and why", an assessment provides guidelines for the "how" of tailored monitoring strategies. Examples from USGS studies follow.

WHERE ARE PROBLEMS MOST LIKELY? MOST SEVERE?

Determining where water-quality problems are most likely to occur allows protection strategies to differ in different locations. Funds for protection, regulation, and further monitoring can be prioritized and spent on more critical areas and issues first. This can operate at a variety of scales. For example, the two studies which follow, a statewide and a national study, can be used to tailor less-frequent monitoring in locations less at risk to contamination.

STATEWIDE STUDY – STATE OF WASHINGTON

Ryker and Williamson (1996) studied the percentage of public supply wells with detectable levels of pesticides in the state of Washington. US law requires each public supply well with 15 or more connections to be monitored quarterly for pesticides, but allows the state to issue waivers based on evidence of low risk. Costs of monitoring in Washington were estimated to be $1100 per well per year, a considerable burden on households in small systems. Detections varied as a function of land use at the surface, the depth of the well, and the nitrate concentration in the well (reflecting contamination from the surface). These criteria were used to classify wells into three risk groups. Low risk wells were granted a full waiver, monitoring for pesticides only once every three years. High risk wells maintained quarterly monitoring, while medium risk wells obtained a partial waiver. Costs of the sampling and assessment were recovered by the savings resulting from the reduced monitoring schedule within three months of the first year.

NATIONAL STUDY – NITRATE IN GROUND WATER OF THE UNITED STATES

Natural and anthropogenic conditions associated with high nitrate levels in ground water were assessed in 20 large areas scattered throughout the United States (Nolan and Ruddy, 1996). Two predominant factors found to be related to high nitrate were high inputs of nitrogen at the surface, and well-drained soils. Maps of these two factors were overlain, determining four risk groups for high nitrate throughout the United States. Areas with at least 2.1 tonnes of nitrogen per square kilometer and well-drained soils as defined by the U.S. Census of Agriculture exhibited the highest risk. Differences in concentration between risk groups were larger in waters from shallow wells (less than 30 m) than from deeper wells.

Twenty-six percent of shallow agricultural wells in the high-risk group exceeded the drinking-water standard of 10 mg/L nitrate. Only six percent of shallow agricultural wells in the lowest risk group exceeded the standard. Areas where nitrate concentrations are expected to be most severe are clearly identified. These results have been of interest to the U.S. Environmental Protection Agency, who must prioritize areas where remediation and prevention programs are established. Monitoring programs for nitrate could focus more intensive efforts in states or counties in proportion to their risk of contamination. Though smaller-scale studies are necessary to provide the detail needed for most local purposes, a consistent national perspective allows national-scale monitoring to allocate more sampling to areas with the greatest variation, typically those areas with the highest risk of contamination.

WHEN ARE PROBLEMS MOST LIKELY? MOST SEVERE?

While spatial variation is of greatest concern for groundwaters, streams exhibit their greatest variability over time. An old (hydrologic) adage goes that "90% of the sediment is moved during 10% of the time." The highest streamflow, which occurs only during a few days of each year, carries markedly higher sediment concentrations than average or low streamflows. Trace metals, phosphates, and some organic compounds such as PCBs and some pesticides, also move in the same pattern. This has important implications for monitoring strategies. Random or infrequent sampling schemes may completely miss any chemicals or sediment particles moving in this fashion.

How can this complexity be taken advantage of in order to tailor monitoring programs? One example was given by the seasonal sampling strategies of Battaglin and Hay (1996). Herbicides such as atrazine and alachlor are applied in great quantities in the spring on land of the midwestern United States. They are found in streams in sufficient amounts and frequencies to cause concern for human health. Their maximum contaminant levels (MCL), above which some enforcement actions may be taken, have been defined as annual mean concentrations. Rules for monitoring (U.S. EPA, 1991) state that a minimum of four quarterly samples are to be taken in order to compute the annual mean. Stream concentrations of herbicides are not evenly distributed throughout the year, but are accentuated greatly in the spring. Battaglin and Hay show that over 40% of annual means based on quarterly samples underestimate annual mean herbicide concentrations. Instead, three samples taken in the spring averaged with 9 zero concentrations for the remainder of the year, provides a much more accurate estimate of the annual mean. Indeed, this three-sample method provides estimates almost as accurate as 12 monthly samples, for considerably less expense. This is a simple example of the statistical principle of sampling more frequently during periods of greater variability, combined with knowledge of how a stream system works. The result is a more efficient sampling design than is a monthly or quarterly program which presumes no prior knowledge of the system.

WHY DOES WATER QUALITY DIFFER BETWEEN AREAS AND TIMES?

In any monitoring program, only a small number of samples can actually be collected and analyzed. Some method must be employed to relate these data to the entire population of interest. One method is to assume that the statistics generated are applicable to the entire area. Without a knowledge of where and when, however, averages computed may be far from the truth for any specific region. Bricker and Rice (1989) provide an example of the benefits of incorporating the "where " and "why" of water quality into a sampling design. Their objective was to compute the mean and standard deviation for the acid-neutralizing capacity (ANC) of streams in western Maryland. As this characteristic is known to be caused by the carbonate content of rocks through which the streams flow, they based their sampling design on the geology of the region. Locations with more carbonate were expected to have high ANC, and lower variability, and so were sampled less frequently than their surface area would dictate. Two benefits resulted from their approach. First, their estimates of mean ANC were less biased, and had lower variance, than those using an areal grid approach with the same sample size. Second, by relating ANC values to rock type, they produced estimates for any location in their study area, rather than being limited to the mean for the entire area as their best prediction of ANC.

Understanding why problems are more severe in certain areas, or at certain times, has a second important advantage - it leads to possible solutions. Without hard scientific information, monitoring is little better than a physician tracking a patient's decline, with no understanding of how to treat the disease. An assessment of cause is like a diagnosis that also leads to improved and less-expensive methods of monitoring the patient's progress. Expensive bone-marrow samples are not warranted for routine infections.

Therefore in order to tailor monitoring programs, there needs to be a baseline of assessment activities to evaluate and diagnose. For example, the assessment of pesticides in Washington State's groundwater cost 1.4 million dollars U.S., a considerable investment. However, the monitoring savings realized were estimated by the state agency at 18.0 million dollars over a three-year period. The key to tailoring their monitoring was an answer to "why", why some wells were vulnerable and others not. Legislators and others who appropriately ask "what additional value is there in more water quality measurements after all these years of effort?" need to see demonstrations that the answers to "why" can provide large savings in tailored monitoring of "where" and "when".

HOW CAN PROGRAMS BE TAILORED TO BE MORE EFFICIENT?

Scientific assessments must provide a base of regular sampling over space, time, and constituent coverage in order to understand how contaminants behave in hydrologic systems. Built into these assessments must be the explicit goal to understand the "why" of contamination, and to communicate this scientific information to policy and regulatory officials. With this effectively communicated, efficiency is gained by sampling more frequently when and where the greatest uncertainty exists and the costs of making an error in judgment are the highest.

THE FINAL LINK — COMMUNICATING WITH POLICYMAKERS

Scientists measure and interpret complex systems. In particular, the natural sciences must deal with uncontrolled variation of driving factors such as weather, temperature, soils, and geology. Environmental studies add to this the effects caused by human behavior, which is if anything less predictable. As a result, great complexity must be studied and "overcome" in order to understand the primary signals present in the data. Scientists, who correctly want to portray the complexity of their results, can often be seen as "always wanting more data" or "irrelevant" to people who want quick answers. There can be a gulf in culture between science and policy.

Policy makers also deal with complex systems, yet are required to make tough "yes/no" choices. They are rarely trained in the disciplines of science and so require concise summaries of results in nontechnical jargon to understand the "bottom line". Scientists are not trained to write this way. They have not often been asked to communicate their results in such a policy-relevant manner. The resulting division is in some respects a problem of translation, where important information fails to be communicated because of barriers of language and style.

The USGS is taking several steps to address this barrier . First, we are producing short summary reports or "fact sheets" of 1 to 4 pages which highlight the relevant findings of a study without jargon or much detail. The nontechnical audience for these reports requires that they have a different tone than longer scientific documents which back up each fact sheet. Second, scientists are having more direct conversations with policy makers. Each NAWQA study unit has formed a "liaison committee" composed not only of outside scientists, but also of users of their information. These committees provide opportunities for direct use of NAWQA data to address local problems and issues. They provide direct conversations between the users and producers of the work. Third, information is being provided to the public more quickly and easily through the Internet. Most fact sheets are available on-line, as are the data produced by the study. Others can see the conclusions reached by the Program, and check the process of generating those against their own by interpreting the data themselves. The NAWQA home page is at: http://wwwrvares.er.usgs.gov/nawqa/nawqa_home.html

CONCLUSIONS

Scientific assessment programs can provide information on where water-quality problems are likely to occur, when they are most likely, and the factors that control them. Knowing which factors control differences in concentration is the key to understanding how to tailor future monitoring, and to addressing policy-relevant issues. As with other societal activities, funding for scientific studies of water quality is increasingly difficult to obtain. Yet the understanding they provide is critical to developing cost-effective and minimally intrusive strategies to manage and protect water resources. Scientists do not often "speak the same language" as the people who would use their information for making policy decisions. Scientists write for scientific rather than for nontechnical audiences. In order for the transfer of information from science to policy to occur, scientists must make policy-relevance an explicit objective of the work they do.

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MONITORING FOR GROUNDWATER QUALITY ASSESSMENT: CURRENT CONSTRAINTS AND FUTURE STRATEGIES

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ABSTRACT

The characteristics of subsurface flow regimes exert important controls on groundwater quality and strongly influence pollutant transport and behaviour. Assessment of groundwater quality must reflect adequate understanding of these regimes, in terms of both monitoring network design and the interpretation of the resulting data. The danger of misinterpreting results is highly dependent on the scale of the heterogeneity of aquifers and the spatial and temporal patterns of pollution in relation to sampling installations and network density. These problems are described and illustrated, and strategies for overcoming them are briefly described, drawing on recent UK experience in national and global progammes.

INTRODUCTION

Monitoring and assessment of groundwater quality over large areas faces both conceptual and technical difficulties. Groundwater occurrence is complex, in particular having a depth dimension which is often problematic to evaluate. Further, the timescales of groundwater movement and residence are relatively long. Lastly, groundwater bodies are, by definition, beneath the land surface and consequently not very accessible for sampling. It is common, therefore, to find that groundwater quality is less effectively assessed than surface water quality.

For these reasons, it is important to be aware of the hydrogeological context within which regional assessments of groundwater quality are undertaken or proposed. This importance is illustrated in the present paper, drawing on recent experience gained by the British Geological Survey (BGS) at two different geographical scales. Firstly, the Survey has prepared for the UK Environment Agency (EA) a national strategy for groundwater quality assessment, and secondly is providing technical support to the WHO/UNEP Global Environmental Monitoring System (GEMS) water programme to strengthen its groundwater component at a global scale. While there is no formal link between the two separate activities, they share many constraints, and the similar strategies which can be devised to overcome such constraints form the basis of the following discussion.

CHARACTERISTICS OF GROUNDWATER BODIES

In the UK, the broad characteristics of groundwater flow regimes are determined by the regional stratigraphy and geological structure. These together define the locations of outcrops of permeable formations and their lower-permeability confining beds, and hence the distribution of areas of groundwater recharge and discharge. There are major differences in groundwater flow regime between the UK's major aquifers which are consolidated formations such as limestones and sandstones, in which groundwater movement is more or less restricted to major fractures, and formations in which movement is dominantly through the intergranular spaces of unconsolidated sediments such as alluvial sands, silts or gravels of major river flood plains and deltas. Both types of groundwater flow regime are common in different parts of Europe, and rightly play an important role in defining national approaches to groundwater quality monitoring in, for example, the Netherlands (van Duijvenbooden, 1993), Germany (Jedlitschka, 1997) and Spain (Estrela & Varela, 1997).

Two general features of groundwater bodies distinguish them from surface waters. Firstly, the relatively slow movement of water through the ground means that residence times in groundwater can be orders of magnitude longer than in surface waters (Fig 1). Once polluted, a groundwater body could remain so for many years because the natural processes of through-flushing are so slow. Secondly, there is a considerable degree of physico-chemical interaction between the water and the containing aquifer. The properties of the aquifer material and the water are important, and the scope for quality to be modified by interaction between the two is enhanced by the long residence times.

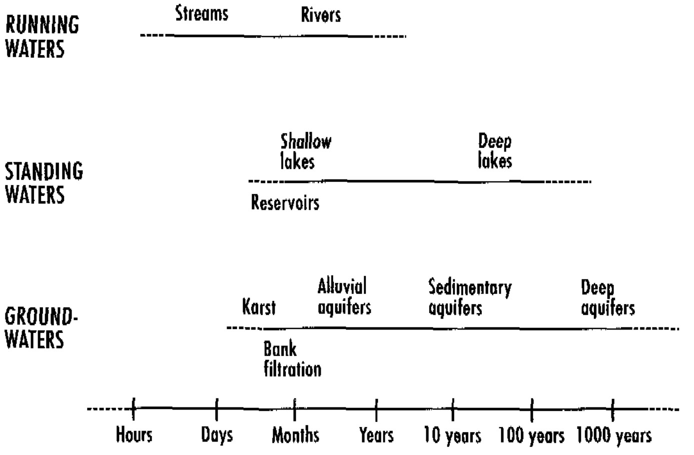

Figure 1: Water residence times in freshwater bodies. *Source after Meybeck et al., 1989.*

Thus, the natural hydrochemistry evolves significantly as infiltrating water moves down groundwater flow lines, away from the outcrop recharge zone and deeper into the major aquifers as they dip beneath confining, lower-permeability materials. As the water moves, reactions which include dissolution, oxidation and reduction, ion exchange and precipitation modify the major ion and trace constituent composition of the groundwater (Edmunds et al., 1987). As a result, even the "baseline" quality of groundwater is subject to spatial and depth variations which must be appreciated before the superimposed impacts of human activities can be detected and quantified. Groundwater quality is thus infinitely variable in space and time, but on different scales to that of surface waters, and this variability is made more complex by the interactions referred to above.

SAMPLING CONSTRAINTS

There are formidable obstacles to achieving ideal sampling which are technically difficult and costly to overcome. Serious sampling limitations often have to be accepted, and it is important that these are fully recognised in the interpretation and application of the results of monitoring. Two distinct types of limitation are important. The first relates to physico-chemical modification of the sample as it is drawn from the aquifer and brought to the ground surface. Features of the monitoring installation and the method of sampling can produce interpretation difficulties resulting from the instability of some analytical determinands. Stability problems during groundwater sampling result from volatile losses, adsorption, oxidation and precipitation. Classifying groups of determinands according to their relative instability and required detection limits (Fig 2) shows that, in general, the more stable inorganic determinands occur, have use guidelines at, and need to be measured in, the mg/l range. Determinand instability increases for the organic compounds, as the required level of detection moves down to the µg/l range. Figure 2 can be used to indicate which groups of determinands require special precautions in sampling. Thus, for those constituents which occur in groundwater in the mg/l range, sample contamination is not generally of great concern. For constituents which are significant in the µg/l range, the problem of sample modification becomes more acute.

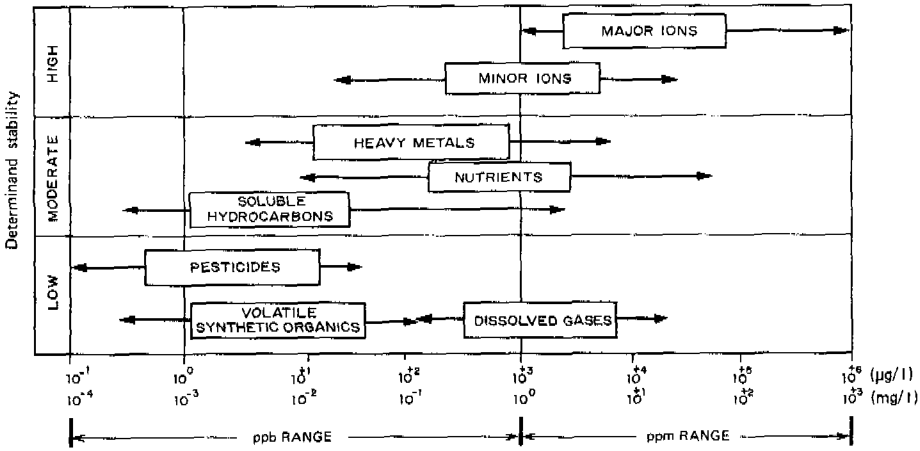

Figure 2: Concentration ranges and relative stability of determinand groups. *Source after Foster & Gomes, 1989*

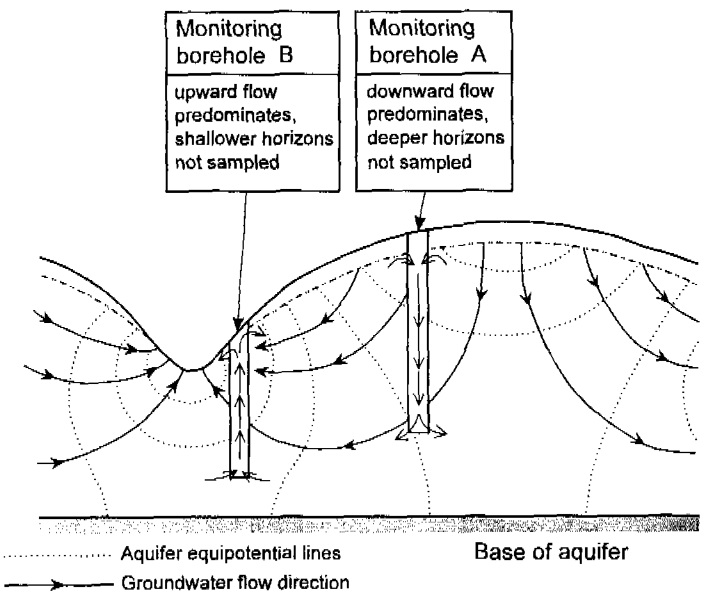

Figure 3: Schematic cross-section of typical groundwater flow system to illustrate the effects of vertical components of flow. *Source after Foster & Gomes, 1989.*

The second type of limitation results from the hydrogeological complexity referred to above. It can be difficult and costly to obtain samples of groundwater which are adequately representative of the three-dimensional picture of the quality of groundwater. Quality may vary greatly over very short distances vertically and laterally because of the scale of complexity in geology and groundwater flow, both horizontally and with depth. In both recharge and discharge areas, vertical components of groundwater flow may mean that simple, fully-penetrating boreholes provide misleading indications of overall groundwater quality (Fig 3). In other usually simpler hydrogeological settings vertical flow components may be less important, and single samples can provide the overall indication of groundwater quality that will adequately meet many monitoring objectives.

Groundwater in the upper part of an aquifer is likely to show the first signs of deterioration in quality resulting from human impact. Monitoring of quality based only on samples drawn from deep abstraction boreholes may not show this impact, even though the upper parts of the aquifer or the recharge zones may already be polluted. Once this pollution reaches the deep abstraction boreholes, much of the aquifer may be affected, and remediation or other quality management actions may be difficult and very costly. An early warning approach for groundwater monitoring would thus require shallow boreholes in the recharge area to detect the influence of diffuse pollution sources before deeper public supply sources are affected. The use in the Dutch national monitoring programme of purpose-drilled boreholes to sample shallow, intermediate and deeper groundwater in a thick alluvial sequence recognises the potential for quality stratification and the need for early warning of the effects of human impacts (van Duijvenbooden, 1993). Finally, because of the slow movement and mixing, sharp differences in quality can exist over short lateral distances within the same aquifer, and regional-scale assessment programmes are unlikely to be able to achieve a sample point density which can adequately reflect these differences.

LIMITATIONS OF EXISTING MONITORING

Some years ago, Wilkinson and Edworthy (1981) identified four main reasons why groundwater monitoring systems fail to provide adequate data:

- the objectives of monitoring are not properly defined
- the monitoring system is installed without sufficient hydrogeological knowledge
- there is insufficient planning of collection, handling, storage and analysis of samples
- data are poorly archived.

Despite these inadequacies, data from monitoring programmes are often used to make predictions of future groundwater quality and as a basis for decisions on major capital expenditure. Although many of these limitations apply equally to surface waters and groundwaters, a particular problem of groundwater monitoring programmes is that they have been established as an extension of, and sometimes by the same people as, surface water networks. They have often evolved, therefore, without the required understanding of groundwater occurrence and movement.

One of the most important limitations in monitoring-based groundwater quality assessment programmes is that of mismatch between network coverage and objectives. This is illustrated by the GEMS/WATER programme, the current objectives of which can be summarised as (WHO, 1991);

- to provide to governments, the scientific community and the public, assessments of the quality of the world's freshwaters in relation to human and aquatic ecosystem health and global environmental concerns by identifying and quantifying trends in quality, define the causes and impacts and provide the information in forms that can be used to evaluate policy options for maintaining or restoring quality.

- to provide as above assessments of the fluxes of pollutants from the major river basins to the oceans.

- to strengthen national water quality networks.

These are clear but ambitious objectives, and to meet them, three types of sampling point were defined:

- Baseline, in small, undisturbed basins without human activity and more than 100 km from major sources of atmospheric pollution such as cities and industries.

- Trend, to observe human impacts on water quality, in medium-sized catchments which respond on a moderate timescale to pollution and changes in land use.

- Global flux, to determine the fluxes of pollutants at the continent/ocean interface.

The last mentioned clearly does not apply to groundwater, and the first is immediately problematic. Sampling from very deep aquifers, such as the Nubian Sandstone in North Africa could produce water unaffected by human activity, but the water itself would be thousands of years old. At the other extreme, shallow aquifers containing very recent groundwater are likely to be highly vulnerable to local sources of pollution, and equally unsuitable to provide baseline information. For lakes and rivers, it was envisaged that baseline sampling could be carried out in small headwater catchments in the upper reaches of river systems, remote from human settlement, although even this is becoming increasingly rare. For groundwater, such locations are not easy to find. The upper reaches of catchments are generally dominated by erosional rather than depositional processes and any alluvial sediments are likely to be small in extent and shallow, such as river terraces and fluvio-glacial deposits. These are likely to be easily polluted and not regionally representative. The groundwater component of GEMS/WATER, therefore, inevitably focusses on the trend category and the assessment of human impacts.

The present coverage of groundwater sampling points in the GEMS/WATER programme is, however, quite inadequate to provide information to meet the ambitious objectives stated above, even if all of the currently-included locations were suitable. Large areas, and hence major aquifer systems, of the world are currently unrepresented in the GEMS/WATER programme, even where there are local and national monitoring activities. Because of the vertical and lateral complexities in hydrogeology and groundwater quality outlined above, regional or global quality assessments face great difficulties in achieving adequate network coverage. Further, management responses to groundwater quality deterioration, and the pollution control measures which might need to be taken, are likely to be required at a much smaller geographical scale than for major surface waters. For groundwater, therefore, management information resulting from quality assessments may be less essential at the global or regional scale than for surface water.

The response to this significant mismatch for groundwater between objectives and network coverage may be that the objectives should be made less ambitious as well attempting to strengthen the data coverage. This remains a current challenge for GEMS/WATER, as the case for maintaining groundwater as an integral part of the programme is strong for two reasons. Firstly, groundwater is a major contributor to surface water resources and wetlands by baseflow, and secondly groundwater provides vitally important domestic water supplies for hundreds of millions of people globally, often in situations where there are no economic surface water alternatives. Groundwater must remain an important component of GEMS/WATER, and needs to be strengthened significantly.

STRATEGIES FOR IMPROVED ASSESSMENTS

As in all assessment programmes, the objectives and information requirements must be well defined from the outset. Many authors in the literature (eg; Canter et al., 1987; Foster & Gomes, 1989; Chapman, 1996), have emphasised this. It is, however, often difficult, as the data collected in monitoring must usually meet the multiple objectives and information requirements of more than one organisation. This certainly applies in the UK where, for national groundwater quality assessment, the objectives are a function of the legislative and institutional framework, the use to which the water is put and the actual or potential impacts on water quality. The primary objectives defined in Table 1 provide the information base needed to manage groundwater quality in the UK in a sustainable way, and to implement European and national legislation. In this way, complex multiple objectives of assessment were "tailor-made" in the proposed strategy to meet the UK's national requirements.

Primary Objective	Information Output
1. Trends	Show trends of groundwater quality changes derived from natural causes, the impact of diffuse pollution sources and changes in hydraulic regime.
2. Baseline for future issues	Provide background information on groundwater quality so that the impacts of future, as yet undefined, human activities can be detected.
3. Spatial distribution	Provide a picture of the three-dimensional distribution of groundwater quality within aquifers.
4. Early warning	Provide early warning in recharge areas on aquifer outcrops of the impacts of diffuse sources of pollution.
5. Monitoring of nitrate	Provide information to meet the requirements of the EC Nitrate Directive to identify Nitrate Vulnerable Zones.

Table 1: *Primary objectives of national groundwater quality assessment.*

Having established the primary objectives of quality assessment, the next step was to design an appropriate national network, which was essentially a function of the selection of sampling point type, density and location, sampling method and frequency and choice of determinands (Table 2). In practice, they are not independent variables because, for example, type of sampling point and method of sampling are closely linked, as are sampling frequency and choice of determinands. Cost is a major factor, always constraining the approach to groundwater monitoring, and appears at the foot of each column in Table 2. Because of the relatively slow groundwater movement but high degree of spatial variability referred to earlier, most information requirements from groundwater quality assessments would be better met in existing monitoring programmes by less frequent sampling of a higher density network.

Sampling point		Sampling frequency	Choice of determinands
Type	Density		
Primary assessment objectives	Primary assessment objectives	Primary assessment objectives	Primary assessment objectives
Hydrogeology (complexity)	Hydrogeology (complexity)	Hydrogeology (residence time)	Water uses
	Geology (aquifer distribution)	Hydrology (seasonal influences)	Water quality issues
	Land use		Statutory requirements
	Statistical considerations	Statistical considerations	
Costs	Costs	Costs	Costs

Table 2: Factors which determine network design.

Other key features of the network design were consideration of aquifer vulnerability in choice of sampling point, modifying sampling frequency to take account of aquifer type and allowing choice of determinands to reflect major land use types. Thus, it was proposed that shallower and thinner aquifers with shorter residence times would be sampled more frequently than deeper and thicker aquifers, and confined aquifers would be sampled less frequently than unconfined. Again the strategy was customised to meet national conditions in the UK.

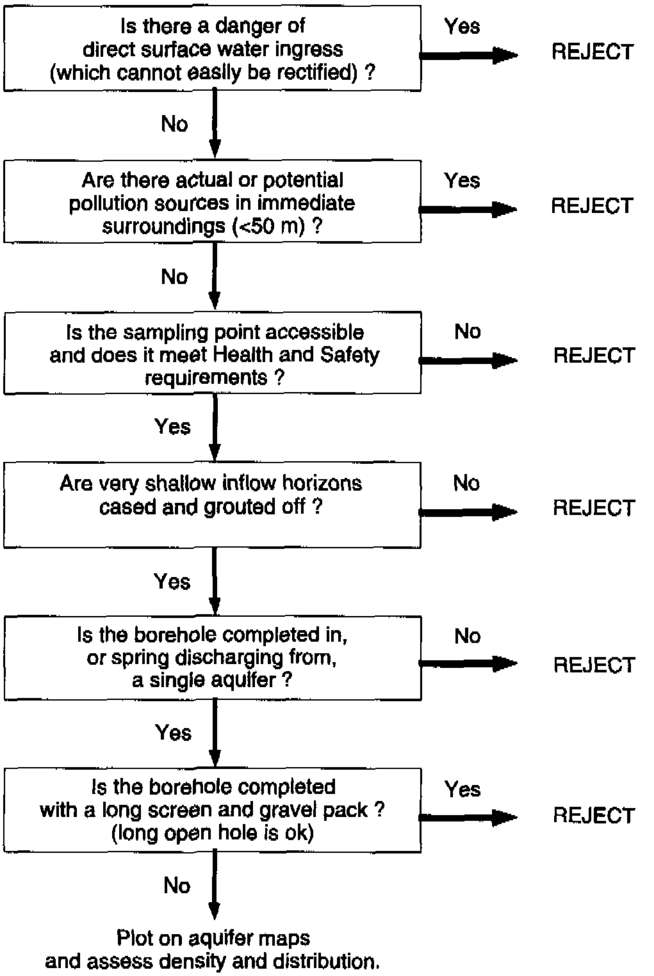

Figure 4: Algorithm for evaluation of abstraction boreholes and springs

The outcome is a proposed tiered monitoring system for groundwater comprising:

- A Reference Network of 150 to 250 sampling points at a mean density of 1/250 to 1/400 km^2, selected from within the National Network

- A National Network of about 3000 sampling points at a mean density of 1/25 km^2

- A Local Network for point source impact monitoring

It should be emphasised that these are mean target densities rather than a fixed grid approach. Further details of the proposed national network are given by Chilton et al., (1995), and in a poster presented at the workshop.

An important desire in improving assessment of groundwater quality in the UK was to ensure that adequate use was made of the existing data and sampling points. As in most such cases, financial considerations limit the number of new groundwater sampling boreholes that can be constructed. The existing network was reviewed and, as part of the process of change from the old to the new network a formal ranking procedure was developed for evaluating the suitability of existing sampling points. A two stage approach has been proposed;

- algorithms to deal with those criteria, principally related to site and location, for which yes/no answers could be used to place sites in or out of the network (Fig 4).
- a numerical procedure to rank and categorise the sampling points accepted by the algorithms (Fig 5).

It was proposed that each factor in the top boxes in Figure 5 be scored from 0 to 1, and then multiplied together to provide a factor for each of the four categories of information about each site. These values would in turn be multiplied together to provide the final ranking factor. Currently this approach to ranking remains a proposal to be refined and implemented by the UK Environment Agency.

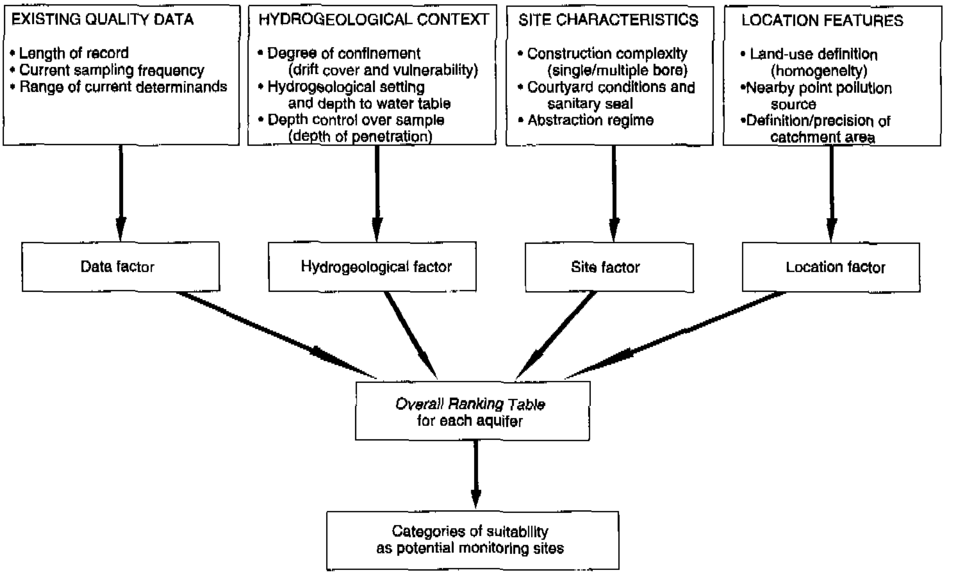

Figure 5: Ranking procedure for evalution of existing monitoring sites.

In the United States, recognising the potential inadequacies of fixed point, fixed interval sampling in providing assessment information for past impacts on groundwater quality, the US Geological Survey has developed the National Water Quality Assessment Programme (NAWQA) of rotating study units (Alley & Cohen, 1991). Within the study units, which are representative of river basins and aquifers within the US, intensive assessment activities are being undertaken on a rotational rather than a continuing basis, with one third of the units being intensively investigated at any one time. For each study unit, 3 to 5 year periods of intensive data collection and analysis will alternate with 5 to 6 year periods of less intensive monitoring. In addition, national assessments will be made of a limited number of high priority water quality issues. Further details of the programme are given in these proceedings by Helsel (1997).

To meet its objectives of providing regional and global assessments, the GEMS/WATER programme has two possible options for groundwater. Firstly, it could do this by synthesis of existing national and other technical reports, much as was done for Europe for the Ministerial Meeting in the Hague (RIVM/RIZA, 1991) or for the Dobris Assessment (EEA, 1995). This approach was adopted by GEMS/WATER for its first global freshwater assessment (Meybeck et al., 1989). The second possibility is to adapt the general NAWQA approach by choosing limited, representative areas for reasonably comprehensive monitoring and assessment. These areas would be selected globally to cover the range of hydrogeological environments, climates and land use types, and would thus provide representative information on the range of groundwater quality issues and trends, which could be used to provide regional or global assessments.

A preliminary and incomplete attempt to do this is shown in Table 3. Assuming the matrix of aquifer types and land use could be reasonably well agreed upon, the question remains as to how many groundwater sampling points would be required to provide reasonable coverage of the representative regions selected, bearing in mind the problems of lateral and vertical heterogeneity in groundwater quality referred to above. Looking at Table 3, some of the combinations of land-use category and aquifer type may be relatively uncommon on a global scale. Nevertheless, there could be some twenty combinations that are significant globally, and some could be sufficiently dominant that more than one type area would be required to meet the overall objectives. If these could each be represented effectively by some 10-20 sampling points, then a minimum of perhaps 400-500 sampling points would be required to provide a much improved capability to meet the GEMS/WATER objectives set out above.

There is currently considerable interest in the possible use of indicator parameters in groundwater quality assessment. The concept is attractive because of the possible savings in analytical costs and the presentational simplicity they might bring. Indicator parameters such as biochemical oxygen demand, dissolved oxygen (DO), ammonia and faecal coliforms have been used for many years in surface water monitoring. They are used as the basis for various indices of river

	Aquifer Types/Hydrogeological Environments					
Principal Land-Use Categories	Large Alluvial Basins	Deltaic Alluvium	Sandstones Limestones	Karstic Limestones	Intermontane Basins/Volcanics	Basement
Temperate agriculture	Rhine, Eastern USA, Danube	Po	Chalk UK, France Ogallala			
Temperate irrigated agriculture	China	Catalunya	Ogallala			
Temperate forest	Russia			China		Scandinavia
Temperate urban	Eastern USA	Po	Eastern US Chalk, Sandstones UK			
Temperate industrial	Poland N Europe Canada	Po	Czech UK, Eastern US			
Tropical irrigated agriculture	Mexico, Brazil, Sind,Punjab, Indonesia	Nile Ganges Mekong	Brazil (Bauru)	Florida Jaffna	Mexico	Brazil Sri Lanka
Tropical forest/ bush/Savannah	Australia			China		West Africa
Tropical urban	Thailand Indonesia	Calcutta		Yucatan	Santa Cruz	
Tropical industrial	Thailand Brazil					
Wastewater irrigation	Tunisia				Mexico	
Mining	Brazil					Canada
Dryland farming or rangeland		Iran Iraq	Saudi Arabia, Jordan		Mexico	

Table 3: Possible type combination of land use categories and aquifer types.

Processes	Primary indicator(s)	Secondary indicator(s)
Saturated zone		
Physical:		
Piezometric change	Water level	
Geochemical: Natural hydrogeological processes:		
Mineral dissolution	HCO_3	Si, $Si_{calcite}$ major ions
Redox reactions	O_2	Eh, Fe_T
Salinity	Cl, SEC	Mg/Ca, $\delta^{18}O$, $\delta^{2}H$, Br
Residence time		^{3}H, ^{14}C, trace elements
Geochemical: Anthropogenic pollution (diffuse):		
Environmental radioactivity	^{3}H	^{36}Cl, ^{14}C
Agrochemicals	NO_3, DOC, HCO_3	K, pesticides
Industrial and urban impacts	Cl, DOC, HCO_3	B
Unsaturated zone		
Physical processes:		
Recharge rates	Cl	^{3}H, ^{14}C, ^{36}Cl
Geochemical processes and pollution:		
Acid attenuation	pH	
Pollution	NO_3	

Table 4: Possible indicators of rapid environment change in groundwater systems. *Source after Edmunds, 1996.*

quality and in setting and evaluating Water Quality Objectives for rivers. Faecal coliforms are used as the classic indicator of the bacteriological quality of waters. For groundwater, the use of indicators to provide a general picture of the "health" of an aquifer is attractive but at the present time it is not easy to choose suitable indicator parameters for groundwater. Possible candidates include electrical conductivity, dissolved oxygen, pH, alkalinity, chloride and nitrate, and possibly total or dissolved organic carbon (TOC/DOC). A preliminary review of the scope for the use of indicators in groundwater systems was given by Edmunds (1996), from which Table 4 is drawn. Further constraints to the use of indicators are provided by the multi-objective nature of most groundwater quality assessments and the dominance of potable supply use; the statutory and legal framework within which most assessments are undertaken mean that extensive suites of determinands are usually required. The only objective of groundwater quality assessments which has been successfully met in some instances by an indicator is the use of electrical conductivity to monitor saline intrusion. Nevertheless, it can be anticipated that interest in the use of indicators for rapid assessment of groundwater quality will grow.

CONCLUSIONS

Approaches to the assessment of groundwater quality are invariably a function of the objectives and the financial resources available. Without clear definition of the former, viable assessment strategies cannot be designed. As in all environmental monitoring programmes, it is not possible to measure everywhere all of the time, and difficult compromises have to be made. The objectives of groundwater quality assessments are often not properly formulated in respect of the information requirements, and there is often a mismatch between assessment objectives and network coverage. The objectives and available resources must be linked to the additional key factors of hydrogeology and land use patterns, both of which can be highly complex.

Groundwaters are distinguished from surface waters by longer residence times and by the scope for interaction between water and aquifer material. The complexity of most groundwater bodies and the risk of physico-chemical modification as the groundwater is drawn to the surface together place major constraints on sampling. These limitations must be properly understood in the interpretation of the results of groundwater monitoring.

On a national scale, the design of networks for groundwater quality monitoring rarely starts from nothing. Appropriate use must be made of existing data and sampling points. Formal sampling site evaluation procedures may be required, and a formal ranking approach has been proposed for the UK, so that only the most suitable of the existing sampling points are selected for an improved network. The proposed strategy for the UK outlined in the paper is currently under consideration by the national Environment Agency. On a regional or global scale, approaches to assessment using a selection of representative land-use and aquifer type classifications are considered to offer the best potential.

The use of indicators is less well developed for assessing groundwater quality than for surface water. The successful development and application of indicators would be highly attractive, but at the present time the scope for such development appears limited.

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GROUNDWATER MONITORING IN GERMANY

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ABSTRACT

Groundwater monitoring plays-an important role in the ECE-Charter on Groundwater Management (1989), in the Groundwater Action Programme of the European Union and in the German Requirements for a Progressive Groundwater Protection. As Germany is a fédéral country the 16 Federal States are responsible for water problems including groundwater monitoring.

The objective of the German groundwater monitoring network is to get a sufficient knowledge of the actual and future groundwater conditions. This is achieved by a overall state-run basic monitoring network complemented by user and emission related monitoring points. Details on operation, parameters, frequencies and costs in Germany's groundwater is given.

GENERAL

For a long time, groundwater has been taken for a well protected resource. The reason for this were the belief in self-purification of the soil and as a rule the protection of groundwater by the covering layers.
Damages to the groundwater (point sources from storage or transport of hazardous substances, landfills, etc.) and diffuse contamination as from agricultural sources have caused a general change of mind.

Groundwater has to be protected generally as it forms a principal source for drinking water (more than 70 % in Germany), as it is interconnected inseparably with surface water and as it represents also a precious ecological part within the balance of the water cycle and is essential for above ground ecosystems.

The priority of groundwater protection is actually reflected in

- the German Requirements for a Progressive Groundwater Protection Policy in the European Community
- the ECE-Charta on Groundwater Management (1989) and
- the Ministerial Seminar of the Environment Ministers of the European Union in The Hague, Nov. 1991, which turned out into a proposal of the Commission for an Action Programme for Integrated Groundwater Protection and Management from 10.07.96.

In all three documents the monitoring of groundwater plays an important role, as it is the knowledge of the actual state of groundwater (quality and quantity aspects) which enables us to take measures against pollution.

GROUNDWATER QUALITY MONITORING NETWORKS

In Germany the responsibility of water lies with the 16 Federal States (Länder). The Federal Water Act (Wasserhaushaltsgesetz) gives only the frame work. That means that each Federal State has to control its water quality and quantity according to the general rules of the above mentioned federal act. To harmonize the water management in Germany the Federal States have created the Federal States Working Party on Water (LAWA).

This Party issued in 1983 the LAWA-Framework Groundwater Quality Monitoring Programme which is followed in the old and new Federal States.

The Monitoring Scheme consists of 3 types

- basic or reference monitoring network
- user related monitoring
- emission related monitoring

BASIC OR REFERENCE MONITORING NETWORK (BASIC AND TREND MONITORING)

Measuring points of the basic network are hydrologically orientated and of wide-spread importance for long term observations of time-dependant developments within individual groundwater areas or groundwater regions. The basic network consists of **baseline measuring points and trend measuring points.**

Baseline measuring points do

- describe natural groundwater quality conditions
- provide a basis for comparison with measuring points having anthropogenic impact.

Trend measuring points do

- determine the trend of groundwater quality during longer periods
- provide a basis for the identification of the impact of hazardous substances
- check the groundwater policy

There can be set up in addition regional or thematic measuring points to determine the stress on groundwater caused by

a. intensive farming
b. storage and handling of hazardous substances
c. pollution by atmospheric deposition

USER RELATED MONITORING

Networks for user-related monitoring are focussed to water for drinking purpose in upstream areas as well as in water processing plants.

a. **Monitoring sites in upstream areas** to supervise groundwater quality within the catchment area of drinking-water processing plant. They render prewarnings especially in cases of widespread contamination.

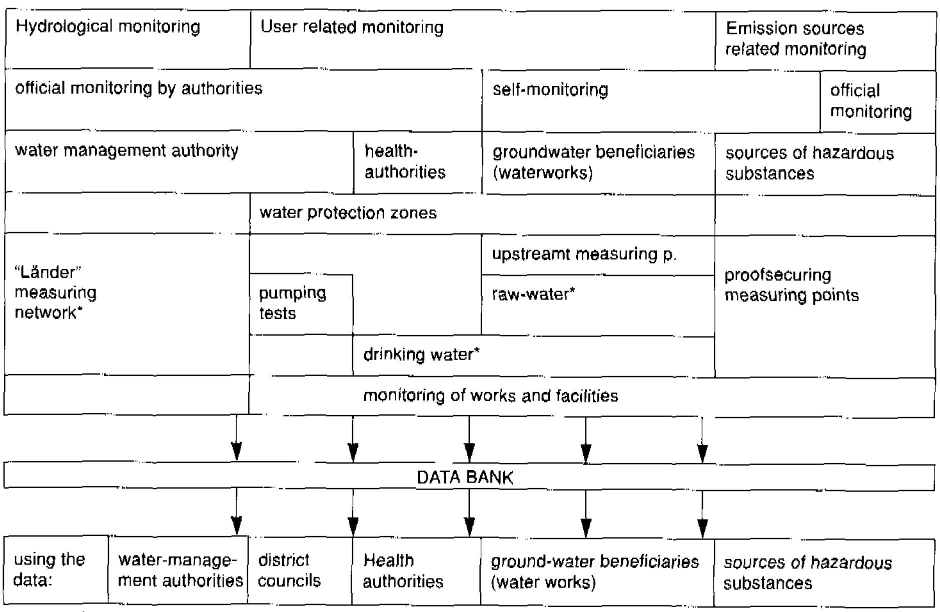
b. **Monitoring of Raw water** (for drinking water)
Reasons for raw water monitoring are:

- checking possible effects on groundwater quality caused by abstraction of groundwater
- identification of changes in raw water quality with regard to safeguard water supply and
- detection of raw water quality in order to run existing water treatment works at an optimum and to control their results.

For the execution of raw water monitoring the water works are responsible. In some of the Federal States extent and realization of raw water supervision is compulsory based on their specific water laws.

c. **Drinking-water monitoring**
Health authorities obtain measurement and analysis results to judge drinking-water quality according to the Federal Law of Epidemics or to the Standards for drinking water (TrinkwV) and Food Control. Drinking-water readings are also provided within the self-controlling of water supply works. Normally the readings in question are not complete water analysis as they might be necessary for the monitoring of groundwater quality.

d. **Self-monitoring in restricted parts of water protection zones**
Here water supply carry out owned monitoring of the water treatment process as well as in the water protection zone.



* individual measuring points may also be included in "Länder" measuring networks

Figure 1: Frame concept for groundwater monitoring. *Source after LAWA-Germany.*

Short programme

Parameter	units	detection limit
Coloration		
Cloudiness		
Odour		
Taste*		
Temperature	°C	
Conductivity at 20°C	µS/cm	
pH		
Titratable acid at pH 4.3	mmol/l	
Oxygen	mg/l	0.5
Calcium	mg/l	2.0
Magnesium	mg/l	1.0
Chloride	mg/l	5.0
Sulphate	mg/l	5.0
Nitrate	mg/l	1.0
Permanganate index	mg/l O_2	

* only if required

Table 1: *Groundwater measurement methods.*

Full programme

Parameter	units	detection limit
Coloration		
Cloudiness		
Odour		
Taste *		
Temperature	°C	
Conductivity at 20°C	µS/cm	
Redox potential	mV	
pH		
pH after saturation with CaCO3		
Titratable acid at pH 4.3	mmol/l	
Titratable acid at pH 8.2 *	mmol/l	
Titratable base at pH 8.2	mmol/l	
Oxygen	mg/l	0.5
Calcium	mg/l	2.0
Magnesium	mg/l	1.0
Sodium	mg/l	1.0
Potassium	mg/l	1.0
Manganese (total)	mg/l	0.01
Iron (total)	mg/l	0.01
Ammonium	mg/l	0.05
Fluoride	mg/l	
Chloride	mg/l	5.0
Sulphate	mg/l	5.0
Nitrate	mg/l	1.0
Nitrite	mg/l	
ortho-Phosphate	mg/l	0.01
Silicic acid	mg/l	
Permanganate index	mg/l O_2	
DOC	mg/l	0.1
spectral absorption at 436 nm	m^{-1}	
spectral absorption at 254 nm	m^{-1}	

* only if required

Table 2: Groundwater measurement methods.

EMISSION RELATED MONITORING

Monitoring network being related to emission sources serve as groundwater supervision regarding all influences that may cause harmful changes in groundwater quality. Emission sources monitoring stations are established to control groundwater quality within the sphere of influence of potential emission sources for hazardous substances. They are established above all when groundwater pollution cannot be entirely excluded for instance: landfills, industrial sites, plants and facilities for storing, handling and reloading, producing, treatment and utilization of hazardous substances, intensively cultivated agricultural sites, percolation of contaminated surface waters and waste water. Figure 1 shows the three different types of monitoring in Germany including the competent authorities.

PARAMETERS AND FREQUENCY

Information on monitoring network, organization of groundwater monitoring stations, maintenance and upkeep, sampling, list of parameters and analysis procedures is compiled in the LAWA-Guideline for observation and evaluation, groundwater quality, part 3 (1993).

The basic network in Germany consists of more than 2000 groundwater monitoring stations (Bavaria 280) which are run by the Federal States. When installing a station a basic monitoring programme is once done which comprises all parameters. Then according to the circumstances monitoring responds to

- a short measuring programme or/and
- a full measuring programme.

Under special conditions special measurement programmes may become necessary (e.g. for metals, organic parameters, volatile halogenated hydrocarbons, biological tests etc.).

The frequency is twice or four times a year; a full programme may be regularly followed by a short programme and vice versa.

The parameters are shown in table 1 (short programme) and table 2 (full programme).

Let me stress particularly the importance of taking samples and analysing.

In order to be able to compare the results of analysis by different laboratories participating in a measurement programme, it is necessary to use uniform analysis methods. The tolerances to be aimed at for each substance depend on the requirements for assessing the ground-water quality, the current state of analysis techniques, and the effort needed to achieve a particular tolerance.

Finally analytical quality control (AQC) is a prerequisite for meaningful results of analysis. The term AQC embraces all the measures which allow one to state the quality of results of analysis. These start with taking the sample and extend to the documentation and assessment of the values. In particular, they include the stages

- taking the sample
- measurements in the field
- on-the-spot pretreatment of samples
- transport of samples
- reception at the laboratory
- laboratory pretreatment

- measurements
- evaluation of the measurements
- documentation.

COSTS (GROUNDWATER QUALITY MONITORING)

The costs can be divided in costs for

- planning of monitoring stations
- construction of monitoring stations
- running the network including analytical network
- maintenance of the network.
- data processing

For Bavaria (70.000 km^2) with 280 basic monitoring stations the following costs may be calculated:

• planning	300.000 DM (whole Bavaria)
• construction	40.000 - 50.000 DM/station
• running including maintenance	6.000 - 7.000 DM/station and year

→ annual costs without data processing for Bavaria about 2.000.000,- DM/year (28.5 DM/km^2.year)

GROUNDWATER QUANTITY MONITORING NETWORKS

MEASUREMENT NETWORK

Requirements
The planning and setting-up of a network for groundwater monitoring (groundwater measurement network) are determined by the following points of view:

Purpose of observing the groundwater
A groundwater quantity monitoring network is intended to provide comprehensive knowledge about the state of the groundwater over as large an area as possible; the quantity data of interest concern the natural long-term and short-term changes in the groundwater level. In many cases networks are set up to determine what effects occur, or have already occurred, as a result of human intervention.

Factors influencing the water
The behavior of the groundwater depends on natural conditions and to an increasing extent on human intervention. These must therefore be taken into account when planning and setting up a groundwater quantity monitoring network. Individual factors are the structure of the subsoil, the

nature of the terrain, surface waters, climatic conditions, and the use to which groundwater is put.

Existing access points
An inventory should be made of existing points of access to the groundwater, such as observation pipes, wells, springs, and excavations. These access points must be checked for suitability as monitoring stations, before they are included in the network or new monitoring stations are created nearby.

ORGANIZATION

In general, three groups of quantity monitoring networks can be distinguished in Germany, depending on the task they are to fulfil:

a. A **basic network** for observation over the entire region or country (without a time limit) of groundwater levels, dependent mainly on climatic influences, in all important groundwater arteries. The density should be 1 station for every 100 sq.km.
b. **Supplementary networks** to increase the density of observation points over certain areas and certain periods of time or for sporadic observations of the groundwater level for particular hydrogeological questions (groundwater models or contour maps).
c. **Special networks** for observations over a limited time and area of groundwater level as part of public building works, for planning purposes and for collecting evidence. Such special networks can also be used for public research projects.

QUANTITY MONITORING STATIONS AND EQUIPMENT

The standard design in Germany is a 300-mm boring containing a 125-mm rigid PVC filtering pipe. The filter portion for the basic network must cover the entire depth of the aquifer and extend to 1 m above the highest groundwater level. There is no sump pipe. Quantity monitoring stations so constructed are also suitable for quality monitoring; multiple monitoring stations should each have their own boring.

As a rule, quantity monitoring stations in the basic network are equipped with continuous recording devices (at present mechanical pen recorders, in future electronic data recorders).

Individual manual measurements are made by rota at intervals of 1 week (normal case) to 6 months.

GENERAL DESCRIPTION OF GROUNDWATER QUANTITY MONITORING

The following description gives an idea of a German Groundwater Quantity Monitoring System (example: Free State of Bavaria with - 70.000 km^2).

1 **Network consists of:**

1.1 <u>State network</u>
Reference network
(600 stations in Bavaria; 1 per 100 sq.km) stations largely uninfluenced by supralocal conditions; assessment throughout the region; no time limit
Supplementary network
(2900 stations in Bavaria)
for defined water-management tasks; wide area, e.g. for mathematical modelling of groundwater

Special networks
(800 stations in Bavaria)
of local character for specific questions (e.g. effect of damming a river); observations limited in time and space

1.2 Private networks
(some 10 000 in Bavaria)
as a rule for monitoring water use and collecting evidence

2. **Quantities measured**
groundwater level and temperature

3. **Particular elements**

3.1 Setting up the network
(currently under revision in Bavaria) planning the network; choosing, planning, and constructing quantity monitoring stations

3.2 Collecting the data
(currently about 1250 pen recorders) measuring, analysing, logging, checking

3.3 Processing the data
(1 million items of data p.a. in Bavaria) storing, editing, archiving, analysing, publishing

3.4 Monitoring
(by measurement van in Bavaria) functional checks, unsilting, camera inspection

4. **Construction**
The standard design is a PVC observation tube, 125 mm in diameter to allow samples to be taken. It must extend to the lowest groundwater level.

5. **Equipment**
In the reference network the stations are equipped with automatic recorders; otherwise, depending on the importance, with recorder or optical plumbline.

6. **Organization**
Central management by the Bavarian Water Office; data collected by district water boards (Wasserwirtschafts- amt); data processed by them and by the central office (when of regional significance)

7. **Costs**
Bavaria: 3.000.000 DM/year without data processing

GROUNDWATER MONITORING ON THE NATIONAL AND THE EUROPEAN UNION LEVEL

It is understood that - as water use and related problems are local - monitoring activities and data bases should be developed on a national or regional level. It is certainly not the task of the EU to set up a groundwater monitoring network on a supra-national level.

Data bases are to be restricted on regional (Federal States of Germany) or national levels (smaller countries). The EU Commission actions for improvement and comparability of national or regional monitoring networks should be undertaken according to the subsidiarity principle, i.e. decisions are taken at the lowest possible policy level. It is up to the regional level to produce regular reports on the state of groundwater.

It seems therefore that:

- EU regions and member states have their own interest in developing basic monitoring networks of groundwater.
- National or regional monitoring networks should produce reliable and comparable data.
- Cooperation and exchange of information between basic networks should produce a coordinated groundwater monitoring.

Thus, objectives of monitoring in the EU levels are:

- To develop collaboration between member states in order to harmonize the necessary elements of groundwater monitoring.
- To help member states to ensure reliable and comparable results from national or regional levels, to evaluate and to publish the results in order to get a representative picture of the groundwater situation throughout the EU.

This may be achieved by common guidelines on installation and maintenance of measuring stations, equipment, staff, sampling, analysis procedures, parameters, frequencies, analytical quality control, data storage and data presentation.

CONCLUSIONS

The protection of our groundwater is a permanent task. To safe guard groundwater in its pristine quality we have to follow the principle "Precautionary action is better than repair".

To do this detailed knowledge on the groundwater is necessary which require a systematic and regular groundwater monitoring. In Germany an overall groundwater monitoring network has been installed since more than 10 years by which trends and contamination can be detected and remedial measures can be launched in time.

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AN OVERALL STRATEGY FOR GROUNDWATER QUALITY MONITORING BY WATER SUPPLY COMPANIES

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ABSTRACT

Dutch water supply companies are confronted with changed information needs regarding the quality of their groundwater resources. Because actual monitoring efforts only partly satisfy these needs, an overall monitoring strategy for water supply companies was developed, that will lead to more efficient monitoring. The strategy comprises the installation of tailor-made groundwater quality monitoring systems at extraction sites. The lay out of each system is mainly determined by the minimal travel time of the extracted groundwater. If information from different extraction sites is combined, this can offer more help to groundwater management authorities in fine-tuning their groundwater protection policy.

INTRODUCTION

About 65% of Dutch drinking water is produced from groundwater and the rest from surface water. Due to the greater unpredictability of surface water quality, water supply companies have put relatively less effort in monitoring the quality of their groundwater resources. Traditionally, most of the groundwater quality monitoring approaches by water supply companies were centred on legal monitoring necessities, imposed by health authorities. However, other monitoring purposes are emerging. The first reason is the growing awareness that the quality of extracted groundwater is less predictable than before. The effects that decades of human influences on groundwater quality eventually will have on the quality of extracted groundwater are still uncertain. It is hard to exclude therefore, that severely polluted groundwater will once show up in the extraction wells, without sufficient warning. The second reason is, that the authorities responsible for groundwater management have called upon the water supply companies to provide them with information on the condition of and threats to their resources (this is referred to as the "signalizing function" of the water supply companies).

This paper treats an overall strategy for groundwater quality monitoring by water supply companies (Baggelaar, 1992 and Baggelaar, 1994b). It highly increases monitoring efficiency, by tailoring monitoring efforts to the information needs and the special characteristics of each extraction site.

FROM INFORMATION NEEDS TO MONITORING LAY OUTS

The study underlying the strategy started off with an information analysis, to arrive at clear definitions and a priority setting of the information needs. It made clear that water supply companies have four distinct purposes for groundwater quality monitoring. In order of priority these are:

1. to fulfil the legal monitoring necessity;
2 to underpin operational decisions in safeguarding the provision of good drinking water, now and in the future;

3. to perform the aforementioned signalizing function;
4. to reassure customers.

Each of these purposes requires specific information on groundwater quality. In the following these specific needs are described and then the most appropriate monitoring lay out to provide this information is derived.

1. To fulfil legal necessity

By law, each water supply company must analyze its extracted groundwater on prescribed parameters with prescribed (minimal) frequencies. Every three months the extracted groundwater must be analyzed on electrical conductivity, temperature, pH, dissolved oxygen, KMnO4-use, NH4, NO2, HCO3, SO4, Cl, Na, K, Ca, Mg, Mn and Fe. And every four weeks the extracted groundwater must be analyzed on *Coli*-bacteria and thermotolerant *Coli*-bacteria. Comparable prescriptions exist for the drinking water. All gathered data must be reported yearly to the health authorities, but non-compliance situations must be reported immediately.

Required monitoring lay out

For this monitoring purpose, it suffices to sample the extracted groundwater on the prescribed parameters with the prescribed frequencies. There is no legal necessity to differentiate between extraction wells.

2. To underpin operational decisions

To enable a safeguarding of the provision of drinking water, a critical deterioration of the quality of extracted groundwater must be foreseen at least 10 to 15 years earlier (fig. 1). And to obtain enough impression on its duration and extent, the maximum prediction horizon should preferably amount to 30 years or more. The critical quality levels of concern here, depend on the capacity of the water treatment plant and on the quality standards for drinking water, as set by the health authorities. The minimal prediction horizon of 10 to 15 years is needed to study, design, develop, implement and test new water treatment plants, or - in case of very strong quality deteriorations - to reallocate the extraction.

Required monitoring lay out

To predict future quality, we must know: a) the quality of the groundwater that will reach the extraction wells within the desired maximum prediction horizon (preferably 30 years or more) and b) the hydrogeochemical processes that act along the flow paths. The most appropriate monitoring lay out will mainly depend on the ratio between the minimal travel time of the extracted groundwater and the maximum desired prediction horizon. If this ratio is less than one, shallow observation wells will provide relevant data for the prediction function. If however, this ratio

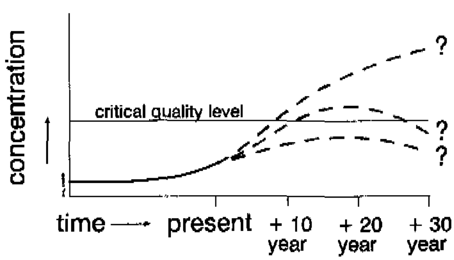

Figure 1: A water supply company needs to foresee the start, duration and extent of a critical deterioration of the quality of its extracted groundwater.

is greater than one, actual data on shallow groundwater quality will be less relevant. In this latter case, developments in the quality of the extracted groundwater and the deep groundwater, at least at 10-15 years travel time from the extraction wells and, if possible, also further away, will bear more relevance for predictions.

3. To perform the signalizing function

In the Dutch National Environmental Policy Plan, the signalizing function of water supply companies is formulated as follows (NMP, 1989):

"The water supply companies are called upon to:

- ... timely signalize which parameters hold risks for the quality of the drinking water;
- ... report yearly on the quality of the drinking water and the raw material that was used for its production".

This information may help the authorities responsible for groundwater to underpin their groundwater quality protection policy. Furthermore, parts of this information may be useful for the European Union, responsible for setting European quality standards and for environmental organisations, because they can support water supply companies in performing the signalizing function.

INTERMEZZO

Groundwater quality protection in the Netherlands

There are two levels of groundwater protection in The Netherlands: a) general protection by the national authorities and b) *specific* protection by the Provinces, focused on the areas where groundwater is extracted for public water supply. These are the socalled "groundwater protection areas", bounded by the lines denoting 25-years travel time of infiltrated water to the extraction site. This means that water infiltrating in this area, will reach the extraction wells within 25 years. Protective restrictions in these areas concern the use of certain pesticides ("the black list") and the application of fertilizers. If the restrictions result in suffered losses - for example by farmers - the Province provides financial restitutions, which are funded by special taxes on the use of groundwater, payed by the water supply companies.

The water supply companies will benefit from optimal protection measures, because a measure that is too weak, will lead to unacceptable pollution of the groundwater, while a measure that is too strong might lead to unnecessary economical damage. However, the authorities have insufficient facilities to evaluate their protection measures, because both the primary groundwater quality monitoring network, operated by national authorities and the secondary groundwater quality monitoring networks, operated by the Provinces, have very limited numbers of observation wells in groundwater protection areas. Furthermore, the screens of these wells are placed too deep (> 5 m below surface) to enable a fast evaluation of measures. Given their direct interest in protection measures, the water supply companies are seeking solutions to overcome this information gap, in close cooperation with the Provinces concerned. These efforts constitute a voluntary addition to the afore mentioned basic tasks of the signalizing function.

Required monitoring lay out

To fulfil the basic tasks formulated by the Environmental Policy Plan, no additional monitoring efforts are required. This is because the information on parameters that hold risks for the quality of the drinking water, will be provided by the monitoring lay out to underpin operational decisions

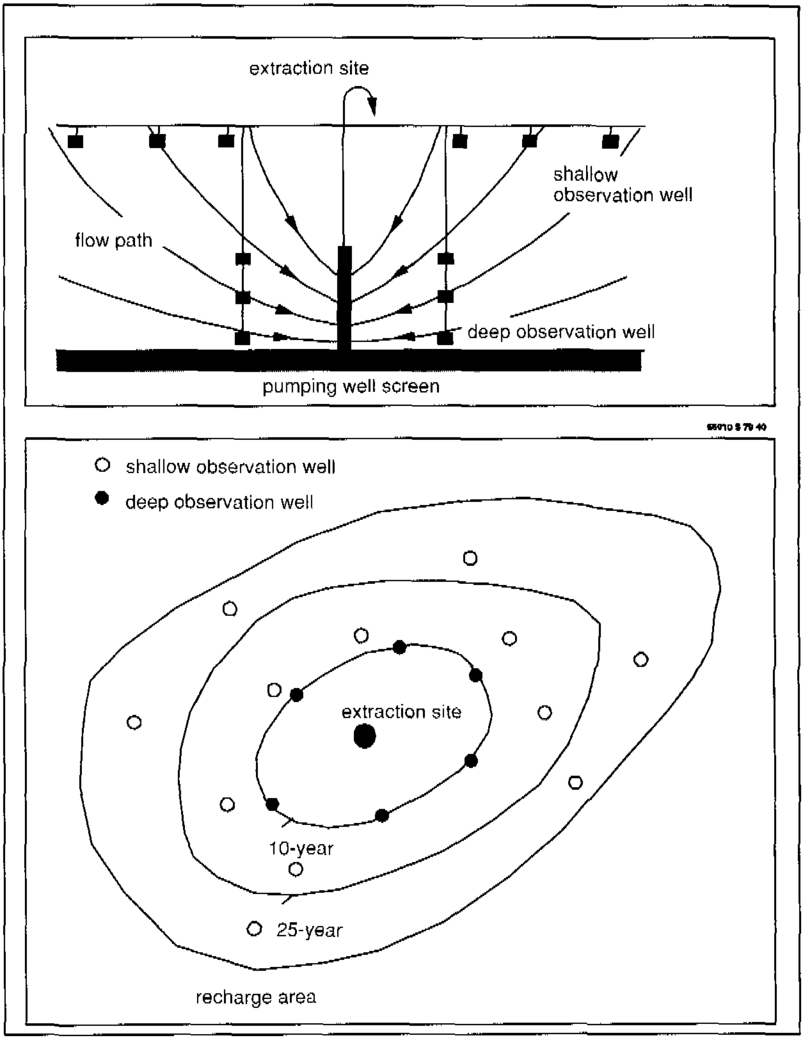

Figure 2: Profile (above) and aerial view (below) of monitoring lay out *III*, *which is recommended in a situation where* the water infiltrated at the surface can reach the extraction wells within a short period. The relevant information is obtained from monitoring extracted, deep and shallow groundwater.

(see number 2 above), while the data on the quality of the drinking water and the raw material *that was used for its production, will be provided by the monitoring lay out to fulfil legal necessi-*ties (see number 1 above). However, an additional effort will be required to report this information in an effective form to the authorities, preferably by combining information from different extraction sites (see further).

To evaluate protection measures two approaches can be followed. Firstly, a *theoretical evaluation* is possible, based on various assumptions regarding hydrogeochemical and hydrological processes in the underground. However, "harder" proofs require an empirical approach. This comes down to an extension of the monitoring lay out to underpin operational decisions, because it also requires data on shallow groundwater quality in a so-called "control area", outside the protection area, to correct for natural variations in groundwater quality (see further).

4. To reassure customers

To reassure customers, water supply companies should regularly make public to be capable of safeguarding the provision of good drinking water, now and in the future. It requires, amongst others, enough working knowledge on status and developments of the quality of the extracted groundwater.

Required monitoring lay out
This monitoring purpose requires no additional monitoring effort, because the information on status and developments of the quality of the extracted groundwater will be sufficiently provided by the earlier mentioned monitoring lay outs.

CONCLUSIONS ON MOST APPROPRIATE MONITORING LAY OUT

Based on the foregoing, our study concluded that the most appropriate lay out to fulfil the monitoring purposes, mainly depends on the minimal travel time of the extracted groundwater, in the following manner:

I minimal travel time is large[1]: monitoring of the extracted groundwater quality;
II minimal travel time is intermediate[2]: lay out *I*, extended with monitoring of the deep groundwater at least at 10 to 15 years travel time from the extraction wells and, if possible, also further away;
III minimal travel time is small[3]: lay out *II*, extended with monitoring of the shallow groundwater quality (see fig. 2).

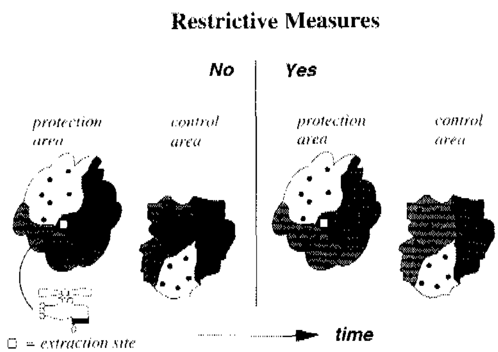

Figure 3: Monitoring strategy for the evaluation of protection measures in the protection area, using a "control area" to correct for natural changes in groundwater quality. Different shades indicate different combinations of landuse and soil type.

Monitoring to evaluate protection measures
If the extraction site is surrounded by a groundwater protection area (minimal travel time less than 25 years), it is appropriate to evaluate groundwater quality protection measures. This requires data on the shallow groundwater quality, both before and after the protection measures were taken. In situations where the groundwater table is low, it is recommended to sample the soil water in the unsaturated zone, to enable a sufficiently fast evaluation. The evaluation uses a statistical comparison of mean concentrations of both periods, with a correction however, for natural changes as measured in a so called "control area" (see fig. 3). The latter is completely

[1] large enough to render the future quality of the extracted groundwater practically unsuspected (a workable criterium is more than 100 years travel time);
[2] more than the desired maximum prediction horizon, but less than the limit mentioned under 1);
[3] less than the desired maximum prediction horizon.

comparable to the protection area in hydrological characteristics and relative distributions of land uses and soil types, but located outside it and therefore unaffected by any protection measures. *In most cases stratified sampling according to landuse and/or soil type will increase the efficiency of this evaluation* (different strata are shown with different shadings in fig. 3). The statistical data analysis for this evaluation is discussed further (see Phase 4, ad e).

Monitoring local pollutions

If potential pollution sources of more local extent are present in areas with small travel times to the extraction wells, local densifications of the network are recommended, to monitor the areal extent and possible migration of pollution plumes (fig. 4). Such information can be provided by comparing data from reference wells upstream (A in fig. 4) and monitoring wells downstream (B *in fig. 4) of the potential source.*

And if salinization by upconing of saline groundwater poses a threat, it is recommended to install deep "salt watcher" wells in strategic positions, enabling continuous registrations of electrical conductivity.

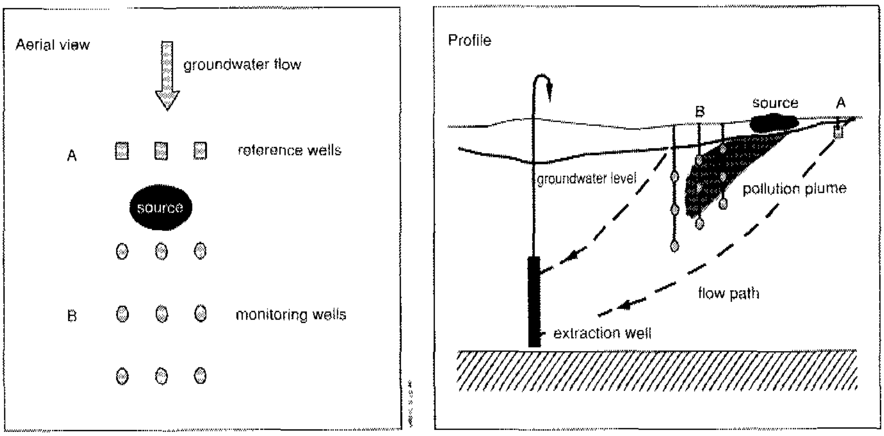

Figure 4: Aerial view (left) and profile (right) of the recommended lay out to monitor point pollution, using reference wells upstream (A) and monitoring wells downstream (B) of the potential source.

GUIDELINES ON HOW TO BUILD MONITORING SYSTEMS

The overall strategy that we developed was laid down in easy applicable guidelines for the water supply companies. It uses the concept of groundwater quality monitoring systems (see the intermezzo) at extraction sites, based upon the general concept of putting monitoring into a systems framework, as developed by researchers from the Colorado State University (Sanders et al., 1983, Ward and Loftis, 1986, Ward et al., 1990).

Following Ward et al. (1990), the building of a groundwater quality monitoring system comprises the following six phases: 1) information analysis; 2) preliminary study; 3) design and installation of the network; 4) set up of the procedures; 5) exploitation of the system; 6) optimization of the system.

The results of the first phase already being available, we developed guidelines for water supply companies on how to execute the other phases (Baggelaar, 1992). The following presents some of the main issues.

INTERMEZZO

What is a groundwater quality monitoring system?

A groundwater quality monitoring system is the combination of a monitoring network and a set of procedures. The monitoring network forms the physical part of the system, that enables groundwater sampling in strategic locations. The other - non physical - part of the monitoring system consists of procedures for the various driving forces of the information flow. In logical order these are sampling, sample analysis, data storage and control, data analysis and finally, reporting the desired information to the target groups.
The monitoring system will regularly provide the desired information on groundwater quality to the target group(s) ("value for money"), instead of just filling a data base with large amounts of data. Its procedures will assure consistency, thus avoiding non-natural variations in the data, that can obscure the information. And last, but not least, the formalization of the system into official procedures for the information flow, coupled with its clear efficiency (which will soon become clear in practice), will enhance the status of the monitoring system, rendering better protection in cases of general budgettary cutdowns.

PHASE 2: PRELIMINARY STUDY

The preliminary study phase comprises the collection and interpretation of all data that are needed to underpin the detailed network design. In this study sufficient working knowledge should be gathered on the location of the extraction area, its hydrogeological build up, the groundwater flow systems, the spatial and temporal provenance of the extracted groundwater, the spatial distribution of groundwater quality and the pollution sources and hydrogeochemical processes that can affect the quality of groundwater travelling to the extraction wells.
The relevancy of most of these information elements for the detailed network design, is highest if minimal travel time of the extracted groundwater is small. This travel time is the main characteristic determining the appropriate lay out of the monitoring network. It follows directly from the *response curve*, a plot of the cumulative percentage of extracted groundwater against travel time (fig. 5). The starting point of the response curve indicates the minimal travel time of the extracted groundwater, while the steepness of the curve indicates the variation in travel times (the latter determines the duration and extent of pollution effects on the quality of the extracted groundwater). In general, situations where groundwater is extracted from a freatic aquifer are

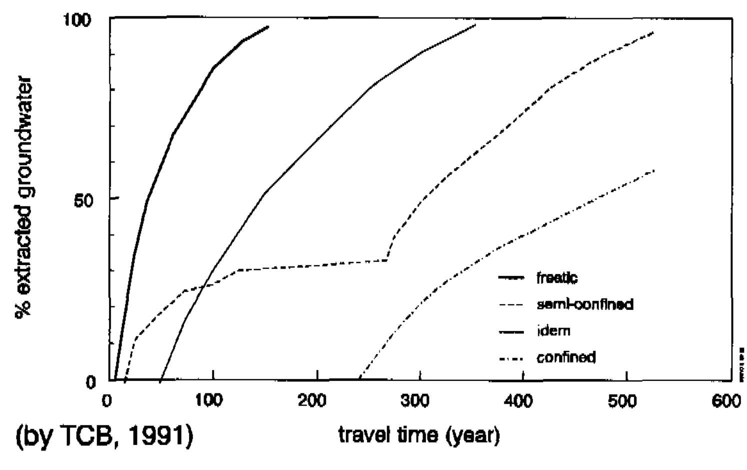

Figure 5: Some examples of response curves (plots of the cumulative percentage of extracted groundwater against travel time), for different hydrogeological situations.

characterized by small travel times (left curve in fig. 5). If the minimal travel time is less than the desired maximum prediction horizon, data on present and historical land use are relevant to predict the quality of the extracted groundwater. This also applies to data on present and historical local pollution sources. And in situations where groundwater is extracted from a confined aquifer, minimal travel time will generally be large (right curve in fig. 5).

PHASE 3: NETWORK DESIGN AND INSTALLATION

Within the overall lay out of the monitoring network, determined by the minimal travel time of the extracted groundwater, some freedom of choice remains for the exact locations of the observation wells. If shallow groundwater quality is of concern, it is recommended to apply stratified sampling according to land use and/or soil type, because these are the main factors determining this quality. If the groundwater table permits, it is recommended to sample auger holes, because this offers more flexibility to adapt the network to changes in land use. In case of a deep groundwater table, the soil water quality can be monitored instead. Furthermore, under certain boundary conditions, the mixing of groundwater samples can result in considerable savings on analytical costs.
If more understanding of hydrogeochemical processes is required, some deep observation wells should be placed along the same flow path. The drilling of observation wells also offers a good opportunity to obtain more information on the hydrogeological and hydrogeochemical build up of the area. The assumptions regarding these characteristics should be checked and, if necessary, adapted, using the incoming information.

Parameters of interest Factors to consider in the selection of monitoring parameters are type and importance of the identified pollution sources, hydrogeochemical behaviour of the parameters, environmental and health issues and finally, possibilities and costs of analysis. In general, the costs of analysis will form the bulk of the exploitation costs of a monitoring system. It will be highly profitable therefore, to tailor the package of parameters to the specific situation and not to take recourse to some standard package.
To obtain a good starting point and to enhance hydrogeochemical understanding of the area, monitoring can start with an "exploration package", containing a large variety of parameters, such as temperature, pH, electrical conductivity, all the major cations and anions and various trace elements. It will suffice to use this exploration package at low frequency (5 to 10 years). If tritium is included in the first round, the resulting estimates of the groundwater age in the various subareas, can serve to improve the understanding of the hydrogeological situation.
For the normal monitoring routine, a "testing package" is recommended, centred on parameters that can impose a real threat. Some of them may already have been identified in the preliminary study, and others from the results of the exploration round. Relevant pesticides may be identified from an inquiry amongst farmers and local distributors of pesticides and from general knowledge on the relation between land use and the application of pesticides. The tailoring of the testing package requires the help of an expert in hydrogeochemistry, but this investment will certainly pay off! To give an example: if acidification poses a threat, a very effective package of parameters can be composed, based on pH, redox condition, sampling depth, land use and hydrogeological situation (Stuyfzand, 1991).

It can also be efficient to apply a conditional two-stage approach in a sampling round. For example: all wells with a pH of less than 5.5 in the first stage of sampling, should be sampled on a variety of trace elements in the second stage. Two-stage approaches do not always work for the extracted groundwater, because it might be composed of various water types. In most cases it will be more efficient to select different packages of parameters for different types of subareas or even for different depths, depending on the hydrogeochemical situation. It must be realised however, that such complex approaches may create great logistic problems for the sampling unit and the laboratory.

PHASE 4: PROCEDURES SET UP

The information flow in a monitoring system is assured by procedures, that prescribe in detail its driving forces. In chronological order these are sampling, sample analysis, data storage and control, data analysis and reporting. Only some highlights are mentioned below.

Sampling procedure Experience has shown, that small deviations in sampling approach may add large non-natural variations to the data, thus obscuring the information they bear (see for example Stuyfzand, 1987). To prevent this, the sampling procedure must prescribe in great detail the various sampling actions, such as routing, preparation of equipment and sample bottles, sampling activities, sample conservation and transport. For the same reason, the procedure must define the pumping regime to sample the extracted groundwater.

Sample analysis procedure If a change in method is unavoidable, a number of samples should be analyzed with the old method and with the new method, to allow sound statistical testing on a systematic change and on a change in precision. If such changes do occur, we can correct for them in applying trend analysis. To enhance quality assurance, the procedure must also prescribe the frequency and methods of laboratory quality control. Such methods should comprise regular calibration, using standard solutions and preferably also regular participation in inter-laboratory trials.

Data storage and control procedure To avoid polluting the data base with erroneous data, all data should be checked upon storage. Depending on the package of parameters involved, checks are possible on ionic balance, on electrical conductivity, on the residual after evaporation and on outliers (Baggelaar, 1992). We define outliers as values that strongly differ from the set of data that they are supposed to belong to, such as previous data from the same observation well and/or data from wells in the same subarea. Various statistical outlier tests are available, mostly assuming a normal distribution of the data, examples being Dixon's test (Dixon, 1953) or Grubbs test (Grubbs, 1969). In case of groundwater quality data however, distributions tend to be skewed to the right, making an outlier test without a normality assumption more appropriate. An example of such a distribution-free outlier test is Veglia's test (Veglia, 1981). Once an outlier is detected, all the actions that have led to its existence should be reviewed and only if irregularities are found, there is ground to remove the outlier. The indiscriminate removal of "strange data", without any check on their backgrounds, should be avoided, because it might harm the effectiveness of the monitoring effort.

Data analysis procedure The procedure for data analysis must describe in detail how to cope with censored values (reported as "less than detection limit") and missing values, which maps, graphs and tables to produce and which hypotheses to test statistically. Depending on the lay out of the monitoring system, one or more of the following analysis methods will be required: a) testing compliance of the extracted groundwater; b) comparing results from reference and monitoring wells; c) trend analysis; d) predicting future quality of extracted groundwater and e) evaluating protection measures. Some methods are briefly discussed below.

ad c) trend analysis Water quality data often come from non-normal distributions and may suffer from seasonal effects and/or serial dependence, making them less suited for trend analysis with classical statistical methods, such as the linear regression method. In the last fifteen years various new methods for trend analysis were made available, that can cope with the special characteristics of water quality data (see, for example, (Gilbert, 1987), or (Helsel and Hirsch, 1992)). However, the selection of the most appropriate method for a particular data record requires a good deal of statistical expertise. Therefore, we developed a protocol for trend analysis and implemented it in software (Baggelaar and Baggelaar, 1989, 1991 and 1994a).

ad d) predicting future quality of extracted groundwater To assist the water supply companies in predicting the future quality of the extracted groundwater, various predictors were made available, each valid for a specific monitoring lay out (Baggelaar, 1992). They vary from a simple extrapolator of the quality of the extracted groundwater (purely statistical, or also allowing some intuitive process knowledge) to a 3-D hydrogeochemical transport model, using data on the quality of shallow groundwater, deep groundwater and extracted groundwater.

ad e) evaluating protection measures Applying the monitoring lay out shown in fig. 3, a measure may be considered effective, if the following corrected mean difference ($\bar{u}$) is stistically signifi-cant different from zero (one sided testing):

$$\bar{u} = (\bar{x}_{p1}-\bar{x}_{p2}) - (\bar{x}_{c1}-\bar{x}_{c2})$$

where $\bar{x}_{pi}$ and $\bar{x}_{c1}$ are the mean concentration of the relevant chemical parameter in the shallow groundwater of the protection area (subscript p) and in the shallow groundwater of the control area (subscript c), *i* being the time index (*i*=1 before and *i*=2 after the protection measures were taken).

PHASE 5: MONITORING SYSTEM EXPLOITATION

Once the procedures are ready, the exploitation of the system can begin, starting the information flow with the first sampling round. The amount of energy that has been put into the previous building phases of the monitoring system, will begin to pay off now, because all collected data will easily find their way on the smooth information highway, leading to regular and clear reporting. As information needs may evolve, it is recommended to evaluate from time to time whether the information generated still fits the needs, for example by including enquiry forms in the reports. If there are signs of changes in the information needs, a new thorough information analysis is highly justifiable. And if necessary, the monitoring system should be adjusted.

PHASE 6: MONITORING SYSTEM OPTIMIZATION

After a certain period of exploitation, more will be known about the statistical characteristics of the groundwater quality in the studied area, enabling a (first) optimization of the monitoring system. An optimal monitoring system generates the desired information against minimal costs (see, for example, (Schilperoort and Groot, 1983), or (Baggelaar, 1995)). Optimization of the spatial density of the shallow monitoring network can be accomplished first, because it only requires spatial statistical characteristics of the relevant parameters. Optimization of temporal densities however, requires temporal statistical characteristics of the relevant parameters, which may take at least 5 to 10 years time to gather.
After at least 10 to 15 years, the "early-warning system" can be evaluated and - if necessary - fine-tuned, using life data. This system comprises a model to predict future quality of the extracted groundwater, using as input data on groundwater quality from the shell of deep observation wells, situated at least at 10 to 15 years travel time from the extraction site.

COMBINING INFORMATION FROM EXTRACTION SITES

If water supply companies perform the signalizing function independent of each other, the authorities responsible for groundwater management will be confronted with various ad hoc signals. It is felt however, that the signalizing function can be greatly enhanced, if information from extraction sites is combined and interpreted in a suitable manner and then reported to the authorities. The combination will enable to differentiate between homogeneous groups of extraction sites, leading to more statistical power, better descriptions and understandings of quality developments of extracted groundwater, stronger evaluations of protection measures and statements with more general value and impact.
A working example of the added value of combined information is given by the RIWA (the Asso-

ciation of Rhine and Meuse Water Supply Companies). Its regular reports on the status and developments of the Rhine and Meuse river water qualities, combine information from many individual monitoring stations. This information has helped the authorities to play a more active role in the protection of surface water quality, now gradually leading to an improvement of this quality.

At present, only data on the quality of the extracted groundwater, provided by the legal monitoring purpose, are stored in a central database (by Kiwa). Notwithstanding the fact that these data probably also reflect many non-natural variations, due to lack of or changes in relevant procedures, their central storage has enabled various overall studies, leading to important reports on the developments of the quality of our drinking water resources, showing the authorities the locations and extent of different environmental threats. It proved very helpful to differentiate between various groups of extraction site, according to vulnerability and provenance of the extracted groundwater, and report results for each group separately (see, for example, Van Beek et al., 1990).

Developments towards combining information
Many water supply companies are implementing the monitoring strategy described in this paper, setting up monitoring systems. This means that within a few years more elaborate data on spatial and temporal aspects of different environmental threats to the groundwater resources of our drinking water will come at hand, enabling a much more effective execution of the signalizing function. On behalf of, and in cooperation with the water supply companies, Kiwa is studying on possibilities to centralize all relevant data and to combine them to information with more impact. First concern of the study is to arrive at consensus among water supply companies on using and combining their data. Then the most appropriate grouping of extraction sites and presentation methods must be established, in close cooperation with the target groups of the combined information (provincial, national and European authorities and perhaps also environmental organisations). Finally, reporting structures will be set up, with clear defined procedures, tasks and responsibilities.

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INDICATOR DEVELOPMENT ON THE BASIS OF RIVER ECOSYSTEM CONCEPTS

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ABSTRACT

The present study aims to develop river ecosystem indicators. Ecosystem based indicators will be derived on the basis of theoretical concepts describing natural rivers. Recommendations will be made for the monitoring of rivers, in particular the Rhine.
River ecosystem characteristics may be derived from river ecology concepts on zonation, stream hydraulics, river continuum, nutrient spiralling, serial discontinuity, flood pulse, riverine productivity and catchment hierarchy. Predominant among these are the abiotic steering variables describing the hydrology, geomorphology and water quality; they act as a template for ecosystem functioning. Functional processes are dominated by the flow of matter which is affected by input, processing and retention of organic matter and nutrients. Spatial and temporal variation of input and retention of matter, and the flow along the length of the river cause shifts in species distribution. These are reflected in gradients of macroinvertebrates and zonation of fish and benthic fauna which form a dominant structural characteristic of the river ecosystem. Retention of matter is proposed as indicator for the functional characteristics and the spatial distribution of species as indicator for structural characteristics of the river ecosystem.
Further, the review of river concepts shows the development in time of a more spatially integrated and interdisciplinary view of rivers. Accordingly the design, implementation and ecological assessment of monitoring programmes which measure an integrated set of water quality, hydrological, geomorphological and ecological variables should reflect such an integrated spatial view. On the basis of these results recommendations are made for further improvements to the present Rhine monitoring programme.

INTRODUCTION

Many Western European rivers and their catchments have changed drastically in water quality, geomorphology and hydrology due to increased population densities, industrialization and cultivation. This has brought pressure on the ecological and drinking water supply function of rivers. Policy makers and water management bodies are being increasingly challenged to cope with conflicting interests. This demands integrated assessment methodologies for making trade-offs between human activities and environmental quality. Indicators are useful tools in these methodologies. An indicator is defined as a piece of information that is part of a management process and has been assigned a significance beyond its face value (Bakkes et al., 1994). In this study, a river ecosystem indicator is defined as an observable and measurable quantity providing information on an aggregated level on the river ecosystem condition for river basin management. River ecosystem indicators that are currently used in monitoring programmes of rivers (e.g. River Rhine) are mainly related to water quality assessment or the presence and abundance of certain indicator species (e.g. Salmo salar). So far, no functional ecosystem characteristics have been used in monitoring programmes. Although it is widely recognized that the selection of ecological indicators should be based on the ecosystem itself, the input of scientific knowledge to indicator development is, to date, limited. The aim of this study is to derive river ecosystem indicators. During the past decade several river ecosystem concepts have been developed describing the processes and structure of natural, undisturbed rivers. These concepts will be used to derive river ecosystem indicators.

REVIEW OF RIVER CONCEPTS

The first attempt to describe the ecosystem of the whole river was the zonation concept. The zonation concept divides a river into zones characterized by fish communities (Huet, 1954) or macroinvertebrate communities (Illies & Botosaneanu, 1963). The zonation reflects differences in water temperature and flow velocity.

The development of the River Continuum Concept (RCC)(Vannote et al., 1980) was an important step in river ecology, as it was the first attempt to describe both structural and functional characteristics of stream communities along the entire length of a river. The RCC argues that the biotic stream community adapts its structural and functional characteristics along the continuous gradient of the abiotic environment from headwaters to river mouth. This is expressed by the distribution of organic matter and macroinvertebrate functional feeding groups. In general, rivers can be divided in three parts based on stream size; headwaters, medium-sized streams and large rivers. The headwaters of rivers are strongly influenced by riparian vegetation. Primary production is low because of shading and the vegetation contributes large amounts of allochtonous detritus. The size of this organic matter is large. The influence of the riparian zone diminishes moving downstream; the importance of terrestrial organic input decreases, whereas primary production and transport of organic matter from upstream increase. The size of organic matter decreases. Large rivers receive organic matter mainly from upstream, and it has already been processed to a small size. Primary production is often limited by depth and turbidity.
Changes in the size of organic matter along the length of the river are reflected in the distribution of functional feeding groups of invertebrates. In the headwaters shredders are co-dominant with collectors. Shredders process larger organic matter. Collectors eat smaller parts of organic matter by filtering them out of the water or gathering from the sediments. Collectors and grazers (or scrapers), which shear attached algae from surfaces, dominate the middle part of the river. In the lower reaches invertebrates consist mainly of collectors. Fish populations show a shift from cool water species low in diversity to more diverse warm water communities.

The RCC constrasts strongly to the zonation concept by emphasizing gradients. An intermediate is the theory of stream hydraulics (Statzner & Higler, 1986). This theory distinguishes a zonation pattern of benthic fauna in which the distinct changes in species assemblages are often linked to transitions in stream hydraulic parameters, such as current velocity, depth, substrate roughness and surface slope.

The Resource Spiralling Concept extends on the RCC by elaborating further on the processing of organic matter along the length of the river. The downstream flow of rivers adds a spatial dimension to resource cycles in stream ecosystems by downstream displacement of material. This results in partially open cycles or "spiralling" (Wallace et al., 1977, Newbold et al., 1981). Spiralling is a function of both downstream transport rate and retention processes (Minshall et al., 1983). A high transport rate, determined largely by water flow, will increase the spiralling length, whereas retention mechanisms, such as physical storage and biological uptake and storage will decrease the spiralling length. In general the spiralling length increases with stream size.

Another concept associated with the RCC is the serial discontinuity concept (Ward & Stanford, 1983). This addresses the effects of dams on rivers. Dams disrupt the continuum and cause upstream-downstream shifts in abiotic and biotic parameters and processes. The effect is related to the position of the dam along the continuum. In general, dams increase the homoge-

homogeneity of a variable between two discontinuities (Ward & Stanford, 1995).

The Flood Pulse Concept (Junk et al., 1989) describes the effects of floods on both the river channel and its floodplain in an unmodified, large river-floodplain system. Floodplains tend to have their own nutrient cycles. Release and storage of nutrients in the floodplain depend on the flood cycle, vegetation cover, and in temperate regions, to the growth cycle of the vegetation. The carbon exchange between floodplain and main channel depends on two factors: the presence of retention mechanisms keeping carbon in the floodplain and reducing leakage to the river channel; and the duration and flushing rate of the flood. In general, biological productivity in a large river-floodplain system is high. Life cycles of biota using floodplain habitats are related to the flood pulse characteristics. The floodplain is used for food supply, spawning and shelter. The main channel is used as a migration route, for spawning and as a refuge during for example droughts or hibernation. River-floodplain systems show a high diversity of habitats. Differences in the duration of flooding, in soil structure and in vegetation result in many different small-scale habitats and physico-chemical conditions. As a consequence, species diversity in river-floodplain systems is also high.

Thorp & Delong (1994) introduced the riverine productivity model. This concept states that carbon in constrained large rivers does not originate solely from downstream transport (as stressed by the RCC) but also from local primary production and inputs from the riparian zone. The community composition of macroinvertebrate functional feeding groups will differ among sites within a large river in response to both the physical characteristics of each habitat and the types of organic matter present. In general, high invertebrate densities are found in riparian zones due to their large habitat diversity and their role in retaining structures of organic matter.

Finally, Townsend (in press) argues in his catchment hierarchy approach, that the entire catchment should be the unit of study for river ecologists. This author states that most river concepts which place emphasis on longitudinal, lateral and vertical dimensions can be integrated within the framework of the patch dynamics concept. Patch dynamics focuses on interactions and changes in patches in the ecosystem. This hierarchical framework of both river and patch dynamics concepts on the scale of the river catchment enables the prediction of spatial and temporal patterns of ecological variables in the river basin. For example, the dominant source of organic matter, such as transport from upstream, lateral input or instream production is predicted in different parts of the riverbasin. The temporal dimension is important in a dynamic environment such as a river in which disturbance affects ecosystem structure and functioning (e.g. variable discharges, flood pulse).

FROM RIVER CONCEPTS TO INDICATORS

Summarizing these concepts, it is apparent that the steering factors for river ecosystem functioning are the abiotic variables, which comprise the hydrological, geomorphological and water quality characteristics of a river. These abiotic steering variables determine the functioning of the biotic ecosystem, which is described by functional and structural characteristics.

The functional characteristics of a river ecosystem focus on resource cycling. Resource cycling in rivers is dominated by the flow of water and matter from the source to the mouth. Matter occurs in the form of inorganic nutrients and minerals, particulate matter, and organisms. Processes influencing the flow of matter are input, processing and retention of matter. Matter enters the river from the vegetation in the riparian zone (allochtonous organic matter), instream primary production (autochtonous organic matter) or via exchange of nutrients, minerals, organic matter and organisms between the river and its floodplain during the floodpulse. There are two types of retention mechanisms: firstly, physical retention structures and processes, such as dams, vegetation in riparian zones and floodplains or sedimentation; and secondly, biological retention which occurs by uptake of nutrients and organic matter in the foodweb. The total of all these factors determines the output of matter at a river or river part.

The structural characteristics relate to species distribution, diversity and abundance of the river ecosystem. Predominant here are typical spatial distributions of species reflected by gradients in macroinvertebrate functional feeding groups or in the zonation of fishes or benthic organisms. These distributions are caused by a spatial and temporal variation in dominant sources of inputs, hydrological and geomorphological characteristics relating to increasing stream order and physical and biological retention mechanisms in the river catchment. Temporal variation is caused by the daily, seasonal and annual cycles, affecting the input of organic matter, hydrology of the river, primary production etc. The spatial distribution of sources of input and retention mechanisms is predicted by the river concepts for different dimensions (longitudinal, lateral, vertical) and different parts (upstream, midstream, downstream) in the river catchment (fig. 1).

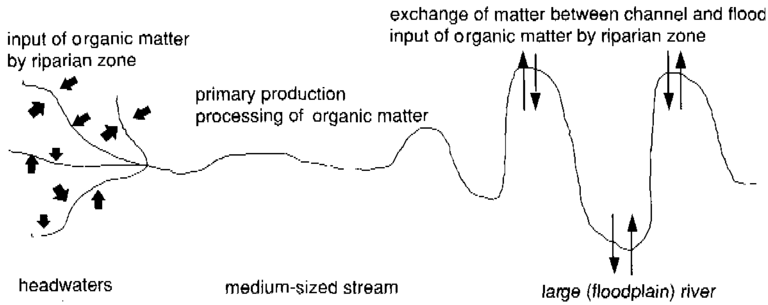

Figure 1: Spatial distribution of input, flow and retention of matter in the river catchment

The river concepts describe a natural, undisturbed river system. The effects of human activities need to be included. A cause-effect chain is distinguished whereby human disturbance changes the abiotic steering variables, which in turn affects the biotic structural and functional characteristics of the river ecosystem. This cause-effect chain is represented in figure 2.

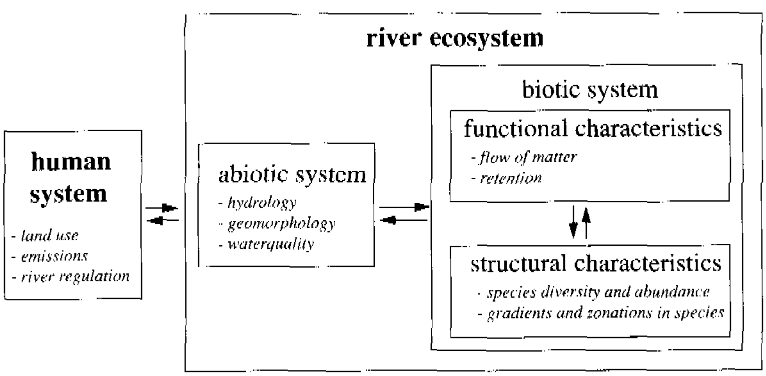

Figure 2: Cause - effect chain of the human system influencing the river ecosystem

PROPOSAL FOR RIVER ECOSYSTEM INDICATORS

From fig. 2 it can be seen that knowledge about the abiotic environment is crucial, firstly because it determines the preconditions for biotic ecosystem functioning, and secondly, because human impact generally affects the abiotic variables of the river first. Information on the abiotic environment encompasses hydrology and geomorphology of the river and riparian zone and water quality. Furthermore, the natural input of organic matter from the riparian zone or floodplains is an important structuring factor in a river ecosystem. The riparian input of organic matter can be assessed by using a combination of remote sensing techniques and a selection of field measurements (Cummins et al., 1989). The composition of the plankton community could be indicative for the exchange of matter between river and floodplain (Walker et al., 1995).

Functional ecosystem characteristics describe productivity, and the assimilating capacity for matter imported into the river ecosystem. An important determinant is the carrying capacity of a ecosystem. A drastic decrease in production or assimilation of nutrients indicates that human disturbance is affecting the functional processes of resource cycling.

Potentially suitable indicators for functional ecosystem characteristics are the flow of matter and its retention. The flow of matter is commonly determined by measuring water discharge and concentrations of sediment, nutrients, organic matter and algae. In many monitoring programmes the variables water discharge and concentrations of matter are measured, but calculations of loads are seldomly carried out. The concepts imply that data on loads of matter should be considered as flows and should also be linked to retention. Retention is defined as the difference between input and output of matter in a river or river reach. It is a measure of the physical barriers and biological retention processes of matter in a river. Retention can be calculated by substracting the load of nutrients (total C, N and P) between sites along the length axis. At present, relatively little information is available to assess data on retention (Grimvall & Stalnacke, 1995). Possible approaches are the comparision of values of both disturbed and relatively undisturbed rivers, taking into consideration differences in climate, geology and land-use of the watersheds or a historical evaluation of the development of retention values in time for a particular river. Such comparisions will help to obtain information on the impact of human activities on retention and its significance for ecosystem functioning. The integration of data on flow and loads in models allows, in principle, a more detailed and extensive analysis of the flow and retention of matter.

Species diversity and abundance represent the structural ecosystem characteristics. They are related to functional ecosystem processes by their biological retention of matter. Biodiversity of a river also has a social significance, especially with respect to species of higher trophic levels (e.g. fish, mammals, birds). The return of a popular species is often stated as an objective of ecological restoration measures to increase the social acceptance for these measures (e.g. `Salmon 2000' in the Rhine, beaver in the Elbe). Information is needed about species diversity, species extinction and species dominance. The spatial distribution of species is proposed as a structural ecosystem indicator as it represents an aggregation of underlying functional processes. An indicator for gradients is the ratio of macroinvertebrate functional feeding groups along the longitudinal axis.

With regard to human disturbance, information about land-use in the catchment, emissions from human activities, changes in water quality and river structure is required to assess or model ecological impacts and to understand spatial distribution and temporal trends. Remote sensing is a suitable instrument for the collection of information at this spatial scale.

THE NEED FOR A MULTIDISCIPLINARY APPROACH: RECOMMENDATIONS FOR MONITORING

From the review of river concepts it becomes clear that river ecologists are developing over time a more integrated view of rivers. With regard to the spatial scale, attention was initially focused on the longitudinal dimension, then on the lateral dimension, and finally it was postulated that the river system should be studied on the scale of the whole catchment. Insight into connections between processes and structures and their temporal and spatial scale has increased.

The integration of ecological, geomorphological and hydrological knowledge has led to a more interdisciplinary approach to river science. The river concepts point to the need for an interdisciplinary approach to the development, monitoring and assessment of river ecosystem indicators. Measurement of river ecosystem functioning requires an integration of water quality, hydrology (discharge and stream velocity) and geomorphology describing the structure of the river channel, riparian zones and floodplains and, finally, ecological information on species diversity and abundance.

Both developments in river ecology and the importance of spatial characteristics indicate the *need for an integrated view at a catchment scale for the development of river ecosystem indicators*. Accordingly, the design, implementation and assessment of monitoring programmes should reflect such an integrated spatial view. In particular, monitoring programmes of the riparian countries of transboundary rivers need to be co-ordinated. The predicted spatial and temporal char*acteristics need to be incorporated in an overall transboundary measurement* strategy and to be taken into account in data assessment.

Comparision of these conclusions with the present monitoring programme of the International Rhine Commission executed within the framework of the actual policy plans viz. the Rhine Action Plan and the Ecological Master Plan of the Rhine (IKSR, 1993, IRC, 1994) leads to the following recommendations. Present measurements of water quality and river organisms in the monitoring programme (table 1) should be combined with information on land use and emissions

Water quality measurements		**Ecosystem assessment**
water	suspended matter	abundance of fish species
discharge	*Total Organic Carbon*	abundance of macro invertebrates
oxygen level	Total Phosphate	plankton:
Biological Oxygen Demand (BOD) of 5 dyas	Heavy metals (Hg,Cd, Pb, Zn, Cu, Ni, Cr, As	chlorophyl a
Total Organic Carbon (TOC)	Organic micropollutants	species composition
Dissolved Oxygen Demand (DOC)		phytosynthesis rate
Total Phosphate	sediment	primary production
Dissolved Phosphate	*sediment size*	*bacterial growth rate*
Ammonium	Total Organic Carbon	
Nitrate	Extractable Organic Halogenated Carbons (EOX)	
Chloride	Heavy metals (Hg, Cd, Pb, Cu, Ni, Cr, Zn, Mn, As)	
Heavy metals (Hg, Cd, Cr, Pb, Ni, Cu, Zn)	Organic micropollutants	
Organic micropollutants		
Radio activity	fish	
Cholinesterase inhibition	Heavy metals (Pb, Cd, Hg)	
Bacterial pollution	Organic micropollutants	

Table 1. Variables measured in the IRC monitoring programme

in the river catchment, hydrological and geomorphological river characteristics and lateral exchange of matter, which is provided by the river ecosystem indicators and variables. This information allows assessment of the impact of human activities on the river system. The entire cause-effect chain could be described and evaluated in an integrated way at a spatial scale of the river catchment, guided by the knowledge from the river concepts.

Since the river concepts are generic and descriptive, they do not prescribe definite indicators and quantitative assessment functions. Nevertheless the concepts indicate dominant variables, cause-effect relationships and spatial patterns. Deviations from the predictions of the river concepts, adapted to the particular hydrological, geomorphological and ecological characteristics of a particular river, could possibly indicate the extent of human disturbance. A process of `learning by doing' will clarify qualitative and quantitative relationships and allow improved ecological assessment of the river condition and improvement of the indicator selection with time and greater data availability. The monitoring of indicators will provide valuable information for the deduction of specific cause-effect relations. This will enable more reliable predictions of the ecological effects of human impacts and increase the predictive value of the river ecosystem indicators.

The concepts could also be used as a basis for the elaboration of an ecological `Leitbild' for the restoration of present rivers, in which natural processes and functions in rivers would be emphasized.

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AWIX: AN ALL-IN WATER QUALITY ASSESSMENT INDEX

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ABSTRACT

Water resource policy and management requires assessment methodologies which aggregate detailed knowledge on water resource quality to a comprehensive and simple quantitative form, tailored to the specific users at a (inter)national, regional or local level. This paper aims to contributie to the discussion on water assessment methodologies by presenting the results of a pilot study 'an all-in water quality assessment index (AWIX)'. In this study indicators and indices are used which provide information on the extent to which water quality meets the quality requirements for selected functions of the water system under study and on the extent to which selected issues cause pressure on this system. Hereby information is aggregated at a level of functions, issues and combinations of both, finally leading to an integrated water quality assessment index.

AWIX: AN ALL-IN WATER QUALITY ASSESSMENT INDEX

Worldwide, human activities exert a strong pressure on freshwater resources. This has led to increasing demands on policy and decision makers at (inter)national, regional and local levels for well-fundamented strategies and solutions in situations of conflicting interests. In water resource policy and management there is an increasing need for assessment methodologies for diagnosis and prognosis purposes in which the complete pressure-state-response chain is considered. Such assessment methodologies should aggregate detailed information to a comprehensive and simple quantitative form, to support the policy and decision making process. In aggregating information, indicators and indices are useful tools. An indicator is here defined as an observable and measurable quantity that has been assigned a significance beyond its face value. An index is defined as a compositie of indicators, comprising information at a higher level of aggregation. The discussion on water assessment methodologies, indicators and indices is still in full development. This paper aims to contributie to this discussion and focuses on the development of a general framework in water quality assessment meeting the following basic requirements:

- fitting (inter)national, regional and local policies. Common features of these policies are in general terms, firstly, that the problems around water resources are approached from the functional level. Principles and objectives are formulated to safeguard the multi-functional use of water bodies. Secondly, stresss to which the system is exposed is should be related to target groups to which policy can be directed.

- high flexibility. The general framework should be applicable under various socio-economic conditions, in data rich as well in data poor conditions and under different hydrological conditions.

Taking into account these requirements, a general approach is presented: Qan all-in water quality assessment index (AWIX)'. The AWIX approach makes use of indicators and indices which provide information on firstly, the extent to which water quality meets the quality requirements for selected functions of the water system under study (e.g. irrigation, drinking water supply, ecology) and secondly, the extent to which issues cause pressure on this water system (e.g. eutrophication, acidification, pollution with hazardous compounds).
Information is aggregated per issue and function (into a so-called IssueFunction Index) followed by aggregation at (1) issue level (into a so-called Issue Index, 11) and (2) function level (into a so-called Function Index, FI), and (3) for a combination of issues and functions (into a so-called Water Quality Index, WQI). Figure 1 illustraties these various indices in relation to functions and issues by a matrix of 2 selected functions and 2 selected issues. Any combination of number of functions and issues can be prensented in such a matrix. Comparing Function Indices gives insight into the extent the water quality meets the requirements for separate functions. Comparing Issue Indices gives insight into what extent a certain issue threatens the water system. This insight allows policy-directed measures to target groups which are directly related to these issues. The Water Quality Index provides information on overall water quality in time or space, in relation to the functions, taking issues that threaten water quality into consideration. Evaluation of the implemented policy is possible through a comparison of Water Quality Indices over time or space.

The Issue and Function Indices and the Water Quality Index are derived in a seven-step procedure: 1) to assign the functions which are considered essential in the water system under study in relation to the policy topics; 2) to assign the issues which are considered essential in the water system under study in relation to the policy topics; 3) to select relevant water quality parameters per function and issue; 4) to express water quality parameters selected in dimensionless values by calculating the fraction which meets the water quality standards set per function; 5) to aggregate the dimensionless values to Issue-Function indices; 6) to aggregate the Issue-Function indices to Function Indices and Issue Indices; 7) to aggregate the Function and Issue Indices to an all-in Water Quality Index (WQI). Aggregation can be carried out using different mathematical formulae and approaches in particular when calculating the Function and Issue Indices (step 6).
To evaluate the AWIX approach on its merits, it has been applied in two case studies 'European Rivers'and 'River Rhine'. 'European Rivers' is a water quality asssessment study comparing various European rivers. Issue, Function and Water Quality Indices have been calculated for European rivers in which three functions were distinguished (drinkingwater supply, irrigation and ecology), and three issues (eutrophication, pollution by heavy metals and organic pollution). From the calculated Function and Issue Indices, ecology has been shown to be under the most stress; irrigation under the least stress, with the most stress caused by eutrophication. From the Water Quality Index, the Eastem European rivers are shown to have a poorer water quality than the Western European rivers and the southem rivers in Western Europe under more stress than the northem rivers.

The other case study, 'River Rhine', is a water quality assessment study of the River Rhine in 1976, 1980 and 1990. The Issue, Function and Water Quality Indices have been calculated for the Rhine for the years 1976, 1980 and 1990, with three functions distinguished (drinking-water supply, irrigation and ecology) and four issues (eutrophication, heavy metals, salination and organic pollution). From the Function and Issue Indices for the three years, ecology has been shown to be under the most stress and irrigation under the least stress; in 1976 this stress was primarily caused by organic pollution, and in 1980 and 1990 by eutrophication. The Water Quality Index showed a strong improvement in water quality over 1976@ 1980 and 1990. In both case studies, the trends shown by the AWIX approach are in agreement with the generally accepted insights.

CONCLUSIONS

The AWIX approach is considered a flexible method allowing assessment of water quality in a highly simplified way, tailored to the needs of policy and management at an (inter)national, regional or local level. It can also be used in data rich and data poor situtions. Functions and issues can be selected depending on the actual policy topics. The method, however, requires reference, target or standard values for the water quality for each selected funcion. A standardized set of criteria for function-directed water quality is, however, often lacking; this can then limit the practical execution of the AWIX method. Comparison of Issue Indices in relation to functions and a comparison of Function Indices in relation to issues enable policy analists and policy makers to entangle the complex water management topics and provides a tool for priority setting.

TRANSFORMATION OF DATA INTO KEY MANAGEMENT INDICATORS

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ABSTRACT

Management of water quality in a major UK water services company is a complex process because of the large number of monitoring points, the complex regulations and increasing public awareness. This paper describes how Anglian Water transforms its data into management information using a highly automated system from planning of sampling to presentation of management information. The concept of 'level of service' and use of the properties of the binomial distribution are demonstrated. For transformation of raw data into key high level management indicators, data are assigned pass or fail status with respect to the appropriate rule. The binomial distribution is then used to assess the probability of non-compliance, for example, 95 percent certainty of failure for more than 5 percent of time is used for sewage treatment works. Use of non-parametric statistics enables construction of meaningful aggregated indicators such as compliant flow and indices based upon relative toxicities to be easily constructed without knowledge of the underlying data distributions.

Data collection is expensive, therefore it is essential that the right data are collected and utilised in a cost effective manner. Presentation of statistics is key to success and modern software and communications make it easy to distribute clear graphics. Other major factors contributing to successful use of high level measures include, realism, understandable by non-technical people and sensitivity i.e. amalgamating does not hide small changes.

INTRODUCTION

Anglian Water is geographically the largest of the ten privatised regional water companies in England and Wales. The group comprises of Anglian Water PLC the holding company, a UK operation consisting of Anglian Water Services Ltd. which holds the UK operating licence and Alpheus which operates private treatment plants under contract and an international operation comprising of Anglian Water International and Nordic Purac which provide contract and technical expertise across the globe.

Anglian Water Services operates in eastern England (Figure 1) and operates 1071 sewage treatment works, 4,128 sewage pumping stations and 31,000 km of sewer serving over 6 million people. The company also provides drinking water for 3.9 million people from 157 water treatment works, 400 reservoirs and towers and 33,400 km of water main. Anglian Water also controls over 3,000 industrial discharges to the sewers and disposes of over 138,000 dry tonnes of sludge per annum, mostly to farmland. Product quality is key to the success of this business and key indicators have been developed enabling managers at all levels to understand trends and their significance.

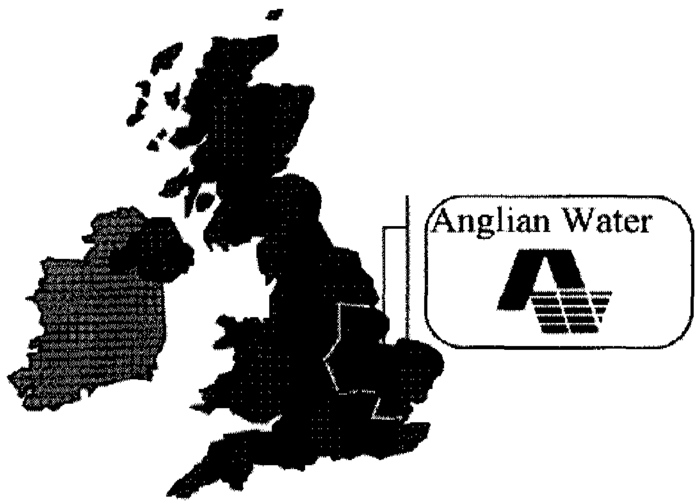


Figure 1: Location of Anglian Water

THE UK REGULATORY FRAMEWORK

The UK Government has established a rigorous framework for monitoring water quality in England and Wales incorporating European Commission (EC) Directives. The Drinking Water Inspectorate (DWI) regulates drinking water whilst the Environment Agency (EA) regulates emissions and monitors river, sea, air and soil quality. The EA was formed in 1996 from the National Rivers Authority (NRA), Her Majesty's Inspectorate of Pollution (HMIP) and the Waste Regulatory Authorities (WRAs).

DWI annually complete a detailed audit of every water company and publicly report on drinking water quality, compliance with monitoring requirements, formal undertakings given by companies to complete capital programmes within a defined time scale and incidents. Anglian Water ranks amongst the leading companies and much of this success is due to automated management of the sampling programme and interpretation of the results promptly advising management where action is needed. Anglian Water has to maintain a public register of water quality from which over 1000 people per annum seek information. This interest by the public is increasing every year.

The EA defines the quality limits for each discharge to the environment in a consent, which may also stipulate such things as screen size and telemetry links,. The EA carry out their own monitoring and produce public reports. All environmental and our discharge data are available to the public.

Together these agencies create a tight regulatory framework with a high degree of public awareness. Therefore, in order to maintain the confidence of our customers we must maintain high standards, and up to date knowledge of performance and trends is critical. In order to demonstrate the commitment Anglian Water has to quality, a brief but comprehensive paper is presented to the Director's Water Quality Group each week along with regular summary reports of specific matters such as bathing water quality. The remainder of this paper describes how we use our data and why we chose the indicators we use.

DATA PROCESSING

DATA CAPTURE

Reliable data are key to any management information system. Indicators and information systems soon lose credibility if incorrect reports and forecasts are made. Hence, we ensure that sufficient energy is input and our samplers are fully trained in order to maintain a high standard. The subsequent effect of poor samples is poor data, which leads to poor management decisions and strategy and eventually loss of customer confidence and loss of business. In order to demonstrate commitment to quality, sampling of drinking water is accredited to ISO 9002 and our analytical procedures are accredited under NAMAS (National Analytical and Measuring Accreditation Scheme) which traces measures back to national standards.

Our water quality data arises from three main sources. Firstly, the largest telemetry system in the UK water industry which we believe is one of the largest in the world, with over 7,500 outstations on 6,200 sites. Secondly, generation of over two million analytical results per annum by our laboratories and thirdly data from the EA. The telemetry data is primarily used for day to day operations giving rise to alarms via a central control centre and decisions about plant utilisation. Although telemetry provides some water quality data, mostly chlorine, turbidity and dissolved oxygen from remote sensors, laboratory analysis is the main source. These data are captured by a Laboratory Information Management System (LIMS) and transmitted to a central archive OASIS (Operational And Scientific Information System) for main processing. Over 90 % of these data are captured directly from instruments and development of robotics is playing a leading role in the strive for efficiency. Data transferred electronically from the EA is to be loaded into OASIS to maximise its benefit by making it available to everyone.

Quality control of data is an essential pre-requisite if customers are to believe what the results are telling them. Remote data capture provides many advantages, particularly elimination of transcription errors and delays. LIMS also provides a front line defence alerting analysts to exceedances of key standards or operational warning levels, AQC (Analytical Quality Control) errors and highlighting data that is outside the normal 95 percentile range for each parameter at each location so that prompt action may be taken when needed. Together, these front line checks ensure high quality data with corresponding customer confidence. Prior to LIMS the error rate was in the order of 1% and confidence correspondingly lower.

CENTRAL PROCESSING

The central computer system, OASIS, provides on-line graphics, compliance information and statistics. Example reports are given as Figures 2 and 3. Data from this system is then further processed producing hierarchical reports for trend monitoring at all levels, highlighting adverse trends, meeting legal requirements and planning future investment. The system is VAX based with access via PCs running Windows.

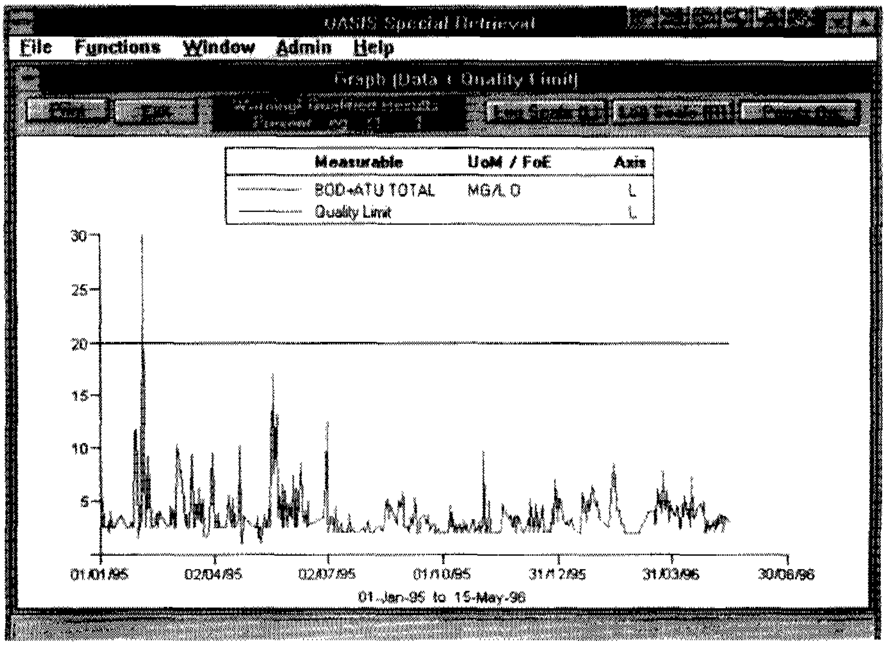
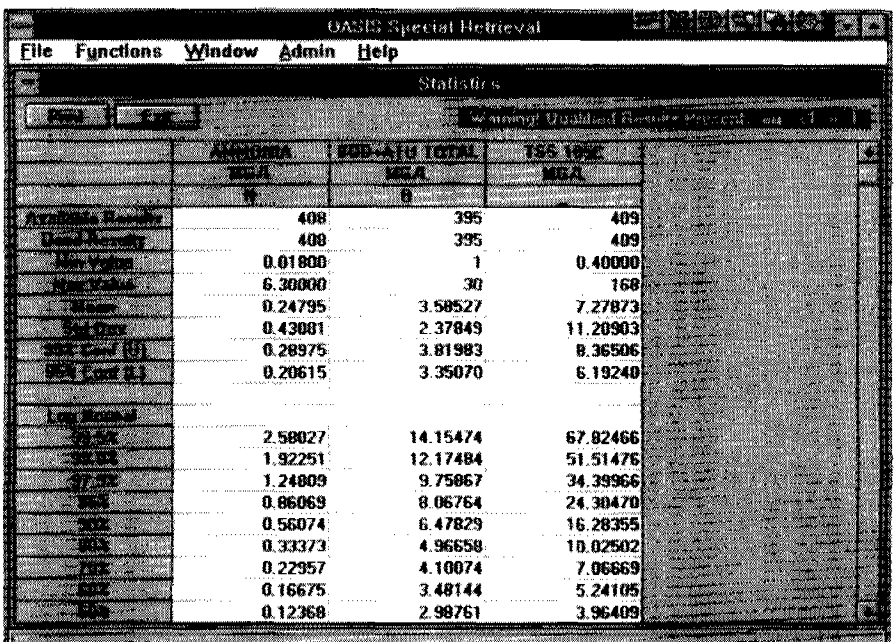


Figure 2: Online trend graph with legal limit



	AMMONIA	BOD+ATU TOTAL	TSS 105C
	MG/L	MG/L	MG/L
Available Results	408	395	409
Used Results	408	395	409
Min Value	0.01800	1	0.40000
Max Value	6.30000	30	168
Mean	0.24795	3.58527	7.27873
Std Dev	0.43081	2.37849	11.20903
95% Conf (U)	0.28975	3.81983	8.36506
95% Conf (L)	0.20615	3.35070	6.19240
Log Normal			
[illegible]	2.58027	14.15474	67.82466
[illegible]	1.92251	12.17484	51.51476
97.5%	1.24809	9.75867	34.39966
[illegible]	0.86069	8.06764	24.30470
[illegible]	0.56074	6.47829	16.28355
[illegible]	0.33373	4.96658	10.02502
[illegible]	0.22957	4.10074	7.06669
[illegible]	0.16675	3.48144	5.24105
[illegible]	0.12368	2.98761	3.96409

Figure 3: Online statistical report

In a large company such as Anglian Water and with the growth in use of PCs it is easy to create an environment where individuals develop 'islands of information'. These can be effective but they are not widely accessible. Therefore, we have created a configuration whereby all key data is held on centrally managed servers, but users can extract all the data they need for transformation into specialist reports. The main data and information pathways are summarised in Figure 4.

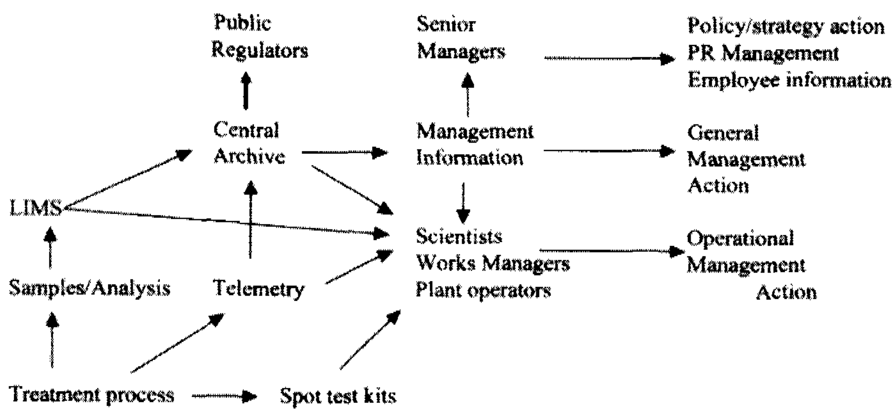


Figure 4: Main scientific data pathways in Anglian Water

MANAGEMENT REPORTING

MANAGEMENT PRINCIPLES

In order to understand and develop the process of management reporting we view the process as having two distinct stages, compliance and performance. We simply define compliance as 'did we pass or fail ?' and performance as 'are we getting better or worse regardless of compliance ?'. Compliance has been implemented by the system giving immediate compliance reports, in the form of alarms for non-compliant results, for every rule that needs to be applied to an analytical result. These rules can include locally set alarms to warn a plant manager that his quality is approaching a limit. In order to transform data into information for monitoring performance we have used the principle of 'Level of Service', which means choosing measures that reflect our customers view of us or our impact upon the environment. Such performance measures are not easily defined and can be quite complex in their construction, but the key criterion for the end product is that they are meaningful (reflect the truth) and can easily be understood by non-technical people.

Setting of management targets is also a key process. Whilst it is a fine objective to state that we will be 100% compliant in all cases, it is perhaps rather unrealistic. Anglian Water has adopted the principle of setting SMART (Specific, Measurable, Achievable, Realistic and Time bound) objectives. For example we may say that by year 5 our target is x% compliant and by year 10 X% compliant. A step wise movement towards the economic optimum. A target of 100% compliance is a fine objective but in many circumstances only attainable with excessive expenditure. An example is sewage treatment works compliance where our target at the moment is 98.5% moving upwards towards 99%. Our targets are also reviewed annually in the business planning process for their applicability. The world does not stand still, a flexible system which can simply be updated to reflect changing circumstances is essential.

As discussed by Breukel (1994) managers tend to only read abstracts and pictures, thus, pictorial reports are key to success. With modern software and communications pictorial reports can be readily disseminated and we use graphical presentation whenever possible.

STATISTICAL PRINCIPLES

Data collection is expensive, therefore it is imperative that monitoring is effectively targeted and statistics are used to maximum benefit. There are many different ways and statistical techniques available, but the core approach we use for management over viewing is the non-parametric binomial theorem, Equation 1. The reason for this is that you do not make any assumptions about the distribution of data. When we need to compute such statistics as 95 percentiles we generally assume log-normality. The main reasons being that most water company data is skewed to the right because treatment processes are generally designed to move concentrations towards zero, which is a boundary condition, and that seasonal influences bring about added complexity. It is rare in our experience to have sufficient data to be certain about the underlying distribution.

Equation 1

$$R = \frac{N!\, P^{(N-F)}\ (1-P)^{F}}{F!\ (N-F)!}$$

R = point on the binomial distribution
N = number of observations
F = number of observations exceeding the standard
P = probability of compliance with the standard

To estimate the proportion of time a discharge is within consent we compute P when N and F are known and R = 0.5, then compute P with R = 0.05 and 0.95 to obtain upper and lower 95 percent confidence limits. We then interpret the results according to the degree of confidence required. For discharges to the environment where we apply benefit of doubt rules, failure is ascribed when P for the upper limit dips below 0.95.

Another important consideration is eliminating measures which are easily affected by the number of observations made. For example, assessing the overall proportion of failures is simple to do but can easily be abused by adjusting analytical frequencies for compliant parameters. Again, use of non-parametric techniques minimises this.

SEWAGE TREATMENT

The choice of statistical techniques is also key to successful management reporting. There is no point in using techniques which do not reflect the regulatory framework. In England and Wales sewage treatment works compliance is based upon a 'benefit of doubt' technique. This has also recently been adopted by the EC in the Urban Waste Water Treatment Directive and recognises that influences beyond the dischargers control such as abnormal weather and industrial customers discharging illegal substances can affect performance. The criteria for the 'look-up table', as the technique is affectionately known, is that failure is not ascribed until we are 95% confident that the discharge has exceeded for more than 5% of the time for a single parameter. Although the statistics are complex, (Equation 1), the look-up table (Table 1) is simple to use (Ellis, 1985). The look-up table can be computed from Equation 1 by setting P=0.95, and iterating through available F for each N until R<0.05.

The performance of the sewage treatment process varies according to the season. In order to minimise such effects, the 'look-up table' and all our performance measures refer to the previous twelve months. A sinusoidal effect can still be observed in the data, this inertia reflects differences between corresponding seasons. It is not considered to be practical or desirable to attempt to compensate for such factors because the treatment process should be robust enough to cope with all but very exceptional conditions.

The key measures chosen for monitoring the performance of the sewage treatment process are :

- The percentage of discharges exceeding standards (Figure 5) - literally the absolute proportion which fail to meet the legal criterion, a compliance measure.
- The proportion of our population equivalent served by the legally failing works - in order to give an overall view of the magnitude of failure, a compliance measure.
- The proportion of non-compliant flow (Figure 5) - i.e. an estimation of the proportion of effluent which was outside the required standard discharged (Equation 1, 1-P when R < 0.05). This Level of Service measure, includes all data which was outside the standard regardless of the discharge passing or failing the look-up table and is constructed by taking the time value (Warn and Matthews, 1984) for each discharge and assembling them for all works weighted by population equivalent served. The reason we use population equivalent is that flow is only measured at the larger sites, generally greater than 1,000 person equivalents. Population equivalent is defined as the sum of resident population, holiday population (peak bed occupancy rate), tankered waste and industrial contribution.

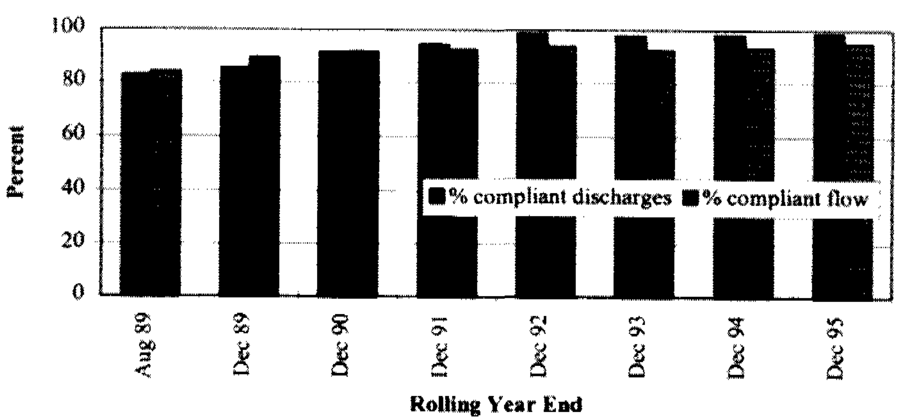


Figure 5: Sewage Treatment Works Compliance showing long term trend in the proportion of Compliant Discharges and estimated Flow within limits

These compliance measures reflect our impact on local environments as consent conditions are set according the needs of the receiving watercourse (Barnden et al 1986). The proportion of non-compliant flow is designed to reflect our impact upon the environment at large.

INDUSTRIAL DISCHARGES

In the UK the water companies are responsible for controlling industrial discharges to sewer whilst the EA controls industrial discharges to the environment. This means that the water companies have to control these discharges such that they do not result in the receiving treatment works exceeding its consent standards. The UK legislation also requires the water companies to levy charges on the industrialists such that the charge reflects the degree of treatment required. This is a direct example of application of the 'polluter pays principle'. In this context we are regulators as well as being regulated.

In order to manage these discharges the water companies issue them with consents which detail volumes, quality and other particular requirements generally as absolute limits. Whilst industrialists are notified of exceedances of their limits, and prosecuted if they are persistently or grossly exceeded, we use the same basic statistical technique as for sewage treatment to take the management view on a company level (Figure 6). In this instance we define persistently failing as 95% confidence that the limit has been exceeded for more than 5% of time (i.e. failing the look-up table, Table 1), gross failure as more than twice the limit exceeded on at least one occasion, occasional failure as one or more exceedance less than twice the limit (these are mutually exclusive) and satisfactory as total compliance. Not sampled is where expect samples to have been taken, sample not required are innocuous discharges such as laundrettes.

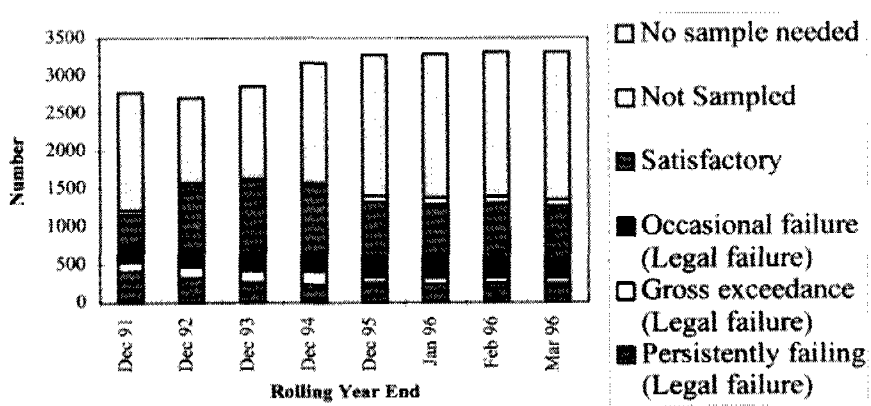


Figure 6: Chart showing trend in the various categories of Industrial Effluent Compliance

Number of measurements for a given parameter	Number of exceeding measurements allowed
4 - 7	1
8 -16	2
17 - 28	3
29 - 40	4
41 - 53	5
54 - 67	6
68 - 81	7
82 - 95	8

Table 1:The look-up table used in UK and the EC Urban Waste Water Treatment Directive for sewage treatment works compliance assessment. Based upon 95% confidence that a discharge has exceeded for more than 5% of time before assigning failure.

DRINKING WATER

The UK Drinking Water Regulations fully incorporate the EU Drinking Water Directive mostly setting standards as absolute maximums. Fourteen additions are made using annual or three monthly rolling means. Our system for producing, delivering and monitoring drinking water is fully audited by the DWI every year and a public report produced. There are also local health authorities and council environmental health officers who take a keen interest in drinking water quality. In addition, we have to maintain a public register of information. Any member of the public is entitled to full details, including analytical results, about their supply free of charge. This results in a high public profile and the cost of failure to act properly can be costly in terms of customer confidence.

In the assessment of drinking water quality we use two main performance measures in the form of indices. The reason for using indices is that we are assessing in our distribution network 150 parameters in 206 supply zones and 6 parameters at 400 storage reservoirs and water towers, i.e. 33,300 opportunities for failure. With point discharges we can make aggregated quantitative assessments, e.g. use of population equivalent data, with ease. In water distribution networks it is not possible in many instances, for example, where there are affects due to a specific length of pipe, to know the population affected. With this degree of complexity, simple techniques like those used for sewage treatment are not practical or sensitive enough to reflect the impact on public health, thus we developed in the mid eighties (Hayes et al, 1985) a drinking water index system which is still used today.

These indices are hierarchical in that index values are produced from an individual parameter up to a single value for the company as a whole. The index value for each parameter at each location is computed using binomial probability and when the failure rate exceeds 0.05 point deduction begins. Use of binomial theorem also reflects in the confidence limits the variable number of data values that you have. The individual index parameters are then grouped by type (Table 2) and for the zone or installation to produce a single value. These groupings are completed using weightings (Table 2) reflecting their public health significance. The weightings also mean that the index which ranges from 100 to 0 is sensitised, in fact, if every parameter failed the index would be a large negative number. . For example coliform exceedance, results in fifty times the point deduction compared to an oxidisability exceedance in the group and twenty five times overall, resulting in a dimension less index. The zone or installation index values are then grouped by operating unit up to the company as a whole. Examples of our company zone and reservoir and tower index trends are given as Figures 7 and 8. The zone index demonstrates sensitivity because although the proportion of exceedances is less than 0.5%, the satisfactory index is only 86. The reason for requiring sensitivity in aggregated indices is that variations in low but critical failures are clearly visible.

Group	Parameter	Group weighting	Total weighting
Organo-leptic	Colour	10	1
	Odour	30	5
Physio chemical	pH	3	1
	Conductivity	3	1
Undesirable in Excess	Nitrate	50	5
	Iron	20	5
	Oxidisabiblity	1	1
Toxicants	Arsenic	50	10
	Lead	50	10
Microbiological	Coliforms	50	25
	Colony counts	5	1
Hardness	Hardness	60	1
	Alkalinity	40	1
Pesticides	Atrazine	10	2
	Simazine	10	2
	Ioxynil	10	2

Table 2 Drinking Water Index:- Example parameters showing groupings and weightings group and overall aggregation

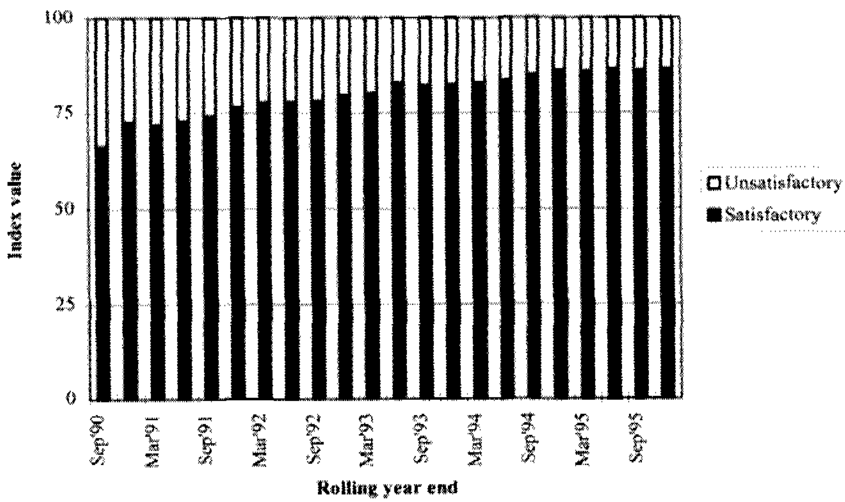


Figure7: Chart showing trend Drinking Water Zone Index

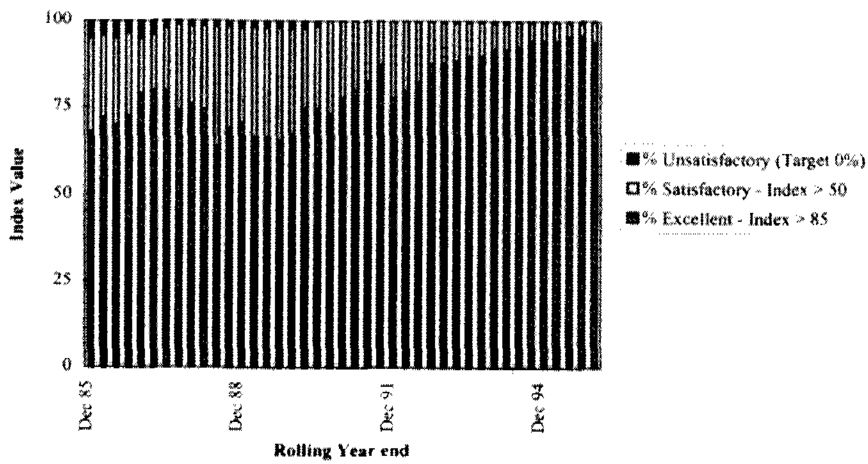


Figure 8: Chart showing trend in the Reservoir and Tower Microbiological Index

CUSTOMER CONFIDENCE

Although we have put in place a comprehensive management information system, this is of little value if our customers do not have confidence in the company or its products. In order to gauge this we also have a continuous customer tracking survey carried out by professional surveyors. Two example outputs are percent of customers very and fairly satisfied with a) tap water safety 79% and b) water quality discharged to rivers 25%. It may seem surprising how low these customer satisfaction indices are, but they are slowly increasing and compare favourably with other companies.

CONCLUSIONS

Collection and maintenance of data is expensive. It is therefore essential that the right data are collected in a reliable and cost effective manner and transformed into appropriate management indicators. These indicators need to be meaningful, sensitive to changes and easily understood by non-technical people. Flexibility is a key component of the system as business needs and customer requirements change but a balance must be maintained to prevent corruption of long term views. One method we have used to overcome this is use of fixed reference values enabling the true level of service to be visualised. Efficient communication of information by graphics is key to success and modern software and communications facilitates this.

The success of use of level of service measures described above is evidenced by the fact they are still used after many years, positive feedback from recipients and the continued quality of our products.

The main conclusion is that success depends on collecting the right data in a quality controlled manner, using the appropriate statistical tools and techniques and communicating the messages in a timely and understandable manner. Although these appear to be simple requirements, they are not very often or easily met and require considerable investment in knowledge and software to ensure an appropriate return on the investment in data collection.

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INTEGRATED APPROACH FOR CHEMICAL, BIOLOGICAL AND ECOTOXICOLOGICAL MONITORING - A TOOL FOR ENVIRONMENTAL MANAGEMENT

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ABSTRACT

Traditionally, environmental monitoring is done with a combination of chemical and biological tools. Chemical analysis is performed to assess potential ecological stresses imposed on the system by oxygen depletion and nutrient enrichment and to monitor the level of selected hazardous substances (heavy metals and persistent organic pollutants (POPs)). Biological monitoring is mainly focused on the assessment of the ecological structure (diversity) and to a lesser extent on the function. Recently, also an ecotoxicological monitoring tool has been launched, involving direct testing of environmental samples for their direct toxicity to selected laboratory test organisms. The introduction of ecotoxicological tools together with the traditional chemical and biological methods should be seen as a result of the increasing awareness of the complex contamination of the environment by a very high number of chemicals - a problem for which a chemical analytical approach alone would at present be technically impossible and also far too expensive to implement. The introduction of new monitoring tools and also the present-day increase in monitoring parameters accentuate the need for management systems to achieve cost-effective monitoring programmes. The headlines for such systems should be tailor-made monitoring and monitoring controlled by well-defined objectives. The primary monitoring of water in Denmark is linked to ground water (chemical residues), to waste water effluents (chemical residues and parameters of relevance to ecological impact assessment) and to surface water (parameters of importance to assessment of eutrophication). Tailor-made monitoring and monitoring integrating chemical, biological and ecotoxicological parameters is till now limited to a few applied research projects.

INTRODUCTION

Holistic approaches in environmental management have been done a.o. in the form of Environmental Impact Assessments (EIA) (World Bank, 1988, Van de Kraats, 1996) and Ecological Risk Assessments (ERA) (US-EPA, 1996). The general feature of these concepts is to (semi)quantify all major influences an environmental system is exposed to and to assess the potential (prospective) impact or risk of these to the various components of the environment. Other concepts are dealing with prospective risk assessment of specific compounds or products: generic environmental risk assessments of new and existing chemicals (European Commission, 1996), biocides (European Council, 1994), pesticides (European Comission, 1994) and pharmaceuticals (European Comission, 1994). These tools have been developed in order to meet the needs of the authorities for regulation of the use and emission of substances potentially hazardous to man and environment. For new substances, the methods are strictly prospective as the assessment is performed before the substances are marketed. For existing substances, existing monitoring data may be included in the assessment. The methodologies are so-called "whole-world" models, where inherent chemical, ecotoxicological and toxicological properties are linked to standard loads, emission factors and standard environmental properties (distribution and dilution).

Recently, an integrated effluent monitoring and assessment approach has been proposed by the

US-EPA (US-EPA, 1996) and by Sweden (Swedish Environmental Protection Agency, 1990), Denmark (Pedersen et al., 1995) and The Netherlands (Ministry of Housing, Spatial Planning and the Environment, 1995). Compared to the single chemicals approach outlined above, the effluent approach is site-specific, taking into account also the potential environmental risk of the complex mixture (ecotoxicological testing of the mixtures combined with chemical sum parameters). The risk predicted is therefore directly applicable for the design of future effluent monitoring and also for environmental monitoring.

While the above concepts are primarily focused on predictive/prospective assessments in a "risk assessment" framework (i.e. the elaboration of risk quotients: ratio of predicted environmental concentrations to predicted no-effect- concentrations), the retrospective monitoring and assessment of existing contamination levels and impacts have until now not been based on a similar transparent concept. In addition to linking chemical, ecotoxicological and biological monitoring approaches in a risk assessment framework, it is just as important to link the prospective and retrospective assessments in order to obtain an "iterative tailor-made assessment". An important starting point is, however, to elaborate clear (politically agreed) objectives for the assessments. The objectives should be sufficiently clear for the identification of goals and for the set-up of well-defined monitoring parameters.

PRESENT MONITORING APPROACHES AND THEIR LIMITATIONS

CHEMICAL MONITORING

Chemical monitoring is traditionally used for monitoring single substances with the primary objective of identifying and quantifying the major contaminants that may be of concern, for monitoring the compliance with water quality standards/limits or for monitoring levels of parameters (nutrients a.o.) of importance to ecological assessments (primary production). Very often, the number of substances and parameters measured are voluminous and not defined in line with the primary requirements for either retrospective or prospective environmental (biological/ecotoxicological) impact assessments. The parameters have often been identified on the basis of historical reasons and the tendency has been towards an expansion of the number of organic xenobiotic substances and metals monitored, as the technical ability of chemical analysis developed. Only rarely an initial, transparent evaluation has been made in order to identify the potential, high priority contaminants in the environment (industrial emission sources, use and release of persistent hazardous substances, etc.) and very seldom a systematic review and a regular update of the parameter list are made. According to a recent review of monitoring practices in Europe (UN/ECE Task Force on Monitoring & Assessment, 1996), more than 350 different parameters are more or less frequently monitored. For water bodies used for drinking water purposes, up to 90 parameters are routinely monitored
12 - 24 times a year (Germany, The Netherlands, a.o.). For water bodies without direct human uses, considerably fewer data are collected. Within the last ten years, many of the POPs have been restricted in use or prohibited (mercury, cadmium, lead, PCBs, DDT and other chlorinated organic pesticides). If the environmental concentration of such substances is continuously decreasing, there is actually no need for continued monitoring. The need for monitoring substances, continuously detected far below their environmental quality criteria or below the detection limits for quantification, could also be questioned.

ECOLOGICAL MONITORING

An important strength of well designed ecological monitoring is that the registration of integrated responses, due to (all) environmental pressures and anthropogenic changes, is possible for long periods. Ecological monitoring is most often performed with the primary objective to monitor the ecological structure (diversity) and, to a lesser extent, the function. In order to make sure that all effects are observed (i.e. to minimise the risk of the decision maker of being blamed for having overlooked a crucial parameter), a tendency similar to that of chemical monitoring has been seen: Ever more species have been included in the programmes. In these situations, the monitoring has been very resource demanding and has been a very sensitive tool for relatively few specialists able to identify the species. The natural variation in the species composition due to climatic conditions and/or interspecies competition is in many situations so high, however, that very long time series are required to unveil significant environmental effects. In addition, the general, unspecific nature of this ecological monitoring makes it difficult to make management decisions based on such results. Again, there is an urgent need for setting up clear monitoring objectives. Some ecological (biological) monitoring systems have relatively specific objectives (e.g. the "saprobic" fauna-index designed for reflecting effects of organic matter and oxygen depletion in streams). Prediction of which types of ecological impacts to be expected or which ecological function or structure to be at risk should be an important task in management oriented ecological monitoring programmes. Identification of the type of reaction of ecological systems to various stresses should be, if possible, based on reviews of the huge existing monitoring data and on information regarding the type and magnitude of the stresses of the area monitored. Such ecological reactions have been known for a long time regarding oxygen consuming organic matter and nutrients, but should also be possible to establish for various types of effluents contributing with hazardous residues.

ECOTOXICOLOGICAL MONITORING

Ecotoxicological monitoring is aimed at the assessment of the environmental impact of hazardous substances, either as a theoretical predictive assessment of chemical monitoring data (assessment of the potential effects of the single components quantified), as a direct measurement of the toxicity of samples of effluents to various laboratory cultures of organisms (often representatives from planktonic algae, crustaceans and fish) coupled to knowledge of the dispersion of the waste water in the recipient, or as a laboratory testing of environmental samples. The strength of the biotesting approach is that the combined toxic impact of all the hazardous components in the sample is accounted for - also those that have not been looked for in the chemical monitoring programme or that have not been detected above the detection limits of the analytical method applied. The primary weakness is that the testing organisms used may not be among the most sensitive organisms for the sample composition in question. Consequently, there may still be a toxic impact on some of the organisms in the compartment, even when the biotests do not signal any toxic impact. Another problem is that, very often, the methods used are screening methods (short-term methods primarily registering the acute toxicity) and that the data from such methods are not easily extrapolated to long-term chronic impact on the biota. Therefore, there is a need for the development of sufficiently sensitive methods based on sensitive life stages, in-vitro systems or biomarkers that are interpretable in an ecological context. In practice, ecotoxicological monitoring should be part of the concept for ecological monitoring, as the overall objective of ecotoxicological monitoring is to predict the risk of impact on ecological systems - an impact which by proper means should be retrospectively monitored by ecological tools.

INTEGRATED CHEMICAL, ECOLOGICAL AND ECOTOXICOLOGICAL MONITORING

The tools used for monitoring should reflect the monitoring objectives. The most frequent objectives of monitoring are linked to assessment of ecological quality, human usage of the environment (drinking water, fishery, food quality) and control of enforcement of effluent reductions and use of regulations of (e.g.) chemicals (efficiency of waste water treatment, decrease in concentration of regulated chemicals). For each of these objectives, a number of potential, predictive/-prospective and retrospective assessment and monitoring tools may be identified (Figure 1). According to this logic, ecotoxicological effluent and recipient monitoring is primarily a prospective tool for ecological impact assessments and should thus be firmly linked to ecological (retrospective) monitoring. In addition to highlighting the risk of toxic effects in the environment, the ecotoxicological monitoring may also be used for identification of the structure (or function) that is primarily at risk and which should especially be focused in the ecological monitoring. Such

objectives	predictive tools	retrospective tools
Protection of ecological structure and function	Ecological risk assessment based on estimated loads of nutrients, organic matter and chemicals Ecological bioassays Chemical analusis for ecologically relevant parameters in effluents (nitrogen, phosphorus, BOD) Ecotoxicological testing of effluents Ecotoxicological testing of environmental samples (water, sediment) Modelling the fate of contaminants	Ecological surveys Chemical analysis for ecologically relevant parameters in environmental samples (nitrogen, phosphorus, BOD) Effect-related biomarkers Estimates of contaminant loads in top-predators (fish)
Protection of drinking water quality	Risk assessment and priority setting based on estimates of loads of hazardous substances Chemical analysis - high priority substances in emissions Modelling the fate of contaminants	Chemical analysis for high priority substances Exposure related biomarkers of relevance for evance for human risk
Protection of fishery and food quality	Ecological risk assessment based on estimated loads of nutrients, organic matter and chemicals Risk assessment regarding protection of fish populations Estimates of critical loads of hazardous chemicals for fish and for consumers protection	Fish population monitoring Chemicals analysis of contaminants in fish (compared to estimated critical loads for human consumption and fish survival)
Trends in high priority contaminant concentration	Identification of emission of hazardous substances based on assessment of sources Priority setting based on environmental hazard assessment and indentification of high risk compartments	Chemical analysis for high priority substances in high risk compartments

Figure 1: Examples of predictive and corresponding retrospective tools related to monitoring objectives.

combined ecotoxicological effluent and receiving water monitoring and ecological monitoring have been performed in USA (US-EPA, 1994) and in Sweden (Swedish Environmental Protection Agency, 1990) and relatively good correlations between predictions based on ecotoxicological monitoring and the reported impact as revealed by the ecological monitoring have been found.

For both ecotoxicological, ecological and chemical tools, there is, however, a serious need for improvement. Ecotoxicological testing methods more sensitive than the existing methods should be developed in order to reduce the risk of false negative results. The sensitivity should, however, be geared to the objective of the tools: to predict effect thresholds for protection of ecosystem structure and function at the protection level defined by the society. Some of the biomarkers may therefore not be directly applicable in this context, due to their very high sensitivity as they often detect the exposure to a stressor only and not a relevant ecological interpretable effect.

A promising new tool for assessment of critical loads of chemicals in fish in relation to protection of fish populations is at present under development (van Wezel, 1995; Petersen et al., 1997). The concept behind this tool is that chemical substances having the same toxicity mechanisms will have approximately identical toxicity thresholds, measured as internal body burdens (doses). Chemical sum parameters primarily representing such substances may therefore be applied for assessment of the risk of such effects on wildlife (fish) populations. These tools have not yet been sufficiently field validated.

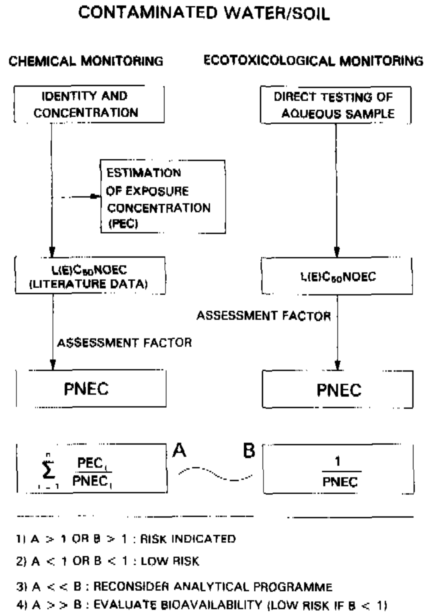

Figure 2: Integrated assessment of environmental risk based on 1) chemically quantified residues and evaluation of toxic thresholds based on literature data, and 2) direct ecotoxicological testing of the complex mixtures. PEC: Predicated Environmental Concentration, L(E)C50: Concentrations lethal to 50 per cent of the tested population, NOEC: No-Observed Effect Concentrations after chronic exposures, PNEC: Predicted No-Effect Concentration.

Toxic effects of chemicals may lead to long-term effects, which are manifested by the disappearance of certain sensitive species, or to a temporal change in the age structure of populations - effects, which may be overlooked by the general ecological monitoring. To ensure that general ecological monitoring unveils significant environmental effects/changes, it is thus necessary to include a great number of organisms in order to examine the biological structure in relatively great detail and to collect long time series.

The development of chemical analytical techniques has primarily been guided by the general concern for contamination with persistent bioaccumulative substances like heavy metals, PAHs, PCBs and chlorinated insecticides. A number of other potentially hazardous substances are increasingly used and thus emitted to the environment: detergents, organic antifoulants, herbicides a.o. For many of these, the analytical technique is not sufficiently developed for detecting concentrations at (or below) their toxic thresholds. Other problems are related to the determination of bioavailable concentrations, extraction from complex matrices and - the most serious problem - the ability to identify only a small fraction of the very high number of chemicals expected to be emitted to the environment. The last mentioned aspect has been addressed in a number of research projects where the objective is to characterise well defined physico-chemical fractions of the mixture by ecotoxicological testing methods and thus to apply an environmental risk assessment framework to fractions of the sample and not to rely only on relatively few analytically identified residues (US-EPA, 1988). This type of methodology is very promising and might form an important part of the next generation of monitoring tools for ecological risk assessment when fully developed and validated. Similar tools should also be used for the assessment of the risk to human beings of using potentially contaminated surface and ground water, but sufficiently safe screening methods for identification of this risk are at present not available. In-vitro toxicity screening methods for chemical testing may in future be applicable tools also for testing complex mixtures.

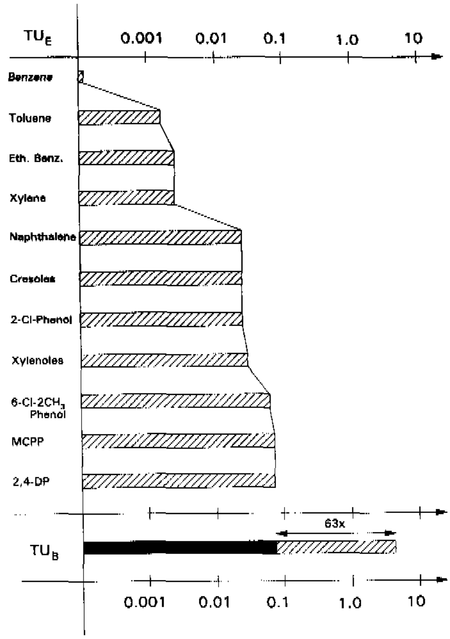

Figure 3: Toxicity of chemical residues detected in leachate from a Danish landfill. Data regarding toxicity to *Daphnia magna* have been derived from the literature (concentrations immobilizing 50 per cent of the organisms after 48 hrs of exposure). TU$_E$: Toxic units for each chemical as the ratio of the measured concentration and the IC50 value. TU$_B$: Toxicity measured for the intact leachate to *Daphnia magna* as the reciprocal of the IC50 value.

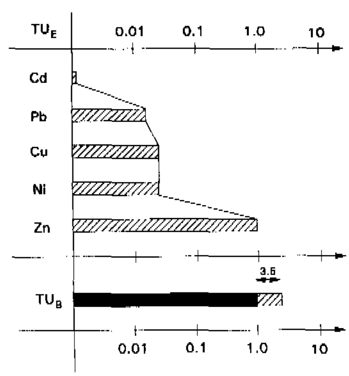

Figure 4: Toxicity of heavy metals detected in leachate from a Danish landfill. Refer to Figure 3 for further explanation.

Effect assessments based on specific analytical data and ecotoxicological testing of the complex mixtures may be compared by application of the toxicity addition model (Figure 2). The additivity model assumes that the toxic effects of chemical mixtures may be assessed by simply summing up the toxic unit of each chemical, defined as the ratio between the concentration of each chemical in the mixture (predicted or measured concentration) and the effect concentration (or the predicted no-effect concentration, PNEC) of the chemical. That this model may be usable in some cases was demonstrated at a landfill investigation (Figures 3 and 4) (Kristensen, 1991). Based on knowledge of the composition of the landfill, a chemical analytical programme was conducted on leachate from the landfill. A comparison of the results of ecotoxicity testing of the

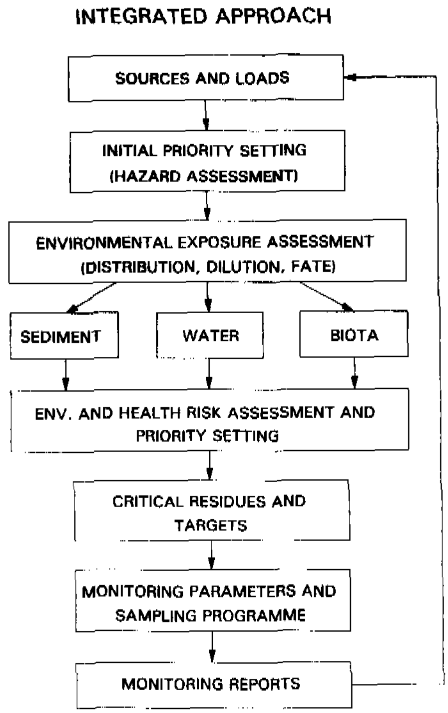

Figure 5: Integrated prospective and retrospective monitoring and assessment strategy. See text for further explanation.

leachate and the summing up of toxicity of each chemical detected revealed that the measured toxicity highly exceeded the toxicity accounted for by the chemical analysis (Figure 3). Based on observations of the concentration-response curves, the contribution of heavy metals was suggested by the ecotoxicologists. The following measurement of concentrations of heavy metals was approximately able to account for the earlier unaccounted toxicity (Figure 4).

An example of an integrated monitoring and assessment methodology is outlined in Figure 5. The methodology should include the following steps: Assessment of the sources of pollutants in the catchment area and the loads to the recipient; an initial priority setting in order to exclude non-significant pollutants; initial environmental exposure assessment in order to achieve gross figures regarding concentrations in various environmental compartments (sediment, water and biota); environmental and health risk assessment in order to assess the potential impact of the estimated concentrations/loads on the environment and human health (eutrophication, toxic impact, oxygen depletion, long-term build-up of POPs, contamination of drinking water resources, etc.); priority setting of contaminants and groups of contaminants based on risk quotients and quality objectives; selection of a suitable monitoring strategy, including a reporting and assessment methodology; and elaboration of a review procedure for the monitoring programme.

In the planning of the monitoring programme, all three monitoring tools have to be considered. E.g., if the risk quotient in relation to the quality objective indicates that a generally unacceptable toxic impact on the biota cannot be excluded, due to the emission of toxic effluents or many substances at critical hazardous levels, an ecotoxicological monitoring parameter should be included. This parameter should be supported by relevant ecological monitoring measures looking for the most likely ecological signal of ecotoxicological impacts and by relevant chemical measurements for indicator parameters. The design of the ecological and chemical programme with respect to parameters, location of stations and frequency has to be made on the basis of load estimates and of prospective assessments.

CONCLUSIONS

In future monitoring activities, an initial planning of the monitoring programme should as far as possible be based on environmental and human health risk assessments, focusing the protection of environment and human health according to defined quality objectives. The use of an integrated monitoring approach should be applied, selecting relevant chemical, ecotoxicological and ecological monitoring tools judged relevant for the objectives of the monitoring. The future monitoring programmes should be planned and conducted in a more proactive fashion, only including those parameters that are of relevance to impact assessment or which should be followed to control the success of mitigating activities in the area. The programme should be regularly reviewed in order to take into account the dynamic changes in the emissions from the catchment area and the resulting changes in risk quotients for risk to ecological systems and human health.

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REVIEW OF THE INFORMATION CO-ORDINATION ACTIVITIES FOR THE TRANSNATIONAL MONITORING NETWORK OF THE DANUBE RIVER BASIN

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ABSTRACT

The Environment Programme for the Danube River Basin (EPDRB) was signed in 1991. For the development of the Transnational Monitoring Network (TNMN) a special Project has been designed. Eleven Danubian countries have been involved in the implementation and realisation of this System. Since 1992 the establishment and development of the System has been managed by the Monitoring, Laboratory and Information Management Sub-Group of the Programme (MLIMSG) under the advising and control of the Programme Coordination Unit (PCU) of the Environmental Programme for the Danube River Basin (EPDRB).

The Information Management Working Group (IMWG) is a substructure of the MLIMSG and it has been established in 1994.

IMPORTANCE OF INFORMATION IN ENVIRONMENTAL MANAGEMENT

Information is the base in the decision making process of the environmental management e.g. water quality management. It is used for this purpose on local, regional, national and international levels. On a local level the information is used for control purposes, for issuing permits and for identifying sanctions in cases of standard violation. On a regional and river basin levels it is used for the analysis and evaluation of alternative policies, specific treatment actions, prioritisation of problems and action plans. On a national level strategies and policies are made, regulatory changes are proposed (laws, regulations, fines and standards) and national action programmes are developed. On a international level, concerning international rivers, conventions and agreements are prepared, joint programmes for pollution abatement and water quality improvement are developed and transboundary impact is assessed.

In all these cases the decision analysis needs reliable, compatible and sufficient information. There is a need in these cases for a well organised, computerised, system to be organise the collected data and information on water quality, emissions, accidents and regulations and institutions concerned.

HISTORY

In 1985 in Bucharest, Romania 11 Danubian countries signed the Bucharest Declaration (BD). BD is a document for the co-operation of states' efforts to carry out joint control of the environmental state of the Danube river.

Some general objectives of the BD are:

- The water resources conservation and the rational use, prevention and control of the water quantity of the Danube river;
- International co-operation for establishing a long-term framework for systematic observation of the Danube river water quality.

The initial activity of the BD has been to set up common monitoring stations on the Danube river at country borders.

In September 1991 in Sofia, Bulgaria, the Environmental Programme for the Danube River Basin (EPDRB) was established. One of the main objectives of the Programme is to develop, organise and implement the TNMN for the whole Danube basin including Measurement, Laboratory and Information Management (MLIM) methods with participation of the all Danubian countries.

In June 1994, in Sofia again, the Convention of the Protection and Sustainable use of Danube River (DRPC) was signed. The overall objective of the convention is to establish a monitoring (measuring), laboratory analysis and data management system for the Danube river basin.

The preparation of the design and the set-up of the MLIM has been supported financially by the Commission of the European Communities - PHARE Programme (Poland and Hungary Assistance for the Reconstruction of the Economy - an EC grant finance programme for Central and Eastern Europe) as a priority of the EPDRB and has been co-ordinated by the Programme Co-ordination Unit (PCU). The further development of the system is planned for the period 1996-98 to be conducted and financed under EPDRB.

MONITORING, LABORATORY AND INFORMATION MANAGEMENT (MLIM) / INFORMATION MANAGEMENT WORKING GROUP (IMWG)

The overall objective of the MLIM Sub Group (SG) and TNMN is to create a strengthened and more strategic approach to environmental information management for the Danube River Basin. In this regard, the results will allow presentation of the information to support decision-making and action at the International (Danube) and National level to prevent, control and reduce the transboundary pollution impact and the impact on the Black Sea.

The work has been designed and carried out by the MLIMSG with the technical support of WTV consortium.

During the first half of 1994 the MLIM SG has been up to three purpose designed Working Groups dealing with:

- Monitoring/sampling (Monitoring Working Group-MWG)
- Laboratory management (Laboratory Management Working Group - LMWG)
- Information management (Information Management Working Group - IMWG)

The general strategy for development of the system is to focus on the quality of the sampling, laboratory analysis and information management and finally, all these activities result in data with well defined characteristics. The processes and activities, used to obtain the data, are formal and controllable to ensure high quality data in the Information system.

To allow the exchange of data between different countries, and to facilitate the development of standard functions, MLIMSG is implementing a common Data Exchange File Format (DEFF).

The Data Exchange File Format (DEFF) is simple and flexible. It is intended to form the interface between requirements of the Transnational Monitoring Network (TNMN) and different National Information Systems (NIS). As a start, the results on the analyses from the TNMN will be stored directly in the DEFF, using a Personal Computer. Later, as each country adapts its organisation and national information system, the DEFF could be extended to cover all data exchange. DEFF incorporates possibilities to export in an appropriate way useful information as time series and can answer "daily work questions" that can arise in the process of turning data into information.

The benefits from the use of the DEFF are that local experts can operate it resulting in lower cost, and pride of ownership leading to motivation to maintain and improve it. Historical data can also be stored easily but its quality may be uncertain. The flexibility of the system allows each country to implement DEFF, in its own way, within its own organisations.

All riparian countries are responsible for the collection of data for the TNMN. The requirements for data quality in the TNMN are extended naturally into each National Information System (NIS).

Validation of the data can be accomplished in a number of ways. Values are checked against detection limits and sensible upper and lower limits.

The data storage system consists of seven tables. Five of them are digital tables with the actual measurements and information on samples. These are the ones that have to be exchanged on a regular basis. The others with relatively fixed data can be updated if needed, but only after confirmation by the organisation concerned (for instance by an International Working Group).

A programme has therefore been designed and it is now being implemented in accordance with the recommendations made. It is steered by MLIMSG and its IMWG. This includes high priority sampling, monitoring and the acquisition of computer equipment. This is to be completed in 1996 in order to establish a Transnational Monitoring Network for reporting information on basin-wide issues.

The organization and information flow diagram of the TNMN is given in the next Figure .

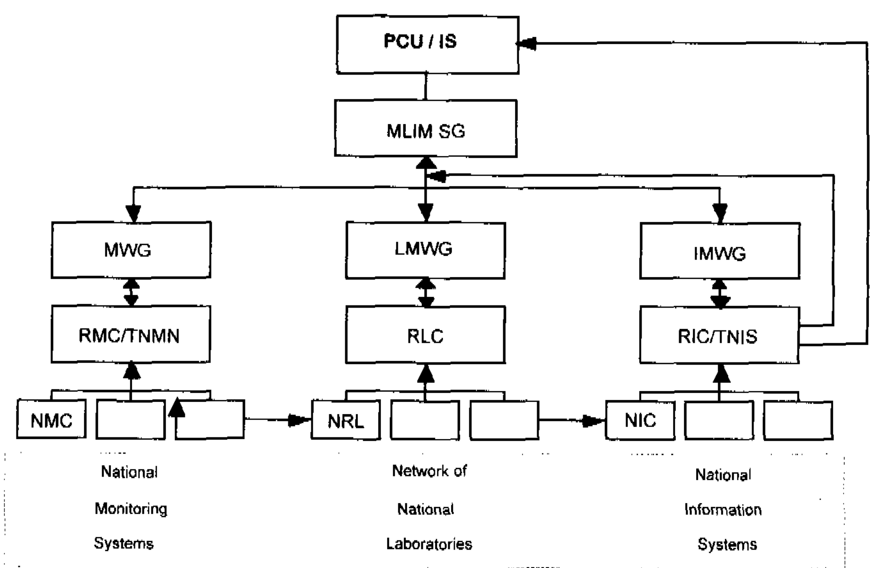

Figure 1: Organization and information flow diagram of the TNMN PCU = Programme Co-ordination Unit)

As a pilot the information system will use information from each country of nominated TNMN stations for agreed determinands and period of time. All data from the Bucharest Declaration's stations, which are fully included in the TNMN (they will be sampled in the first phase of the implementation of the TNMN)will be dealt with similarly.

ANALYSIS AND ASSESSMENT OF THE CURLRENT SITUATION

For the most part, the Danubian countries have systems for collecting samples, analysing samples and laboratory data checking, validation, data processing and exchange about information of ambient water quality and quantity. They are all very different and it is impossible to stick to one common information system for the whole Danube river basin. Base differences between them are:

- collection of the data;
- storage of the data;
- sampling standards and methods;
- list of determinands;
- methods of analysis;
- data base structure;
- methods of the data validation and data checking;
- data processing;
- output information;
- protocols of exchange of information;
- methods of estimation of the water river statement;
- standards and classification of river water

All these differences have leaded to development of a common project to prepare the Transnational Monitoring Network.

An estimation of the accuracy level of the data is given in Table 1.

	data	estimation level
Kind of collected data and evaluation	Physical	Moderate
	Chemical	Moderate
	Nutrients	Moderate
	Inorganic major	Moderate
	Iinorganic trace	Moderate
	Inorganic others	Moderate
	Oorganic non-specific	Moderate
	Oorganic specific	Fair
	Radioactivity	Moderate
	Biological	None
	Micro biological	Fair
	Water quantity	Moderate
Distribution and data exchange	Physical	Moderate
	Chemical	Fair
	Nutrients	Fair
	Inorganic major	Moderate
	Iinorganic trace	Fair
	Inorganic others	Fair
	Oorganic non-specific	Fair
	Oorganic specific	Fair
	Radioactivity	Fair
	Biological	None
	Micro biological	None
	Water quantity	Moderate

Table 1: Estimation of accuracy level

The main problems are lack of common methods of analysis, quality of the collected data, its historical length, data exchange protocol and data processing tools. The assessment of the state of Danube water has to be made but good time series of comparable data with high quality are a prerequisite.

The data available in TNMN is mainly of moderate quality. The distribution and data exchange level in both systems are in the initial stage of development. Theoretically, they are well organised but when the system will be put in operation it can be truly tested.

TNMN has started to operate on an experimental level using historical data from the Bucharest Declaration and in two to three years good quality data will be available. This will help ease the implementation of statistical analysis and assessment and will facilitate taking measures to improve the quality of the Danube river water. A good organisation of the measurement of water flow on all designed monitoring stations is important because this is one of the major gaps of the information supply. Without reliable data on water quantity it is impossible to make any analyses, profiles, load estimation and to prepare alternatives for the decision making process. The Data Exchange File Format has been developed for the purposes of data distribution in TNMN and in 1996 it operates on a basic level.

PRIORITY ACTIONS TO IMPROVE THE INFORMATION MANAGEMENT

According to the provisions of the Danube River Protection Convention all the activities from the implementation of the Bucharest Declaration (1985) have to be established under its responsibility.

The use of the Bucharest Declaration (BD) activities, data, results and experience, the existing network of institutions, laboratories and experts is a framework, and ground for stepwise implementation, of the MLIM SG's proposals and results.

Fundamental elements ofthe Transnational Information System (TNMN) are:

- The nomination of National Information Focal Points in each of the Danube riparian states. Practically, there has to be a small staff at the National Information Centres (NIC). The NIC will be the responsible operational unit. NIC operations are focused on the collection of ambient water quality data;

- The nomination of a Regional Information Centre (RIC) to co-ordinate all activities of NIC's, to organise communication activities and protocols between them and to report to PCU and to all interested parties.

For a proper operation of the TNMN the nomination of NICs is very important. This nomination is in process now. These NICs should have the following responsibilities:

- acting as 'centres of excellence' for information management
- to collect all available and relevant information on surface water quality in the Danube and its tributaries, including basic hydrological data
- to guard the quality of the data (accuracy. correctness, reliability etc.) - to control the format of the data exchange
- to distribute data from the TNMN
- to develop new methods of data analysis
- to advise national and international parties
- to propose improvements to the TNMN working group on information management
- to take responsibility in introducing accepted international policy on a national level as agreed by the Internatinal Secretariat of the Commission

- to take active participation in the TNMN
- to be involved in training regional staff
- to coordinate reporting and use of resulting data

The main tasks of the RIC are:

- Organising the data exchange activities and protocols of ambient water quality data exchange between NIC's and PCU and IC;
- Realising relationships with other information systems in the EPDRB and DRPC;
- Development of the agreed updating of the Information system;
- Reporting to PCU and IC and to all interested parties.

In accordance with recommendations of MLIMSG each NIC should have an information expert (graduated engineer/scientist), who would oversee the collection and handling of data for the TNMN and be responsible for its transmittal to the central group on time and at the agreed level of quality. The expert should be assisted by two more members of staff, one of them a programmer and the second, while probably not a graduate, familiar with the use of computers. All three members should be on a partial-time in fulfilling the needs of the TNMN.

In order to implement DEFF, each country has to develop a data exchange software interface that can select data from its own database and store it into DEFF.

The development and implementation of this software interface should be done by local experts in each country. This is a flexible and cheap solution which enables data exchange between different information systems. The crucial characteristic of the file structure will not be the exchange format itself but the quality of the data in it. Because the local interface is not developed yet (it should be developed starting at the beginning of 1997) it has been decided by the IMWG and MLIMSG to use DEFF as a data entry tool as the DEFF is a user friendly table oriented software written on Paradox 5.0 for Windows.

The actions implemented under the Programme are as follows:

- support the decisions in Phase I (preparation phase for investment activities or for immediate, of the highest investment priorities) and Phase II (continuation of Phase I activities, to support investment actions and to support the Strategic Action Plan) of the Danube Programme;
- provide ready access to information at a reasonable cost for governmental staff, research scientists and institutes, non-governmental organisations and the general public;
- build-on existing systems or procedures and involve appropriate human resources development activities.

Working within the framework of the internationally funded Environmental Programme for the Danube River basin, and addressing the targets and actions set in Strategic Action Plan according to the provisions of the Convention, the MLIM Sub-Group should:

- provide an information management system for TNMN, that supports the efficient exchange of data and information on transboundary impact; provides useful management information for policy development and for decision making; and interfaces with national information systems in each riparian country,
- provide training opportunities at levels appropriate to the needs of each Danube riparian country - the derivation of the data-exchange information systems and establish the facilities and using existing systems.

These prime objectives are to be achieved, and other tasks as identified in the Implementation Plan of the TNMN. Experts of the Information Management Working Group should in 2 years solve the following problems:

- Testing of DEFF programme
- Produce final version of a DEFF system
- Prepare for application of DEFF system
- Develop the interface of national databases with the DEFF
- Finalise incorporation of Bucharest Declaration data

The future information strategy in the TNMN for the Danube River Basin will include and link the information flows and activities on a national and international level. Several information systems should co-ordinate this activities. On international level the joint information strategy will be implemented through the international joint system such as shown below:

- Transnational Monitoring Network and Emission control system for the processing and the concentration of the data on the river water quality and pollution;
- Accident Emergency Warning System to deal with accidents and transboundary pollution;
- Danube Information System, Danube Environmental Geographic Information System and Danube Environmental Decision Support System to support decision making process, programme preparation and exchange of information in the most appropriate way.

On a national level the important activities will be supported and implemented by the National systems already existing in the countries such as:

- National Monitoring Systems;
- National environmenta1 Information Systems; - National Technical Information Systems;
- National Emission Control Systems;
- Principle International Alert Centers (for the Accident Emergency Warning System for the Danube river basin).

The information management will provide support for the environmental protection and the sustainable development. The TNMN will support the water quality control monitoring, the measurements, the data handling and the data transmission.

The unified systems together with the network of experts already in place will provide the basis of cooperation and the joint efforts in the Danube region.

To make this process sustainable and for the implementation of the overall information strategy, the financial support for the countries in the transition from the EPDRB to the International Commission of the DRPC has to continue for some period of time. Another very important issue is the strategy to be supported by the Danubian countries, the Task Force of the EPDR1B and by the International Secretariat of the DRPC.

EXPECTED DAMAGE

A surrogate measure for quantifying the advantages of groundwater monitoring networks for water management

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ABSTRACT

Groundwater monitoring networks for water management supply information which can be used as a basis for making decisions. It is rarely possible to quantify the economic value of these decisions in a simple and objective manner. This article proposes a surrogate measure of the advantages of groundwater monitoring networks: "expected damage". This is presented as the economic loss which results from less than ideal decisions which, in turn, are the result of uncertain information. It demonstrates how any disadvantages to certain decisions can be ascertained, and which factors play a role in this procedure. The concept of expected damage is illustrated with a practical example.

DEFINITION OF THE PROBLEM

Monitoring networks provide information which can be used in the decision-making process. Because they have an economic value, the decisions themselves are the advantages of the monitoring network. Many of the publications which have appeared on this subject in the past twenty years have used statistical concepts to design groundwater monitoring networks. In these publications, optimal monitoring is usually defined as sites or frequencies which achieve the greatest reduction in the degree of uncertainty of the information. The economic advantage of a reduction in this degree of uncertainty is rarely analysed. Evaluating the economic advantage of groundwater monitoring networks for water management actually means evaluating the economic value of decisions. The outcome of these decisions concerns aspects such as cleaner water, enhanced ecological value and/or better health. It is rarely easy to determine the economic value of these aspects - particularly in the case of strategic water management. The political motives which played a role in the decision-making process pose an additional problem when attempting to determine the advantages. In many cases, water management decisions are not only based on information concerning the water system, but on political motives as well. This creates additional problems with respect to the quantification of the advantages. For example, to what extent can these advantages be attributed to the geo-hydrological information, and to what extent can they be attributed to political motives? It is even possible that a decision is inconsistent with the information (on the water system) supplied by the monitoring network. If that is the case, it can even be argued that the monitoring system has been a waste of effort. Clearly, the quantification of the advantages is an extremely difficult and subjective task.
A ìsurrogateî measure of the advantages of a groundwater monitoring network is described below. It can be used as a means of assessing the advantages of (groundwater) monitoring networks more easily and more objectively.

MEASURES FOR QUANTIFYING ADVANTAGES

A surrogate measure of advantages is required in order to illustrate the economic value of measuring and monitoring systems. One approach is to establish the damage arising from decisions which were based on inadequate information and which were, therefore, incorrect. Which means that the starting point is not the value of the information, but the "expected damage" arising from the lack of reliable information. Within the context of groundwater measuring and monitoring networks, this starting point implies establishing the extent to which the economic damage is the result of geo-hydrological circumstances. There are several advantages to this approach. Firstly, there are many generally acknowledged methods for estimating the damage resulting from geo-hydrological conditions. Secondly, the advantages of measuring and monitoring systems are considered in terms of the information supplied. And so the political element in the decision-making process plays no part in the quantification of the advantages of groundwater monitoring networks.

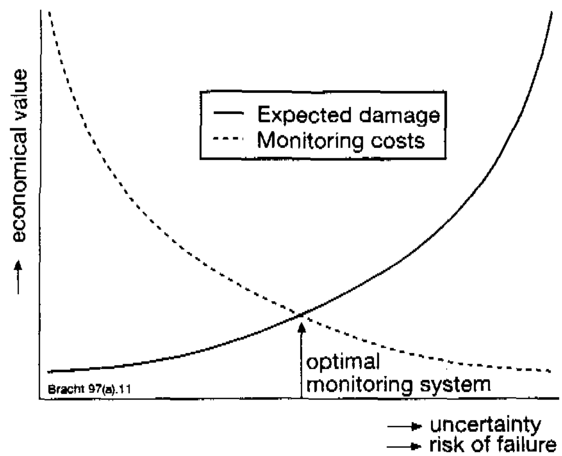

Figure 1 Relationship between uncertainty about the information, risk of failure, and expected damage.

This entails adopting the following basic assumptions:
The minimal expected damage is equal to zero. It is thus assumed that an optimal decision will be made if all of the necessary information (I) is available, and if this information is 100% reliable (I_{max}). In practice however, this is never the case (I ñ I_{max}). It is assumed that the probability of making a less than ideal decision will concomitantly increase with an increase in the uncertainty of the information from the monitoring network. This increase is proportional to the difference between I and Imax. And so the risk of making incorrect decisions is used to quantify the advantages of measuring and monitoring systems. The information that is generated by the measuring and monitoring system can thus be seen as a means to reduce the risk of failure (see figure 1).
The optimal monitoring network is found by comparing the monitoring costs inherent in a monitoring network layout which provides information with a certain degree of uncertainty, and the expected damage connected with the risk of failure associated with this unreliability. Thus, if the network is optimal, the costs of an optimal monitoring network are equal to the expected damage. A design cycle does not inevitably result in the optimal network. It is conceivable that the owner of the monitoring network (or others with a say in the network) will opt for a different monitoring network, perhaps due to financial or emotional reasons. In this case, the owner of the monitoring network is deliberately opting for a different risk (cf. Freeze et al., 1990).
In spite of the simplification described above, it is still difficult to establish the economic value of information, even within the concept of expected damage. The expected damage is almost always influenced by a set of variables and parameters in which groundwater data is often only one component.

FACTORS DETERMINING EXPECTED DAMAGE

The expected damage is essentially influenced by the following three factors: the groundwater applications, the water management task for which the information from the network is used, and the level of water management.

GROUNDWATER APPLICATIONS

We can distinguish between three major uses of the water system (Glasbergen et al., 1988): water as a means, water as a raw material and water as a substance to be discharged. Industries and property owners are particularly interested in water as a means. The expected damage of this use is related to land subsidence (structural damage to buildings, changing ground levels, and/or reduced load-bearing capacity) and to incorrect estimates of the storage capacity of an aquifer.
The main users of water as a raw material include companies which supply water to the industry sector and for domestic use, agriculture and the conservation of nature. The expected damage to suppliers of water to industries and for domestic use consists of a reduced yield from a groundwater source (or water with an "incorrect" composition) and an incorrect prediction of the drop in the water table (a greater extent of damage). In the case of agriculture, the expected damage can be expressed in terms of reduced crop yields. In the case of the conservation of nature, the expected damage comprises extra management or restoration work which is necessitated by undesirable ecological features.
Parties which use water as a substance to be discharged include the sewerage managers (discharges of waste water), industries (cooling-water disposals), and agriculture (discharge of excess water from ditches, drains etc.). In these cases, the expected damage is primarily damage to civil engineering works, poorly functioning, inadequate or oversized drainage infrastructure, reduced crop yields, undesirable vegetation (biotopes) and instability.

THE EFFECTS OF WATER MANAGEMENT TASKS AND THE LEVELS OF WATER MANAGEMENT

Van Bracht (1994) designed a conceptual model of integrated water management which describes water management tasks and flows of information. The following seven management tasks can be distinguished:

- making an inventory of the demands of the interest groups (task I)
- making an inventory of the social, economical and political motives (task P)
- establishing the potential uses of the water system (task U)
- assigning functions to the water system (task F)
- designing target scenarios (task S)
- designing management measures (task M)
- evaluating the water management (task E)

Information on the water system is required if one is to successfully implement tasks U, S, M and E. If the potential uses of the water system are established on the basis of inaccurate system information, this may generate expectations which are far from realistic - and even impossible. The effects can be considerably damaging: the target scenarios which are formulated during the implementation of task S may be unrealistic, and impracticable or less than ideal management measures may be devised or applied when implementing task M. The use of uncertain information in the implementation of task S may result in a situation in which the conditions which are chosen for the water system are incorrect with respect to the functions that are selected in the area in question. Once again, the result may be inappropriate or less than ideal management measures. The same negative results may be obtained if incorrect information is used for the implementation of task M. Finally, implementing task E on the basis of uncertain information may result in an inaccurate understanding of the water system, and so any approp-

riate corrective measures will not be applied on time. Clearly, therefore, the greatest reduction in expected damage can be achieved by minimising the uncertainty and the risks involved in the implementation of task U. Essentially, the smallest reduction in expected damage will be achieved by minimising the uncertainties associated with task M or E.

The expected damage associated with strategic water management that is the result of incorrect or less than ideal water management activities, will ultimately be considerably greater than the expected damage associated with operational water management. This is primarily caused by the "scale" effect. The water management plans which the water manager draws up, are put into operation by the operational water managers. In extreme cases, an impossible scenario for a certain region may result in wasted investments and/or damage to buildings, crops and/or nature.

EXAMPLE

In the following example, the design of a groundwater monitoring network in the Province of Friesland in the Netherlands is discussed in more detail. The groundwater level in this area is naturally shallow and can be managed using an intensive surface water system. The composition of the soil is a mixture of humus clay and peat. The main uses of the land are nature reserves, grassland and residential areas. A groundwater level monitoring system is to be developed for this area for the benefit of the regional water manager. This monitoring system will be used for the implementation of task E (evaluating the water management) of the water management model referred to above. In order to realise task E, it will be necessary to test the actual groundwater level against the chosen values in the target scenarios. Any deviations from these values should urge the water manager to aim for a more effective water management. The need for information is focused on the determination of the seasonal mean. The mean spring groundwater level is the most significant. The temporal seasonal mean is established by measuring the groundwater level at fixed intervals. The spatial seasonal mean is determined using the measurements which were recorded in a number of wells that were taken from a stratified random sample. The information content of the monitoring network can be determined by using the standard deviation of the spatial and temporal seasonal mean. The spatial and temporal statistical properties of the area are derived from historical measurement data. Basic statistics are suitable to determine the relationship between the standard deviation of the temporal and spatial seasonal mean and the measuring frequency and network density of the monitoring network (Sanders et al., 1983). The monitoring costs primarily depend upon measuring frequency and network density. An investigation into the past can determine the relationship between the groundwater level and the specific yields for various groundwater use (for example, Workgroup Help table, 1987, and Witte, 1990). These specific yield functions for crop yield (dairy farming), ecological values (moist herbal vegetation) and stability (subsidence) are shown in Figure 2.

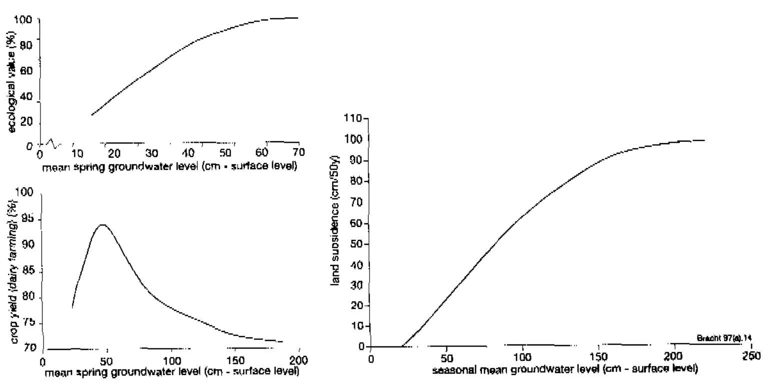

Figure 2 Relationship between mean spring groundwater level and crop yield (%), ecological values (5) and land subsidence (cm)

It is possible to calculate the relation between the economic value (in Dutch guilders) and the mean spring groundwater level using standard yield data for agricultural crops. The damage related to subsidence in residential areas is equal to the costs which must be made to compensate for the subsidence for each groundwater level (for example, costs involved in raising the ground level). The "restoration method" (Van Bracht, in preparation) offers a solution when dealing with natural values. This method assumes that, following a disruption, natural values are always restored. The method can be used, for any given structural change in the groundwater level, to determine how, and at which cost, the natural values can be restored. In extreme cases where restoration of the original state is no longer an option, the restoration costs imply the investment which is needed to obtain a similar area and transform it into a nature reserve.

A mean spring groundwater level of 40 cm below ground level has been designated in the target scenarios for agricultural and natural areas. A level of 80 cm below ground level has been chosen for residential areas. If, based on the data from the groundwater monitoring system, the actual mean spring groundwater level is set at 40 cm and 80 cm below ground level, there is a certain probability (risk) that the groundwater will exceed or fall short of this level. This risk is proportional to the uncertainty (standard deviation) of the seasonal mean. The expected damage can be determined by using basic statistics, the statistical properties of groundwater level in the area and the yield functions in Figure 2. The relationship between the monitoring costs and the expected damage for different values of the standard deviation of the mean spring groundwater level is shown in Figure 3. It also displays the economical optimum of the groundwater monitoring network in the research area. It depicts, in an objective way, the performance of a monitoring network.

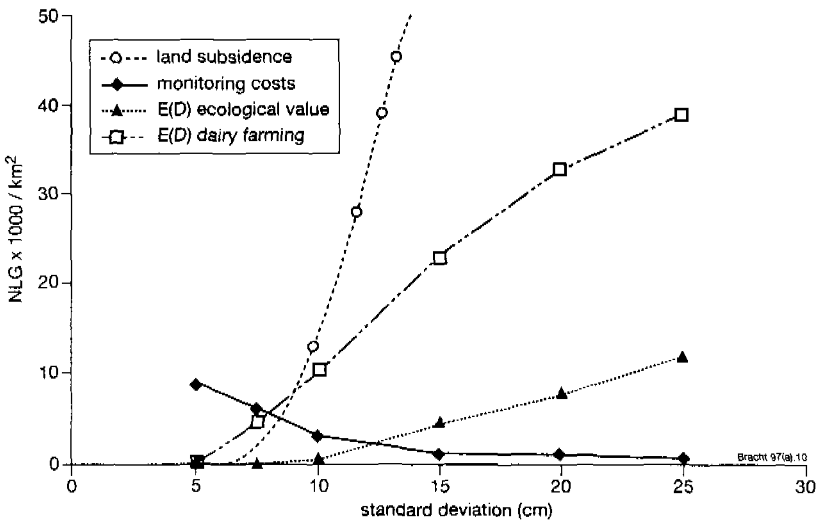

Figure 3 Relationship between the standard deviation of the mean spring groundwater level, the monitoring costs and expected damage E(D).

DISCUSSION

As Figure 3 reveals, the concept of expected damage offers obvious advantages. A different optimum applies to each form of groundwater use. Therefore, the optimal monitoring effort will vary with the different land uses. If the land uses change in the course of time, it will be necessary to adapt the monitoring effort. An important aspect of the concept of expected damage is the assumption that knowledge of the groundwater level will be used to prevent any negative effects. In reality, this is not always possible. The concept of expected damage is applicable where the before-mentioned yield functions are known. These functions will however vary, depending upon the area in question. Obviously, the expected damage will be more difficult to quantify in complex situations, for example if more variables apply. All in all, it can be argued that the concept of expected damage can offer an objective quantification of the economic optimum of a monitoring system.

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RESULTS OF A TREND ASSESSMENT OF NEW ZEALAND'S NATIONAL RIVER WATER QUALITY NETWORK

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ABSTRACT

Five years of monitoring data from New Zealand's National Rivers Water Quality Network have been analysed for trend using nonparametric methods that account for seasonality and flow-dependence. The full set of results is contained in three 77x30 matrices which do not simply allow for "quick" recognition of significant trends. These matrices contain thirty columns being site identifiers, flow, and results with and without flow-dependence removed (there is interest in both cases). The three matrices are for absolute and relative trend slope estimates and also for attained significance levels (i.e. p-values). Because these matrices do not simply convey the results, we have instead used a series of maps with arrows depicting time-trends at the various sites. Calculation of the (binomial) probability of a map's pattern of results being attributable merely to chance has permitted rather general statements to be made about overall spatial as well as temporal trends. In particular there was a general improvement in water quality over the five-year period, particularly in the South Island - a heartening result. Water temperatures fell significantly during the five years. It is hoped that the form of presentation of results, using maps and general statements, will be more comprehensible to the public, and will serve to highlight some research needs.

INTRODUCTION

New Zealand's water resources management relies primarily on the operation of seventeen regional authorities (mostly elected Regional Councils). Nevertheless, central government has retained an interest in monitoring the state of the nation's environment, most particularly given explicit requirements in the Resource Management Act 1991. An important goal of water quality monitoring is trend analysis, driven by the desire to know whether the increased effort in waste and land-use management is having a beneficial effect. For example, there have been many upgrades of waste treatment plants and an emphasis on minimising direct runoff of dairy farm wastes. The need for such trend information was foreseen some years ago, before the present Act was passed, and so the National Rivers Water Quality Network (NRWQN) was initiated with central government funding in 1989. It was funded, *inter alia,* on the basis of being able, after five years' operation, to regularly detect any "significant trends" (equal to the standard deviation of the detrended and deseasonalised data). Herein we report on features of development and presentation of a trend analysis for the first five years' of data.

NATIONAL RIVERS WATER QUALITY NETWORK

The NRWQN commenced operation at the beginning of 1989, following an agreed design (Smith et al., 1989; Ward et al., 1990; Smith and Mc Bride, 1990; McBride and Smith, 1996). Monthly measurements are made of flow and fourteen physical/chemical determinands at seventy-seven river sites distributed throughout the North and South Islands (total area =

250,000 km^2). (Biological variables are measured also.) Water quality variables measured reflect the predominantly agricultural base of the country's economy.
Sampling is as close as possible to the same time-of-day at each site and is performed by NIWA's fourteen national hydrometric field teams. Laboratory analyses are performed in NIWA's Hamilton laboratory. Data quality assurance procedures for field and laboratory data (Smith and McBride, 1990) have been adhered to rigorously. Data are stored using specially constructed archiving software, designed to minimise ambiguity in storage and retrieval (Ward et al., 1990). Annual reports (seven to date) have been produced.
The Network design nominated two objectives: (a) to develop better understandings of the nature of the water resources and hence to better assist their management, (b) to detect significant trends in water quality. In respect of the first objective, the first two years' data have been used to characterise the physico-chemical nature of the river waters and examine the relationships between this and environmental factors (Smith and Maasdam, 1994; Maasdam and Smith, 1994). More recently, a trend analysis based on the first five years' data has been completed (Smith et al., 1996b). This is the first time such a comprehensive water quality trend analysis has been reported in New Zealand, so there is considerable interest in its results, both from water management agencies and from the public.

TREND ANALYSIS METHODS

For these data, trend analyses using ordinary linear regression versus time is not appropriate because the required assumption of normally distributed residuals (Shapiro-Wilk and skewness tests) is often violated (Smith and Maasdam, 1994). Also, ordinary regression may fail to account adequately for strong seasonal components of variability, so that statistical power to detect trends can be greatly diminished. Instead, the nonparametric "seasonal Kendall Sen slope estimator" has been used to assess the magnitude of any trend, accompanied by the seasonal Kendall trend test to assess its statistical significance (Helsel and Hirsch, 1992). We have used the Wqstat II package (Colorado State University, 1989).

SEASONAL KENDALL METHODS

The main feature of these methods is that they account for seasonality by considering only within-month slopes. One does not consider the slope between, for example, data for January and October (as is done in ordinary regression). All possible within-month slopes are used. The methods are nonparametric and so do not require assumptions of normality of populations.
The seasonal Kendall Sen slope estimator is the median of all possible combinations of within-month slopes. The seasonal Kendall test can be viewed as a nonparametric test for zero slope of a linear trend (Gilbert, 1987). It poses the null hypothesis that data are independent of time and season; the alternative hypothesis being that for one or more seasons there is an upward or downward trend (a two-sided test). A strong advantage of this test over parametric alternatives is that one does not have to make any assumption, apart from monotonicity, about the functional form of any trend that may be present (e.g. linear, exponential, etc.); the test merely contemplates whether within-month/between-year differences tend to be monotonic.
Because the sample size is the same for all tests for the first five years' Network data the *p*-value of the seasonal Kendall trend test serves as an excellent index to compare the strength of trends between sites and between determinands. It would not do so if sample sizes varied (McBride et al., 1993). The mechanics of the test are given elsewhere (McBride and Loftis, 1994).

FLOW-ADJUSTMENT

Another known source of variability is the dependence of some determinands on flow, so trend analyses have all been performed both with and without flow-adjustment. The former results indicate whether there is some underlying cause of trend other than rainfall-flow variability; the latter indicate the trend that is observable.

Two possible types of variation with flow occur in our data, using the terminology of Helsel & Hirsch (Helsel and Hirsch, 1992): *dilution* (where a determinand, e.g. conductivity, is diluted); or *wash-off* where increased flow can increase a determinand's concentration by entrainment, e.g. total phosphorus). We used LOWESS smoothing (LOcally Weighted regression Scatterplot Smoothing: Cleveland, 1979) -an automated computationally-intensive regression that tracks the central tendency of the data in such a way that one does not have to specify the functional form of the regression. It is therefore robust (Lettenmaier et al., 1991).

Trend analysis on flow-adjusted values was performed in a two-step procedure. For each determinand and site, raw data were plotted against flow and LOWESS smoothed using Data Desk™ (Velleman, 1992) with a span of 30% (span is the percent of data used in each locally weighted regression; 30% is a good general compromise between under- and over-smoothing for the Network data). The trend analysis was then performed on the adjusted residuals, where an adjusted residual for a sampling occasion is defined as: raw datum - smoothed datum for the flow on that occasion + median of all sixty data. Because this analysis removes the background variability with flow, trends tend to show up more clearly.

For an example of the effects of flow-adjustment, see Fig. 1. Figure 1a indicates a statistically significant upward trend in conductivity, notwithstanding that to the naked eye the data may appear too "noisy" to allow such a conclusion, while Fig. 1c shows that with the flow-conductivity relationship taken into account there has in fact been little change in conductivity, suggesting that inputs to the river of conducting material have remained constant over the period examined.

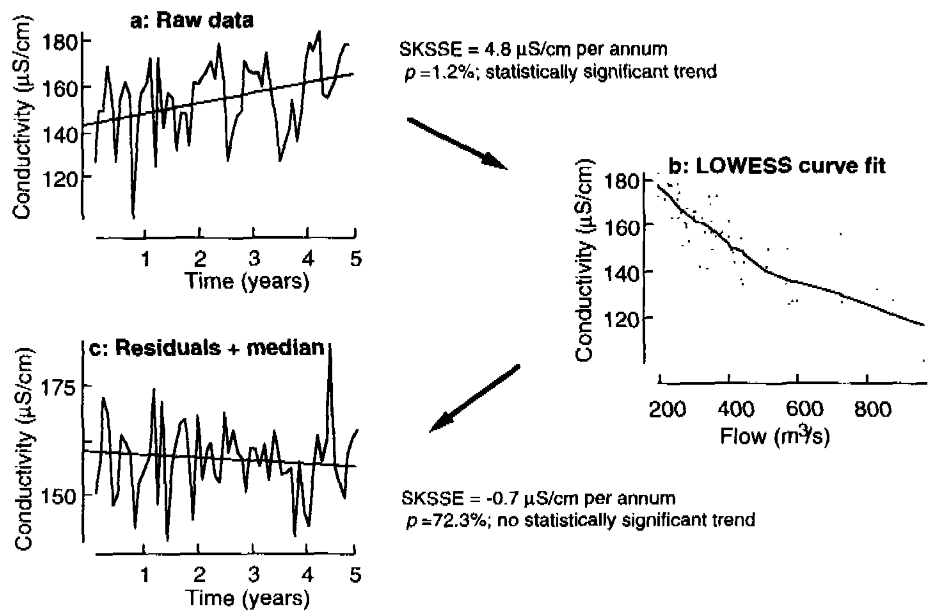


Figure 1: Example of flow-adjustment (conductivity in the Waikato River at Site HM4): (a) time plot of raw data; (b) LOWESS-smoothed data against river flow; (c) time plot of residuals from (b) plus median of raw values. Flow trended downward over the five year period. The line in (a) and (c) is the seasonal Kendall Sen slope estimator (SKSSE). The *p*-values are derived from the seasonal Kendall test.

PRESENTATION OF TRENDS OVER TIME

The full array of results for the seventy-seven sites, with and without flow-adjustment, is contained in three 77x30 matrices. The thirty columns contain: site identifiers, flow, and results with and without flow-dependence removed. The three matrices are for absolute and relative trend slope estimates and also attained significance levels (i.e. *p*-values) for the seasonal Kendall

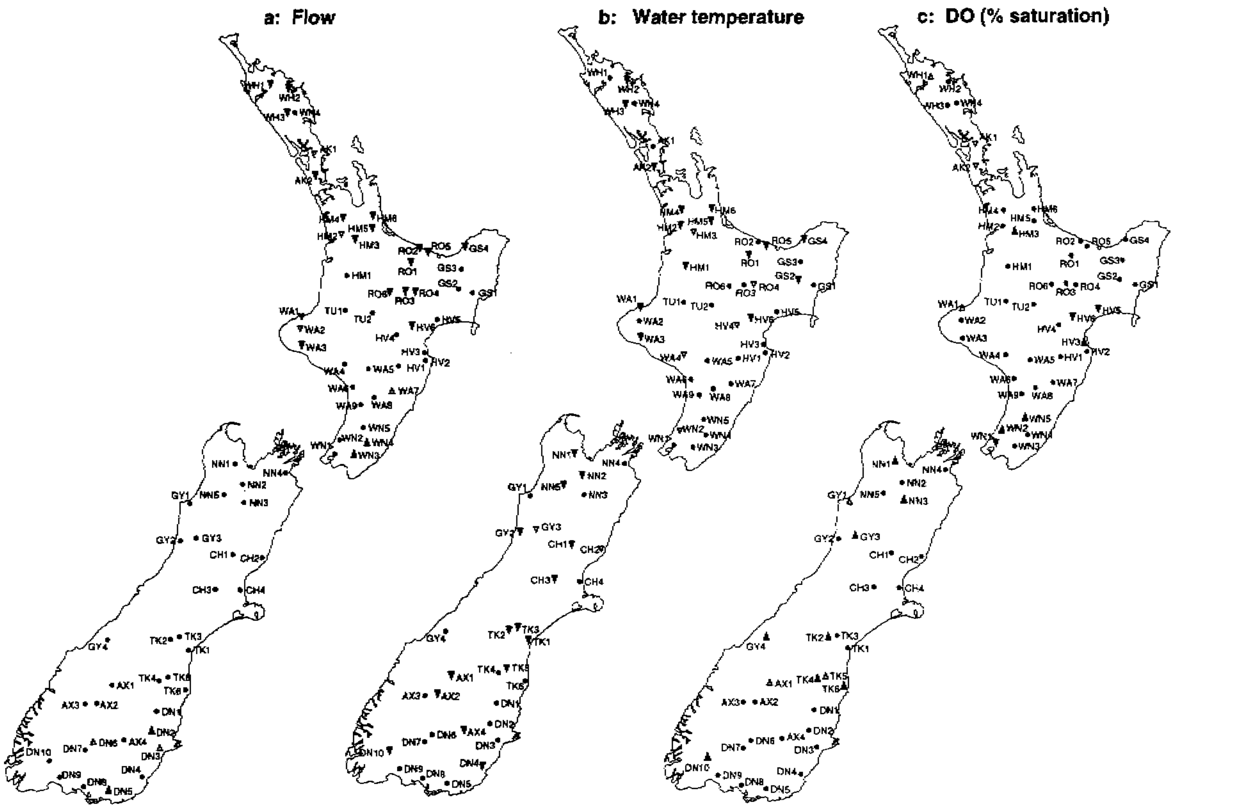
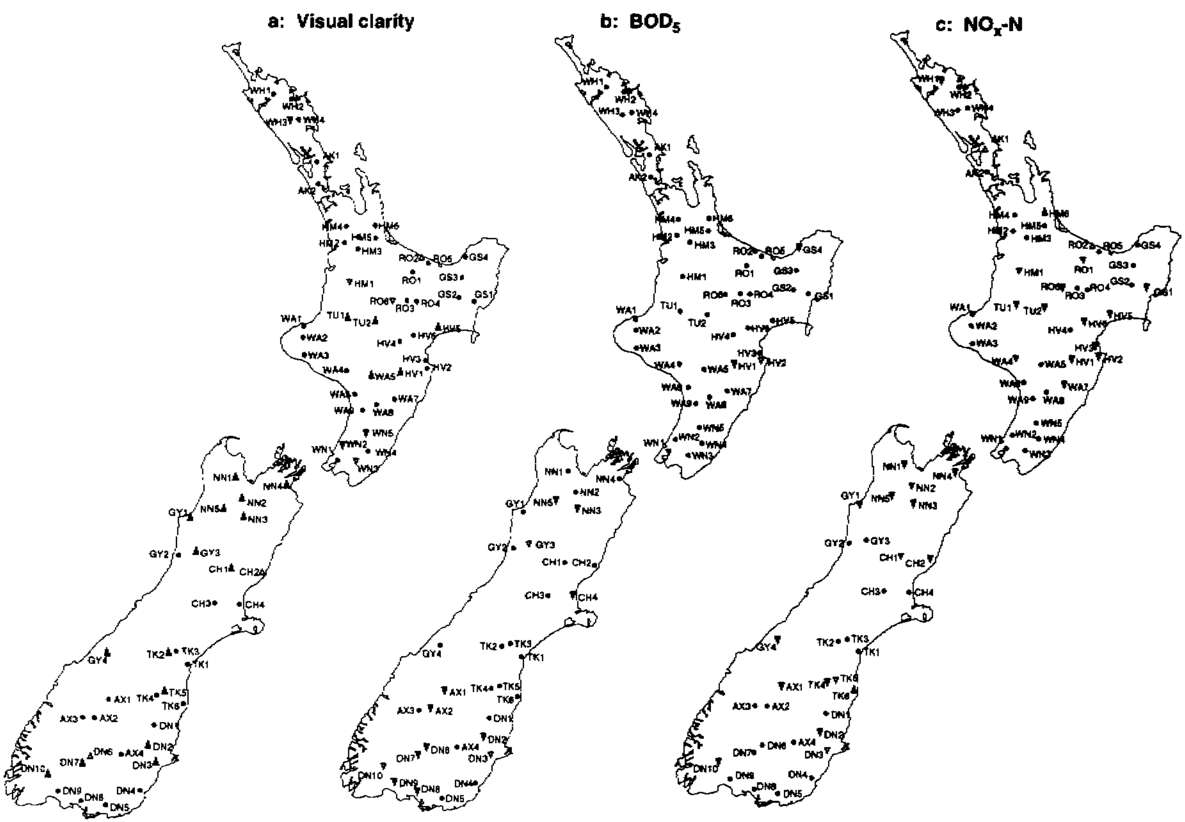


Figure 2: Trend results for flow (a), water temperature (b) and DO for all 77 sites over the North and South Islands (total area = 250,000 km²).

Notes. The first two letters of the site identifier indicates the location of the data collection team - e.g., WN2 is the second site in the Wellington team's list. Trend codes are: (▲ = upward trend, $p < 5\%$; (Δ = upward trend, $5\% \leq p < 10\%$; ● = no trend, $p \geq 10\%$; ▽ = downward trend, $5\% \leq p < 10\%$; ▼ = downward trend, $p < 5\%$.



Figuur 3: Trend results for visual clarity (a), BOD_5 (b) and oxidised nitrogen (c) for all 77 sites. For explanations of conventions used see the note appended to the caption of Fig. 2.

trend test. Because these tables cannot be used to simply convey the results, we have instead used a series of maps with arrows depicting trends (Smith et al., 1987; Lettenmaier et al., 1991). We also test the hypothesis that the observed numbers of upward and downward trends for each constituent over the seventy-seven sites could have arisen merely by chance. Rejection of this hypothesis permits the inference that a trend occurred over regions (the whole country, or each of the North and South Islands separately), as well as over time.
The following conventions have been adopted:

1 statistical significance is inferred when the p-value of a test is less than 0.10;
2 "trend" in a determinand at a site denotes a value of a site's seasonal Kendall Sen slope estimate for which a statistically significant seasonal Kendall trend test has been obtained: absolute trends are expressed as determinand units per annum, relative trend values are expressed as percentages per annum. The distribution of site trends i is displayed on a series of maps of both Islands, each showing all seventy-seven sites: sites with trends are identified by upward or downward arrows, sites with no trend are indicated by dots (following the USGS approach (Smith et al., 1987)).
3 "national trend" in a determinand is the median of all site slope estimates. It is reported only for determinands where the obtained proportion of upward or downward site trends are found to be statistically significant (at the 10% level, using binomial probabilities). Trends over each of the North and South Islands are computed similarly.

INDICATIVE RESULTS

Maps for selected determinands showing significant trends over the five-year period, for flow-adjusted data only (except for flow, Fig 2a), are shown on Figs. 2-3.
Figure 2a shows that nearly all sites in the northern half of the North Island exhibited downward trends in flow. There was a small downward national trend (absolute and relative trends of -0.094 $m^3 s^{-1}$ per annum and 0.8% per annum). A predominance of South Island sites had positive slope estimates, though few were s ignificant. As a result one may expect differences in trend estimates with and without flow-adjustment (as was found for pH, visual clarity, conductivity, ammonium, total nitrogen, dissolved reactive phosphorus and turbidity).
Figure 2b shows that all thirty-seven flow-adjusted water temperature trends were downward (forty-one without flow-adjustment), and these were distributed evenly over the whole country (national trend = -0.28°C per annum). This corresponds to a loss of over 1°C over the five-year period. Only fourteen slope estimates for temperature were positive, none of which was significant.
Figure 2c shows a significant increase in DO (as percent saturation) over the period in the South Island, giving a relative national relative trend of +0.2% per annum. This is coincident with downward trends in biochemical oxygen demand (Fig. 3b), a national relative trend of -4.7% per annum.
Figure 3a shows upwards trends in visual clarity: all seventeen trends in the South Island were upward, but the twelve in the North Island were split evenly between upward and downward trends. The national trend was 0.023 m per annum (2.8% per annum), representing an improvement in water clarity over the five-year period of more than 10%.
Figure 3c shows a downward national trend of -2.6 mg NO_x-N m^{-3} per annum (4.4% per annum); only three of the thirty-two trends were upward. Both Islands showed downward trends.
Other results, not shown here, include: upward national trends for conductivity, and absorption coefficients (indexes of dissolved colour); and downward national trends for ammonium, dissolved reactive phosphorus and total phosphorus. Trends tended to be stronger in the South Island. No trends were found for pH or total nitrogen.

DISCUSSION AND CONCLUSION

We have carefully maintained our sampling and physico-chemical analytical capabilities over the study period and we conclude that our methods have not influenced trend detection (laboratory methods have not changed).

The results of the trend analysis contain a surprise for many people: the large *downward* trend in water temperature. This is related to the cooling of the New Zealand region in 1991B93, probably due to the combined effects of the Mt. Pinatubo eruption and the 1991-94 *El Niño* (J. Salinger, NIWA). Other results (e.g. decreasing nitrate and BOD_5 concentrations) suggest that increasing efforts in pollution control and waste management are having the desirable effect of improving water quality, possibly aided by decrease in the national sheep and beef stock and fertiliser use. T his is a matter that should now be investigated. Other results, e.g. increasing conductivity and absorption coefficients along with increasing clarity, are fostering further scientific interest in the optical quality of New Zealand waters (Smith et al., 1996b).
In order to present intelligible and defensible findings, we have sought to minimise the use of tabular forms of such results, using instead maps and generalised statements regarding national trends. These statements refer to the combination of time-trend at sites *and* their spatial distribution, on which little has been published to date. Refinement of methods for inferring temporal and spatial trends are clearly called for. As data records become longer this will also have to incorporate reversing/cyclic trends between years and/or seasons. Meanwhile we hope that approach taken to date will serve to inform water management agencies and other interested parties on environmental trends and the importance of ongoing monitoring.

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THE SERCON PROJECT

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ABSTRACT

Methods used at present in the UK for assessing the conservation value of rivers lack a degree of rigour and repeatability. SERCON is essentially an evaluation technique designed to improve these characteristics. River attributes are grouped within six generic conservation criteria - Physical Diversity, Naturalness, Representativeness, Rarity, Species Richness, and Special Features. A conservation index is derived foe each criterion, an Impacts indes is calculated from an additional suite of attributes, and background data are added to provide the contect for each evaluation. The system has been designed for use principally on computer, altough users can refer to a printed manual instead.

THE DEVELOPMENT, STRUCTURE AND FUNCTION OF SERCON

Information on river conservation value is needed for several reasons.

- Some organisations (such as the Environment Agency in the UK) have a statutory conservation duty and need to heve regard for the conservation status of rivers within their jurisdiction.

- External pressures, in the form of EC Directives (e.g. Habitats Directive, proposed Ecological Quality of Water Directive) mean that Member States will need to address the issue of how rivers and their corridors can be monitored, not just from the perspective of water quality but in broader environmental terms.

- Where catchment developments are proposed, there may be a need to describe the relative values of each watercourse and thus allow a decision to be made on the most appropriate location.

- River restoration projects are now becoming increasingly common in Europe and elsewhere. Methods are needed for defining the present status of a degraded river, and later for comparing the success or otherwise of restoration works.

SERCON ('System for Evaluating Rivers for Conservation'), a new technique for assessing the conservation status of rivers and their corridors, has been designed to meet these needs. Its aim is to widen the scope and increase the rigour and repeatability of ecisting evaluation techniques, and is intended for all those with an interest in river conservation and management.

From the outset of the profect, the following adventages of developing SERCON were identified:
1. greater uniformity in data collection and objectivity in evaluation
2. identification of gaps in the scientific knowledge of specific rivers
3. the assessment of rivers within a wide range of environmental quality
4. a simple way of communicating technical information to planners and policy-makers
5. an appraisal of the rehabilitation potential of degraded rivers

6. a tool in environmental impact assessment
7. a framework for extending SERCON to include other environmental attributes

SERCON is essentially a technique for scoring a range of riverine features, using a comprehensive library of reference data. It has been designed principally for use on a PC, but it can also be used with the aid of a paper manual.

At the start of the project, a list of potential riverine attributes was compiled each of which was considered to be important in the process of river condervation evaluation. These included a wide range of fluvial features and species groups present in rivers in the UK, the characteristics of the catchment, and the potential impacts to which river systems may be subjected. Rather than assess each attribute isolation, SERCON is structured so that each one (sometimes in various forms) is used to build up a picture of the river, primarily in terms of accepted conservation criteria such as Naturalness, Representativeness, and Rarity.

In addition to attributes used for evaluation, other data on the river's physical features are recorded (e.g. channel gradient, stream flow stability) so that each assessment can be placed in a geographical or geological context. Information is also recorded on catchment naturalness (based on land use and human population density) and water quality, and a separate part of SERCON is used to evaluate aquatic impacts.

The main core of final output is a group of vonservation indices, one for each of the six main criteria, derived from infoemation on each attribute. An index can range from 0-100, with 0 representing no conservation value, and 100 the highest value that can be assigned from the information available. A seventh category (Additional Features of Importance) allows the user to draw attention to unique or unusual features of the river, but it does not contribute in any way to the derivation of conservation indices. The same is true for assessment of catchment naturalness and water quality. The scores for aquatic impacts (and the total impact index) are used to highlight problem areas within the catchment, but are not used to downgrade conservation indices.

The PC version of SERCON operates essentially as an aid ato the user in completing the calculation of indices for conservation criteria and for Impacts. However, this relatively simple calculating function is backed by the full body of explanatory text in hypertext format, supported by on-screen diagrams and colour photographs of certain riverine and riparian features,. In addition, attribute scores (not raw data) can be stored in a databese from which they can be recalled for modification, or for comparisons between rivers. Presentation of scores and indices is echanced by a series of tect and graphical reports which can be displayed on screen or as hard copy.

SERCON is implemented on PC as a Microsoft Windows™ application that presents a familiar interface to most users. Experience suggests that learning to use SERCON takes only a few minutes for an occasional Windows user who has seen a demonstration. Data are stored in Microsoft Access™ files which means they can be accessed and queried from other software, if desired. Graphical output can be pasted into the user's own documents, thus enhancing the output possibilities. To reduce teh disk storage requirements, photographs in SERCON have been compressed using fractal compression software so that they take up a relatively insignificant amount of space.

CONCLUSION

SERCON has undergone a series of tests durong its develoopment and immediately afterwords. It has also been described and demonstrated at a wide range of venues in the UK and abroad, and has formed the besis for an evaluation system recently designed for fresh waters in Sweden. The responce to SERCON so far has been encouraging, and it is felt that the objectives of the project have been met. Nevertheless, there is still a need for more rigorous testing of the system, and refinements will undoubtedly be needed for Version 2 of SERCON. Apart from its use as an evaluation method, it is becoming clear that many see it as a useful educational tool (especially the PC version), illustrating many of the important features of rivers and explaining why they are important for conservation.

THE AUSTRIAN WATER QUALITY MONITORING SYSTEM - INFORMATION FOR DIFFERENT LEVELS OF THE DECISION MAKING PROCESSES

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ABSTRACT

As a basis for decision making processes key figures characterizing water quality are needed on each administrative level. The higher the administrative level, the higher is the degree of data aggregation. Therefore, there is a strong need for reliable and comparable key figures. The Austrian Water Quality Monitoring System provides an excellent set of basic groundwater quality data - gained by standardized procedures for sampling, analyzing and data handling. This data base is used to study possible effects of data aggregation on different levels. Results indicate clearly that the values of resulting key figures depend on the applied aggregation strategies. Some examples are given to illustrate this.

INFORMATION NEEDS FOR DECISION MAKING:

Decision making processes in modern water management require information on different scales ranging from local surveys to country-wide monitoring programs and international levels. In general, the larger the geographic unit affected by decisions the higher will be the degree of consolidating the information needed. Detailed information obtained by investigations on local scales provide a basis for decisions on different levels and will be used either as local information or in form of complex indicators, indices or key figures which are able to characterize hydro(geo)logical or administrative units (consolidated data). If consolidated data are used, the processes of consolidation from one level to another level has to be as transparent as possible. To decide whether information can be consolidated and whether different consolidated information can be compared it is necessary to know the main features of the primary data (quality of the data and the criteria of their collection).

THE AUSTRIAN WATER QUALITY MONITORING SYSTEM (AWQMS):

Since 1990 new legislation and administrative procedures regarding water pollution control in Austria have formed the basis for a new AWQMS for both ground water and running waters. Federal and provincial authorities as well as a large number of private laboratories are involved. Groundwater samples are collected four times a year at about 2,000 sites. River water samples are collected six times a year at about 245 sites, sediments as well as biota are sampled once a year. At some river sampling sites samples are taken twelve times a year due to bilateral agreements on transboundary water management issues. At each sampling site and sampling date about 60 parameters are measured comprising parameters for a general hydrochemical characterization of the water (e.g. nutrients, TOC, DOC), organic micropollutants (e.g. pesticides) and heavy metals. Special attention is drawn to standardization and analytical quality assurance.

Taking into account the targets of the AWQMS, the design of the monitoring network (e.g. location of the sampling sites) and the administrative framework the AWQMS integrates elements of background, impact, trend and compliance monitoring (Chovanec & Winkler 1994, Chovanec et al. 1996, Schwaiger et al. 1994, Vogel 1995, WWK/UBA 1993 and 1995).

INFORMATION ON DIFFERENT SCALES:

All information obtained by the AWQMS is based on the investigation of single sampling sites; the number and location of these sites are chosen in order to characterize the water quality of the hydro(geo)logical units of Austria (groundwater areas, river catchments). Those water quality data are used either on their own or - if necessary - in combination with additional environmental data (e.g. land use, contaminated sites, waste water effluents) for the purpose of water quality management. Maps (figs 1-3) showing the important ground water areas in porous media, the network of monitoring sites in porous media in the area close to Vienna and the monitoring sites in karst water.

The Austria Water Quality Monitoring System (AWQMS)

Investigated Groundwater Areas in Porous Media

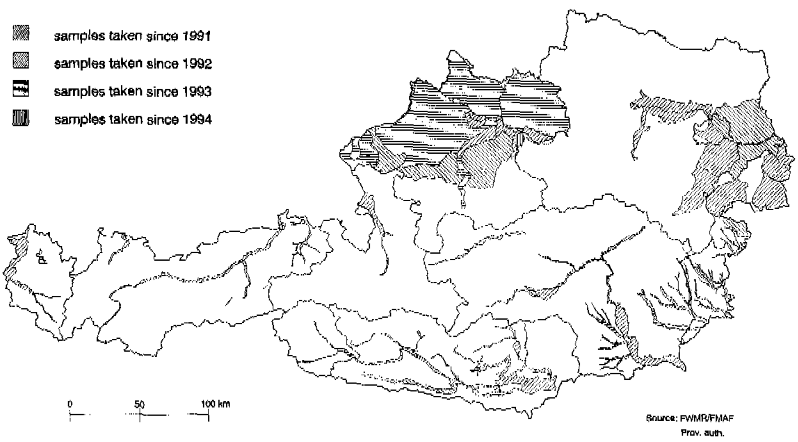


Figure 1: Investigated groundwater in porous media.

The AWQMS is able to provide reliable information for local, regional, provincial and national environment management decisions because of standardization and analytical quality assurance according to the water management needs of Austria. Decisions on the local or regional scale (concerning e.g. screening for the detection of contaminated sites, urban and landscape planning, impacts of municipal/industrial emissions and diffuse sources) mostly require appropriate information on single sites. For provincial and federal decision finding processes or legislation (in Austria e.g. enforcement of the Environment Control Act, Federal Water Act, Environmental Information Act, Pesticide Act, Chemical Substances Act) more consolidated information - in terms of data characterizing hydro(geo)logical units or other key figures - is essential.

Due to the fact that many environmental problems cannot be solved on a national level, the availability of comparable consolidated information on an international scale is becoming more and more important e.g. for transboundary river basin management (Rhine, Danube,...), for the environmental management within the European Community and - on a global scale - for the follow-up of the UNCED process as the implementation of the AGENDA 21 (UNCED 1992).

For international purposes sound consolidated information can be extracted from the AWQMS depending on the requirements of different issues that have to be met (e.g. Strategic Action Plan

Detail Network Design - Selected Groundwater Areas in Porous Media

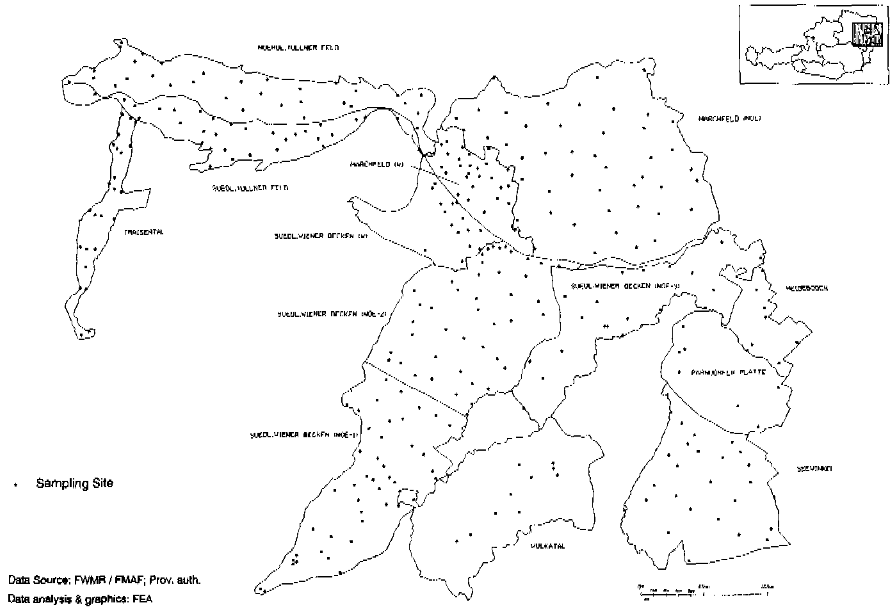


Figure 2: Detailed network design - Selected groundwater areas in porous media.

Detail Network Design in Karst and Crevice Groundwater

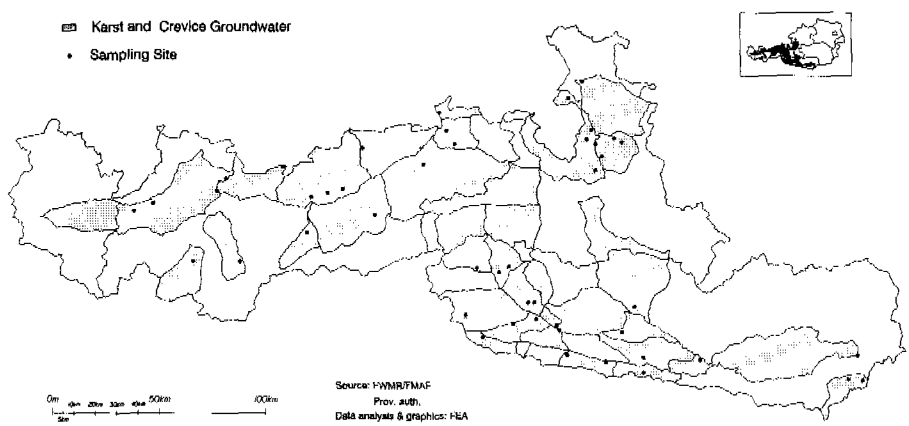


Figure 3: Detail network design in karst and crevic groundwater.

for the River Danube Basin, information for the European Environment Agency). However, if consolidated data - indicators, indices or key figures - of different administrative or geographical units are used and compared it is crucial to ensure that the targets, strategies, design and methods of the different national monitoring programs allow this comparison.

ASPECTS CONCERNING THE AGGREGATION OF DATA - VARIABILITY OF RESULTS

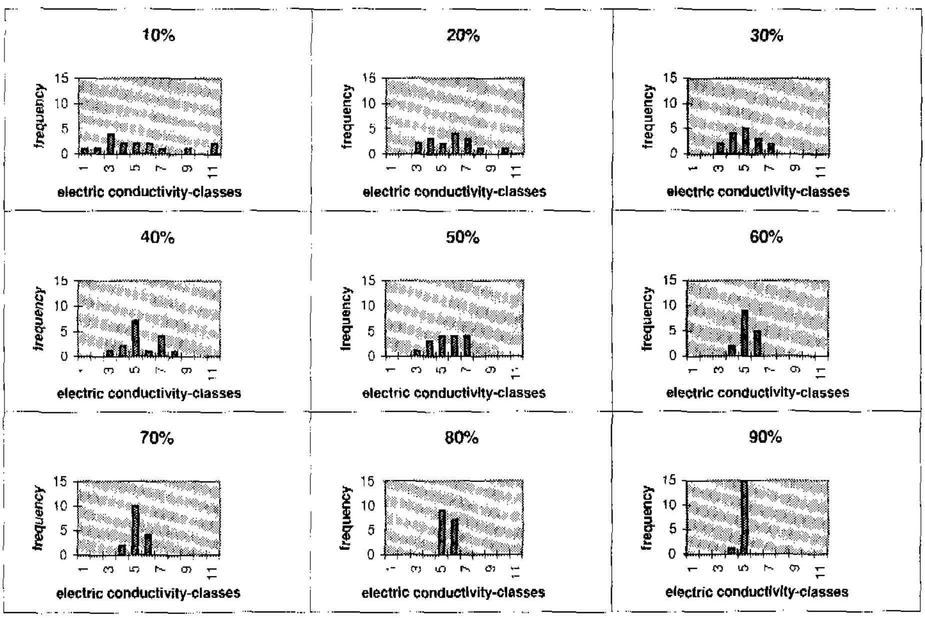
The following examples taken from the groundwater program illustrate some problems arising from the aggregation of data

DENSITY OF SAMPLING SITES:

To estimate the groundwater-quality of a larger area a sufficient number of sampling sites is necessary. Quality-data from a single groundwater sampling site provide correct information for this site (stratum and time). Whenever this information is used to extrapolate it to another spot - even if it is a very close one - errors might occur. The adequate number of sampling-sites needed for the characterization of a certain area depends on the hydro(geo)logical situation as well as on the contamination sources (no contamination, point sources, diffuse sources). If the density is insufficient the calculation of an average quality may lead to wrong results.

Figure 4 shows the reduction of the variability of the arithmetical mean in correlation to the density of sampling sites in a groundwater area close to Vienna (a highly structured area that is used for agriculture, industrial activities, settlement etc.). The basic network comprises 85 sampling-sites covering the 1014 square km large area in a regular pattern (density: 1 site per 12 km^2). 10% of the sampling sites have been randomly chosen 16 times and the arithmetical mean of the electric conductivity (used as a very general indicator for water quality) for each run

Reduction of variability of arithmetical mean characterizing electrical conductivity in ground water areas *according to the number of randomly selected sampling sites*



Electrical conductivity (µS/cm): 1=600-650 4=750-800 7=900-950 10=1050-1100 2=650-700 5=800-850 8=950-1000 11=1100-1150 3=700-750 6=850-900 9=1000-1050

GW-area: Suedl. Wiener Becken
total number of sampling sites: 85

Figure 4: Effect of density of sampling sites on the variability of "average values".

were classified within frequency histograms. The same procedure has been applied for 20, 30 etc. up to 90% as well. The higher the number of sampling sites, the smaller the variability of the means. Whereas the very small variability of the higher percentages is strongly influenced by the fact that a higher number of identical sampling sites is chosen in different random samples the smaller percentages with a variation from 700-750 up to 1100-1150 (10%) or up to 1000-1050 (20%) clearly indicate the possible error caused by the insufficient sampling site density. Therefore, to characterize a groundwater aquifer a certain density is necessary to obtain reliable results. In this highly structured area we consider the density needed to obtain reliable figures characterizing the aquifer as a whole to be about one sampling site per 20 to 25 square kilometers. The density chosen for the national monitoring (AWQMS) is about one site per 10 to 15 square kilometers. This density has been chosen considering the information needed for regional decision making as laid down in the Water Act and some related ordinances. This network design should ensure a reliable general description of the quality of the aquifer and the regional quality pattern providing a basis for remediation measures. Contaminations restricted to small areas (e.g. caused by leaking septic tanks) cannot be reliably detected. An empirical basis for the development of this network design has been established by the evaluation of a local monitoring program and of a pilot study (Grath et al. 1992). Results of the local monitoring program and the AWQMS (two independent networks with comparable density in the same area) have shown comparable results (arithmetical mean of nitrate about 50 mg/l WWK/UBA 1993, 1995; Kaupa et al. 1988).

AVERAGE CONCENTRATION MAINLY DEPEND ON THE STRATEGY APPLIED - THE PROBLEM OF MAKING SUBSAMPLES

In most groundwater areas sites have different groundwater quality. If the area is divided (e.g. in background and impact areas) average values for the subsamples considerably vary according to the selection criterion applied. This effect is shown in figure 5 where a number of samples is ranked according the mean nitrate values of single sampling sites. Three different aggregation criteria (all in terms of quality) are applied.
The same problem occurs if not sampling sites but groundwater aquifers are grouped according to criteria which cannot be defined exactly.

Therefore the "splitting" of groundwater areas has to be based on very clear criteria and has to be done very carefully. The procedure has to be described. Whenever those data are aggregated on a higher level and the additional information is lost - misinterpretation may occur.

The next example stresses the same point on a high aggregation level. The AWQMS covers about 2000 groundwater sampling sites - most of them situated in porous media, only 135 are situated in karst areas. Groundwater in porous media can be found in flat valleys along the main rivers in Austria and in the low-lying areas in the eastern parts. These areas are densely populated and intensively used for agricultural activities. Karst groundwater resources are mostly situated in alpine regions with moderate landuse (density of sampling sites can be low). Mean nitrate concentration for groundwater in porous media in Austria is 28 mg/l (arithmetical mean of 1700 sampling sites), mean nitrate concentration in karst areas is about 3 mg/l (arithmetical mean of 135 sampling sites). These figures clearly indicate that groundwater in porous media and in karst areas are not similar.

Groundwater in porous media and in karst areas are equally used for drinking water purposes. Therefore, if the results are weighted according to this fact (1:1) a value of about 16 mg/l (the mean between groundwater in porous media and karst groundwater) could be considered as appropriate. If the arithmetical mean of all sampling sites is calculated (which could be considered as the best way in terms of comparability), the result is 26.5 mg/l. All figures vary significantly, but none of these sufficiently reflect the situation. Additionally, some aquifers in porous media show an average nitrate concentration from minus 5 to above 50 mg/l , and these

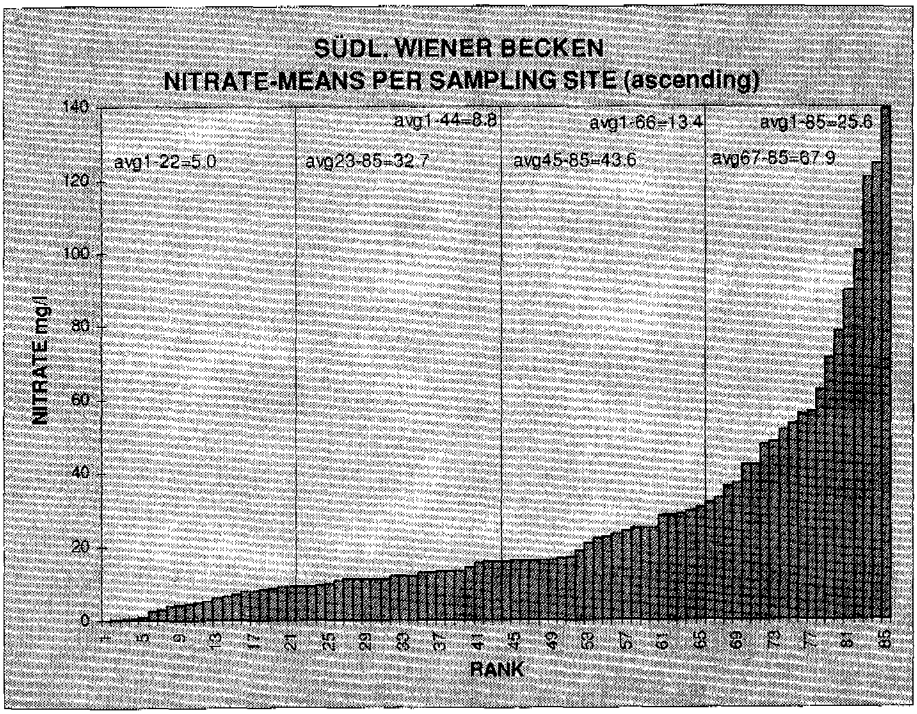


Figure 5: Effect of the criteria for "splitting" groundwater areas on the "average values" of subsamples.

extremes are not reflected. In many investigations dealing with water problems the median value has been chosen to characterize the situation. In our case the arithmetical mean seems to be more appropriate.

Whenever groundwater situations of different aquifers are compared, quality-data should include information on data distribution (quantiles, frequency histogram etc.) based on representative values for single sites. Mean values on their own are insufficient to characterize a larger area.

CONCLUSIONS

Examples from data of the AWQMS show, depending on the consolidation strategy applied, that resulting "key figures" will be different. Therefore, it is important to define needs and targets as detailed as possible. Questions as "What is the average nitrate level of groundwater?" are too vague if applied to a higher aggregation level than a single aquifer and allow a variety of interpretations. Moreover, application of various consolidation strategies will result result in considerably different "average nitrate levels of that country" - even if the same data set has been used to achieve these figures. By comparing the groundwater situation in different areas (countries, regions) the information on the data-distribution based on representative values for single aquifers should be provided.

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TOXICITY OF COMPLEX EFFLUENTS BEFORE AND AFTER DEGRADATION

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ABSTRACT

Due to a range of limitations, e.g. presence of unknown chemicals or combination toxicity, a chemical-specific approach is insufficient to assess the environmental hazard of complex effluents. The use of biological parameters, i.e. toxicity tests, can be a useful tool to overcome these limitations. Therefore methods to assess complex effluents should include ecotoxicological tools such as toxicity tests.

The experiments presented in this paper deal with the toxicity of 10 different complex effluents in combination with an additional degradation period (after degradation in a purification plant, to simulate degradation in surface water). Toxic effects were studied using organisms from four different trophic levels, while the possible degradation was followed by measuring the DOC-content in effluent samples during four weeks. Several effluents caused toxic effects both before and after the degradation test, while specific differences in sensitivity between species were demonstrated. Furthermore it was concluded that the degradation test caused significant changes in toxicity of effluent samples, even when the degradation could not be quantified as a change in DOC-content.

INTRODUCTION

The quality of the surface water in the Netherlands and other European countries is monitored extensively for different toxic compounds and many chemical and physical parameters. Approximately 30-40 compounds are regularly monitored. However, numerous other compounds (>100,000) may be present as well. The major part of these compounds cannot be reliably quantified due to a lack of (sensitive) analytical methods, or due to the costs of sampling and analysis. Besides these analytical problems, present monitoring activities might be insufficient to give an appropriate idea of the actual water quality, due to:

- the scarcity of data concerning long-term (eco)toxicity and environmental fate
- effects of individual toxicants do not necessarily predict the interactions or the combined effects that may occur in complex mixtures
- differences in bioavailability can influence ecotoxicity significantly.

Most of these problems also occur during the assessment of environmental risks of (industrial) effluents. Especially for complex effluents, in which many (unknown) toxicants might be present, the chemical-specific approach is of limited value. In these cases assessment of the environmental risks should be based on an integrated approach, which uses both chemical and biological characteristics of the effluents.

In the Netherlands a proposal was therefore made for an assessment method of complex effluents, in order to overcome the problems mentioned. This method is called 'Whole Effluent Environmental Risk', or WEER (Tonkes and Botterweg, 1994). The WEER-method is not meant to predict the effects on the receiving surface water, but to complete the assessment of unknown components in effluents using an integrated approach instead of a chemical-specific approach. The WEER-method can therefore be seen as an addition to the chemical-specific approach.

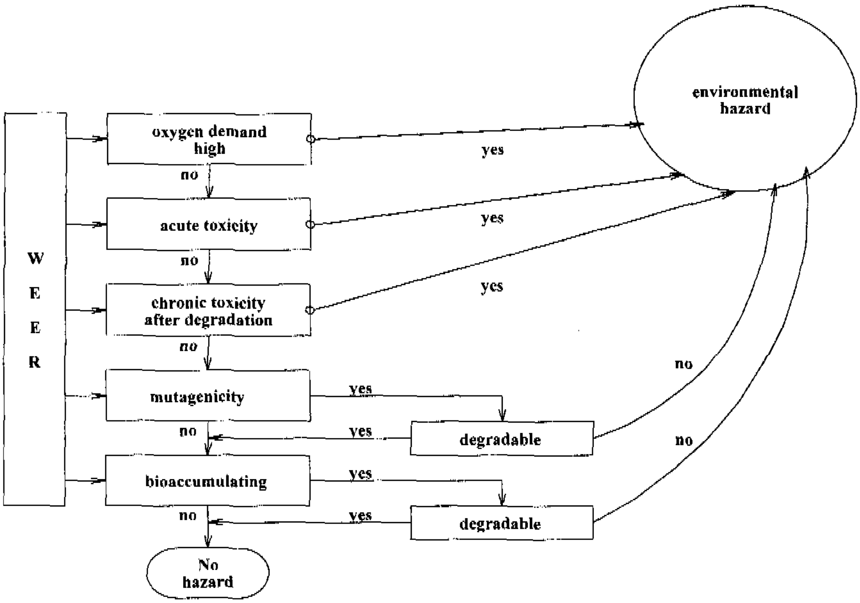

Figure 1: An overview of the Whole Effluent Environmental Risk (WEER) method (Tonkes & Botterweg, 1994).

The proposed method follows the scheme presented in figure 1 (Tonkes & Botterweg, 1994) and assesses oxygen demand, acute and chronic toxicity, mutagenicity and bioaccumulation. Next to this, a degradation test was developed as part of the WEER-method, although this test is not intended to be used as a criterium. Basic assumptions for the WEER-method are:

- all parameters are to be assessed separately and can determine the environmental hazard independently of each other;
- an estimation of the chronic toxicity by means of acute toxicity data is unreliable;
- assessing the persistence of the total mixture is not useful, since the degradable compounds do not necessarily relate to the compounds causing an environmental hazard . Therefore, parameters such as toxicity, mutagenicity and bioaccumulation should be tested before and after degradation.

Within the framework of the WEER-method, experiments were started to study the toxicity and oxygen demand of 10 different complex effluents, the results of which are presented in this paper. The primary goals of the study were 1) to determine whether the proposed toxicity tests and the measurements of the oxygen demand are sensitive enough to detect differences between effluents 2) to test the applicability of the proposed degradation test, and 3) to determine whether the degradation test can cause significant changes in toxicity.
The tested effluent samples originated from several Dutch industries as well as from municipal waste water treatment plants along the river Meuse.

METHODS

Following the framework of the WEER-method, acute toxicity tests with organisms from four different trophic levels were used to study toxicity before the degradation, while toxic effects of effluents after the degradation period were tested using chronic toxicity tests. The possible degradation was followed by measuring the Dissolved Organic Carbon content in effluent samples during four weeks.

SAMPLING AND STORAGE

After selecting 10 complex effluents, originating from different branches of industry, time-proportional samples were collected during 24 hours. All effluent samples were homogenised and stored frozen (-20°C) until use.

OXYGEN DEMAND

Biochemical (5 days) and chemical oxygen demands were measured for all effluent samples using standardised protocols (NEN 3235 5.3, 1976; NEN 3235 5.4, 1972).

ACUTE ECOTOXICITY TESTS PERFORMED BEFORE THE DEGRADATION TEST

Acute toxicity tests were performed with several organisms:

- bacteria, *Vibrio fischeri* (formerly *Photobacterium phosphoreum*, Microtox®; NVN 6516, 1993). This test assesses possible effects of the effluent samples on the bioluminescence of the bacteria
  V. fischeri within 30 minutes of exposure. The results are expressed as EC_{20}-value, which is the volume fraction (%) of effluent reducing the bioluminescence with 20% compared to control values.

- algae, *Selenastrum capricornutum* (ISO 8692, 1989).
  This test assesses possible effects of the effluent samples on the growth rate of the algae within 72 hours of exposure. The results are expressed as EC_{50}-value, which is the volume fraction (%) of effluent reducing the algal growth rate by 50% compared to control values.

- rotifers, *Brachionus calyciflorus* (Rotoxkit F™) (Creasel, 1990).
  This test assesses possible effects of the effluent samples on the survival of the rotifers within 24 hours of exposure. The results are expressed as LC_{50}-value, which is the volume fraction (%) of effluent reducing the survival by 50% compared to control values.

- crustaceans, *Thamnocephalus platyurus* (Thamnotoxkit F™) (Creasel, 1992).
  This test assesses possible effects of the effluent samples on the survival of the crustaceans within 24 hours of exposure. The results are expressed as LC_{50}-value, which is the volume fraction (%) of effluent reducing the survival by 50% compared to control values.

- water fleas (crustaceans), *Daphnia magna* (ISO 6341, 1989).
  This test assesses possible effects of the effluent samples on the mobility of the waterfleas within 48 hours of exposure. The results are expressed as EC_{50}-value, which is the volume fraction (%) of effluent reducing the mobility by 50% compared to control values.

- fish, *Brachydanio rerio* (OECD 203, 1992).
  This test assesses possible effects of the effluent samples on the survival of the fish within

96 hours of exposure. The results are expressed as LC_{50}-value, which is the volume fraction (%) of effluent reducing the survival by 50% compared to control values.

THE DEGRADATION TEST

Especially for the WEER-method a degradation test was developed, following the principles of the OECD guideline on 'Ready Biodegradability for specific chemical substances' (OECD 301, 1992). However, two important adjustments were made for this method:

- Clean surface water is taken as inoculum and mixed (1:1) with the effluent sample.
- The degradation test is performed at 15°C.

Surface water was sampled from a near-shore location in the western part of the Lake Markermeer on two different occasions, July 1996 (samples 1-5) and December 1996 (samples 6-10). The test temperature of 15°C was chosen since this resembles most closely the yearly averaged temperature of surface water in the Netherlands.

Aniline, which is readily biodegradable under aerobic conditions, was chosen as a control substance for the degradation procedure. The aniline was added to an extra test vessel containing the 1:1 mixture (= control) and chemical analyses (GC-MS) were regularly performed to verify disappearance of aniline within a few days. The degradation tests were performed in the dark during 28 days. Possible degradation was followed by measuring the dissolved organic carbon (DOC) content in weekly samples.

CHRONIC ECOTOXICITY TESTS PERFORMED AFTER THE DEGRADATION TEST

After the degradation step chronic ecotoxicity tests were performed with two organisms:

- water flea, *Daphnia magna* (OECD 202, 1993).
  This test assesses possible effects of the effluent samples on both the survival as well as the reproduction of the water fleas within 3 weeks of exposure. Results are expressed as $EC_{50\ reproduction}$, LC_{50} (respectively the volume fraction (%) of effluent reducing the reproduction or the survival by 50% compared to control values) or the corresponding NOEC-values (No Observed Effect Concentration).

- fish, *Brachydanio rerio* (Early Life Stage-tests; slightly modified from OECD 210, 1992).
  This test assesses possible effects on both the survival as well as on the morphological development of the fish. For the last parameter all larvae showing abnormality of body form are recorded. Tests were started with fertilised eggs less than 4 hours old and were ended after 8 days. Results are expressed as $EC_{50\text{-}normal\ development}$, LC_{50} (respectively the volume fraction (%) of effluent reducing the normal development or the survival by 50% compared to control values) or the corresponding NOEC-values (No Observed Effect Concentration).

RESULTS

OXYGEN DEMAND

Results concerning the oxygen demand of the ten effluent samples are presented as the biochemical and the chemical demand (table 1). The preliminary criteria proposed by the WEER-method are 50 mg/l for the biochemical demand and 200 mg/l for the chemical demand, which

	Biochemical oxygen demand mg / l	Chemical oxygen demand mg / l
Proposed criteria	**< 50**	**< 200**
sample 1	16	102
sample 2	> 16 *	147
sample 3	2	74
sample 4	2	37
sample 5	3	32
sample 6	8	190
sample 7	3	25
sample 8	< 1	11
sample 9	2	44
sample 10	3	82

*=test failed to produce a good estimation of the biochemical oxygen demand

Table 1: Biochemical and chemical oxygen demand of ten different effluent samples as well as preliminary criteria as part of the WEER-method.

means that effluents exceeding those criteria are considered to pose a potential environmental hazard. Results indicated that most effluents stayed well below these criteria, with an exception for the chemical oxygen demand in effluent sample 6. Analysis for effluent sample 2 failed to produce a good result for the biochemical oxygen demand.

ACUTE ECOTOXICITY TESTS PERFORMED BEFORE THE DEGRADATION TEST

An overview of the results for all 10 effluent samples is presented in table 2. Based on these results, it was concluded that out of the 10 samples tested 7 samples showed effects in at least one of the tests (table 2). However, only 1 effluent sample exceeded the preliminary criteria proposed by the WEER-method (effect parameter lower than 10 volume fraction (%) of effluent) and could, therefore, be considered to be a potential environmental hazard.

	Microtox EC_{20}	Algae EC_{50}	Rotoxkit LC_{50}	Thamnotoxkit LC_{50}	Daphnia EC_{50}	Fish LC_{50}
Proposed criteria		**≥10 % v/v**	**(ident. to. ≥ 100 ml/l)**			
1	-	-	-	-	-	-
2	8	10-100 *	-	-	-	-
3	41,8	10-100 *	-	-	-	-
4	-	-	-	-	-	
5	-	-	-	-	-	-
6	39,9	51,8	-	-	-	-
7	-	55,4	-	-	56,7	-
8	-	73,2	-	-	-	-
9	-	-	-	-	54,7	-
10	-	51,4	-	-	-	-

\- = no effects observed (effect parameter > 100%)
* = dose-effect relationship was unclear.
% v/v = volume fraction (%) of effluent

Table 2: Results of acute ecotoxicity tests (% v/v) performed with ten effluent samples (before degradation) as well as preliminary criteria proposed by the WEER-method.

Furthermore, differences in sensitivity between tests were observed. In the algae toxicity test, for example, effects were demonstrated in 60% of the effluents. In two of these effluents dose-response curves were less distinct since some concentrations also stimulated the growth rate of the algae. On the other hand, acute toxicity test with the Rotoxkit, the Thamnotoxkit and the fish did not show effects in any of the effluents.
It is concluded that differences in toxicity response were found both between effluents as well as between tests.

THE DEGRADATION TEST

Results of the degradation test are shown as the percentage of change in the concentration of dissolved organic carbon (table 3). Although the degradation test is an essential part of the proposed WEER-method it is only an auxiliary test. Therefore no criteria are set for this test. The weekly measurements on the amount of Dissolved Organic Carbon (DOC) in the effluent samples demonstrated a significant decrease in 2 out of 10 effluents tested, while the DOC-content of the other effluents samples was stable or fluctuated only slightly. An example of the weekly DOC-measurements is presented in figure 2. The results illustrate a constant DOC-content in effluent samples 6 and 8, while the DOC-content of effluent sample 2 decreased during the tests with 30%.

	Biodegradation test % change after 28 days
Proposed criteria	**no criteria set**
1	- 40 %
2	- 30 %
3	-
4	-
5	-
6	-
7	-
8	-
9	-
10	-

- = no significant change observed

Table 3: Average change (%) in Dissolved Organic Carbon-content after four weeks in ten effluent samples.

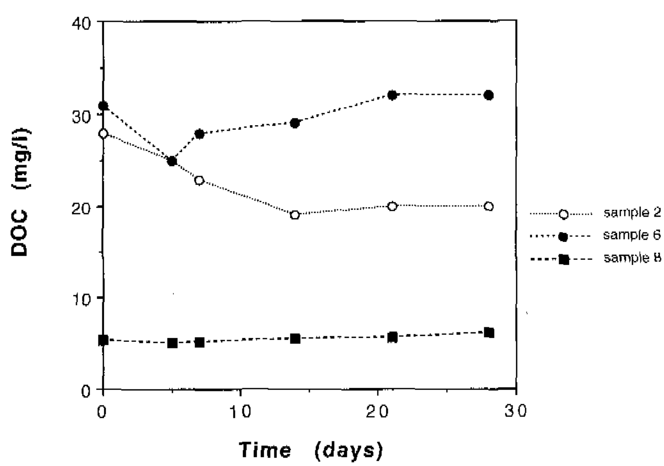

Figure 2: An example of the results of the standardised degradation step, showing the weekly measurements of DOC-content in 3 effluent samples.

CHRONIC ECOTOXICITY TESTS PERFORMED AFTER THE DEGRADATION TEST

An overview of the results for all 10 effluent samples is presented in table 4. It was concluded that 6 out of the 10 samples tested, showed effects in at least one of the tests and, conse-

	Waterflea D. *magna*		Fish B. *rerio*	
	NOEC mortality	NOEC reproduction	NOEC mortality	NOEC development
Proposed criteria	**No effects (NOEC > 50 volume %)**			
1	-	-	-	-
2	-	-	5*	-
3	-	9	< 2,8*	-
4	-	-	< 2,8*	-
5	-	-	< 2,8*	-
6	-	9	-	-
7	9	-	-	-
8	-	-	-	-
9	-	-	-	-
10	--	-	-	-

- = no effects observed (NOEC > 50%)
* =dose-effect relationship was unclear

Table 4: Results of chronic toxicity tests performed with ten effluent samples (after degradation) as well as preliminary criteria proposed by the WEER-method. Data are presented as volume fraction (%) of effluent. Maximum effluent concentrations are 50%, due to a 1:1 mixture with surfacewater.

quently, exceeded the proposed criteria (no effects in chronic toxicity tests after degradation)[1]. For three of these samples effects were, however, only observed in the ELS-test with the fish B. rerio and without the detection of a clear dose-response relationship. Since low effluent concentrations significantly reduced the survival of the fish larvae, while no effects were observed in higher concentrations, it can be questioned whether the effects were caused by toxicants and whether the effluent samples should indeed be considered to pose an environmental hazard. The chronic toxicity tests with the water flea further demonstrate that differences between effluents can not only be found between tests but also within tests, depending on the endpoint. Effluent sample 3, for example, did not affect the survival of the water fleas but the reproduction was strongly decreased, while effluent sample 7 did significantly reduce survival although the reproduction of the surviving water fleas was not affected.

	Oxygen demand	Acute toxicity	Chronic toxicity	Potential Environmental Hazard
1	-	-	-	-
2	?	Yes	Yes ?	**Yes**
3	?	-	Yes	**Yes**
4	?	-	Yes ?	**?**
5	?	-	Yes ?	**?**
6	-	-	Yes	**Yes**
7	-	-	Yes	**Yes**
8	-	-	-	-
9	-	-	-	-
10	-	-	-	-

- = Effluent sample did not exceed proposed criteria.

Table 5: Overview of the ten effluents tested, showing the samples exceeding the preliminary criteria which are part of the proposed WEER-method. The presented judgement on potential environmental hazard is preliminary, since not all tests proposed by the WEER-method were performed and criteria used are not yet agreed upon.

INTEGRATING THE EFFECTS BEFORE AND AFTER THE DEGRADATION STEP

The results presented in table 2 and 4 demonstrated several differences in toxicity before and after the degradation test. Effluent sample 9, for example, reduced the survival of water fleas significantly when tested before degradation, while no mortality could be demonstrated after the

[1] Criteria is set at NOEC > 50 volume % effluent, since this is the highest concentration tested due to the 1:1 mixture of effluent and surface water.

degradation step. Effluent 6, on the other hand, did not affect survival of the daphnids in acute toxicity tests, while low NOEC values were detected during chronic toxicity tests after degradation. Furthermore, it should be noted that degradation was not detected (based on the DOC-measurements) in either effluent sample 6 or 9.
An overview of the results is presented in table 5. Those effluents exceeding the preliminary criteria proposed by the WEER-method are indicated. It should be noted that all parameters are to be assessed separately and can determine the environmental hazard independently of each other. The overview is preliminary since the results represent only part of the total WEER-method.

DISCUSSION AND CONCLUSIONS

In this study clear differences between effluents were detected in both the (biochemical and chemical) oxygen demand as well as in the (acute and chronic) toxicity tests. Furthermore, specific differences in sensitivity were demonstrated between toxicity tests. The acute test with the algae S. *capricornutum* was especially sensitive and showed effects in 60% of the effluents. Acute toxic effects were also detected using the Microtox-test and the test with *Daphnia magna*, while no effects were observed using the Rotoxkit F, the Thamnotoxkit and the acute test with the fish *Brachydanio rerio*. Based on all test performed, it is concluded that the discriminating power of tests is suitable.
Even though all effluents were sampled after treatment by an appropriate waste water treatment, a significant reduction in DOC-content was demonstrated during the degradation test in 2 out of 10 effluents. It can therefore be concluded that the developed degradation test is a valuable addition to the WEER-method, simulating processes with might occur in the receiving surface water.

Based on the toxicity results for daphnids, it was concluded that the degradation step caused significant changes in the effluent toxicity. However, these changes could not be addressed to a reduction in DOC-content. Possible changes in the toxicity for the other organisms could not be assessed, because these tests were not repeated after the degradation step.

In summary, the sensitivity and applicability of the presented tests seems satisfactory. Consequently, the proposed toxicity tests are a valuable addition to the chemical-specific approach and as such a suitable tool to assess the environmental hazard of complex effluents.

ACKNOWLEDGEMENTS

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MONITORING COMPLEX MIXTURES; THE ROLE OF ECOTOXICOLOGY.

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ABSTRACT

Due to the enormous number of potentially polluting substances, a chemical-specific approach is insufficient to provide the information needed to protect surface waters from pollution effects. This is especially the case for complex mixtures. Therefore it is essential to develop chemical and biological tools to signal changes in and control water quality.

Ecotoxicology can provide an additional type of information next to physico-chemical types of information. At this moment ecotoxicological tools or parameters are already available to tackle problems or limitations which are for example encountered in the case of complex mixtures. It serves as an important addition that makes a more integrated type of decision making possible.

All in all, this paper focuses on how to build up tailor made monitoring programmes using present day ecotoxicological methods and techniques, in order to fulfil the information needs for pro-active water quality management.

INTRODUCTION

The monitoring of effluents, surface waters or sediments, and assessing their risk to human or environmental health by looking at individual substances, is not always feasible. In many cases effluents, surface waters and sediments consist of mixtures of substances. These mixtures are mostly very complex in their composition. This makes it difficult to get a complete impression of the potential hazards or risks. The most important bottlenecks are:

1. a complete chemical-specific analysis is not attainable because of detection limits and (high) costs;
2. many substances are unknown;
3. the mixture can have different toxicological properties in comparison to the properties of the separate substances, because of the combined mode of action;
4. the characteristics (ecotoxicological or chemical) of many substances are not available or incomplete.

The application of ecotoxicological tools makes it possible to tackle these limitations.
Toxicity tests, are for instance, often the most practical method to determine the toxicity of complex mixtures. And, combined toxicity in complex mixtures might better be dealt with by implementing group parameters, a toxic unit concept for known chemicals and the use of biological criteria for bioassay or field measurements to address the components which are not identified (van de Guchte, 1995).

In this paper the role of ecotoxicology in the assessment of complex mixtures will be discussed. In order to do this attention is given to the present water management policy, the needs of decision-makers, the available ecotoxicological tools and the future needs and developments.

PRESENT POLICY IN RELATION TO COMPLEX MIXTURES

In Europe two approaches are frequently used in water quality management. They are the emission-based approach (e.g. BAT, BEP) and the water quality-based approach (e.g. EQS, biological surveys). Both have their advantages and disadvantages. The differences between both approaches are evident (table 1).

	Emission-based	**Water quality-based**
Effluent limits	No site-specific load	Site-specific concentrations
Required treatment techniques	Based on intrinsic (toxic) properties of chemicals in effluent	Based on water quality criteria or preventing toxic effects in the receiving water
Knowledge requirements	Basic chemical and ecotoxicological data	Basic chemical and ecotoxicological data; physical, chemical and biological characteristics of the receiving water, and the fate and effects of discharged chemicals
Monitoring	Effluent	Receiving water
Competition[1]	Equality for the law	Inequality
Practice	May tend to worst case approach	May tend to dilution as a solution

[1] Competition between industries or companies in general.

Table 1: Differences between the emission and the water quality-based approach. *Source after Stortelder and van de Guchte, 1995.*

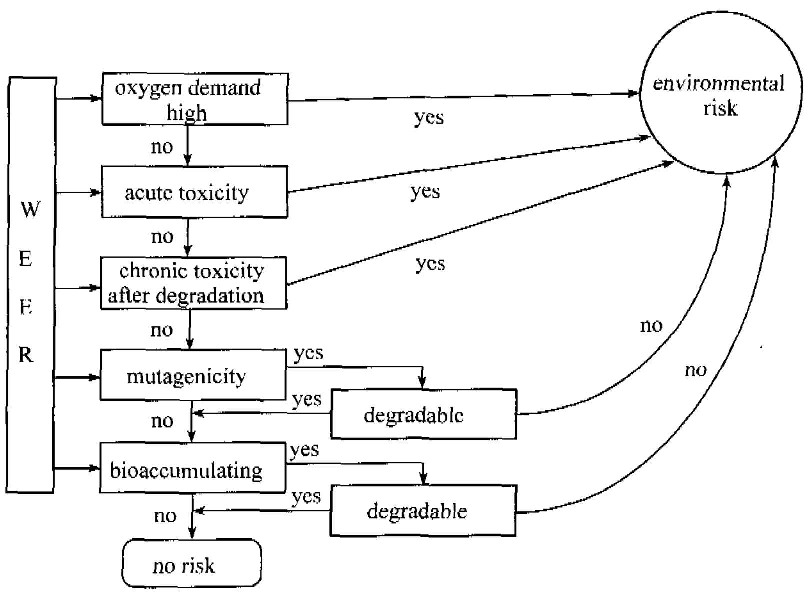

Figure 1: The Dutch Whole Effluent Environmental RISK (WEER) method. *Source after Tonkes and Botterweg, 1994.*

In the Netherlands, the emission-based approach is the predominant step in the control and the management of water pollution caused by waste water discharges. Next to this the water quality-based approach is used to assess the effects of discharges on water quality standards. These generic standards have been implemented in order to protect the whole aquatic system (e.g. a river or a lake). In contrast to the water quality-based approaches in the US and UK, only little attention has been given to the assessment of the effects of specific discharges.
In all countries a combination of both approaches is applied. However, the balance of this combination differs (strongly) among countries.

Both the emission and the water quality-based approach are based upon a chemical-specific assessment. However, for complex mixtures this assessment is insufficient (see introduction). Therefore in the Netherlands a new method has recently been proposed, especially for complex mixtures. It is called the Whole Effluent Environmental Risk, or WEER (figure 1). It is meant to complement the assessment of unknown components in the chemical-specific assessment. Use is made of different tests to assess toxicity, mutagenicity, persistence, bioaccumulation and oxygen demand of complex mixtures. This method should make it possible to demand (additional) improvement of waste water discharges where a chemical-specific evaluation is unreliable or incomplete.

In this respect the development, evaluation and improvement of effect-oriented (biological) indicators in general is needed and promising. Ecotoxicology plays an important part in this. Examples are the use of bioassays, bio-alarming methods and biological surveys.
Whole Effluent acute Toxicity (WET) tests are among the recent developments within the - Dutch - emission-based approach.

WHAT IS NEEDED IN PRACTICE?

As water managers need to make decisions, monitoring programmes should be designed in such a way that account is made of their needs by using criteria which address topics like cost-effectiveness and discriminatory power between sites or between regulatory measurement options.
Next to this, scientific criteria, like representative indicators, ecologically relevant parameters and cause-effect relating information should be used, to enable decision-makers to structure relevant priority scales and to make the right decisions.

The overall objective for the set-up of monitoring programmes principally is to achieve a cost-effective and impact oriented programme. Thus, in order to assess impacts data should be collected based on clear objectives. Integrated monitoring is needed to attain this, especially where the prediction of possible environmental impacts is among the objectives.
Monitoring the quality of effluents, sediments and surface waters basically supplies water managers with input data into their decision-making and regulatory frameworks.

From a management point of view one can distinguish three types of monitoring in order to attain information needed for decision-making:

- data used to signal present quality state or trends in time; e.g. the concentration of chloride in the river Rhine;

- data used to control the present state, or to reduce contaminant inputs; e.g. the concentration and amount of black list substances in waste water discharges;

- data used to predict future quality developments, where the aid of simple or complex modelling is often needed; e.g. production figures of substances, both historical and foreseen.

From an ecotoxicological point of view also three types of monitoring can be distinguished:

- chemical analysis of water, suspended matter, sediments and/or organisms;
- laboratory bioassays, biomarkers and/or biological early warning systems;
- biological field surveys on indicator species, process parameters and/or ecological communities.

Each type has its own strong and weak points, but it is supposed that the integration of all presents the most comprehensive information. Ecotoxicological monitoring aims at such an integration and thus becomes an important aspect.
This is especially the case for complex mixtures.

In 1994 an EC-project named 'Monitoring Water Quality in the Future' was initiated in order to make recommendations concerning standardisation, optimisation and organisation of monitoring activities in the European Community. In the framework of this project five reports were produced (Villars, 1995). One report focuses specifically on complex mixtures (Tonkes et al, 1995). Its objective was to present a review of possible strategies for the monitoring of complex mixtures in effluents, and to a lesser extent, the freshwater environment and sediments. Next to this it presents a proposal for one or more strategies for use in the European Union and it gives recommendations regarding these monitoring strategies. One of the recommendations is to use a stepwise procedure (going from coarse to fine) for the assessment of water and effluents, in order to gain more information at less cost. Cost-effectiveness should be an important aspect in each monitoring programme.
Furthermore, the existing monitoring strategies used in a number of countries are discussed. This is not limited to Europe. An example (figure 2) is the overall strategy for the monitoring of effluents or waste water discharges.
In 'Monitoring Water Quality in the Future' the use and development of biological, and especially ecotoxicological tools like toxicity and bioaccumulation testing are highly recommended.

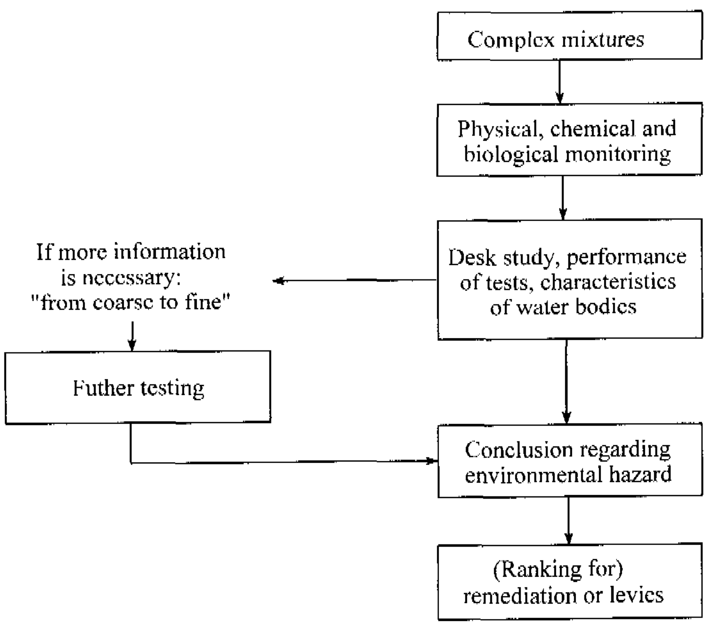

Figure 2: *The overall monitoring strategy for effluents. Source after Tonkes et al, 1995.*

Based on the above it seems that tailor-made monitoring of complex mixtures serves the needs in different management strategies, and uses different available ecotoxicological tools in optimising these programmes at different levels of information needs and data integration.

ECOTOXICOLOGICAL TOOLS

Many standardised tests are already available. However, the application of a large battery of such tests is expensive. This the reason why new cost-effective ecotoxicological tests, methods and stepwise assessment strategies are (being) developed. The characteristics of some ecotoxicological tests are presented (table 2).
The use of Whole Effluent Toxicity (WET) testing has recently been recommended in many countries, in order to conduct a more realistic environmental hazard assessment. This 'WET'-approach to control toxic waste water discharges involves the use of toxicity tests for monitoring and assessment. A number of toxicity and mutagenicity tests is available for this approach. It has to be seen as additional to the chemical-specific approach. Some advantages are:

- evaluation of prevention measures and treatment techniques without extensive chemical analyses;
- awareness of the toxicity level;
- stimulation of the identification of unknown chemicals and sources, in the case that known chemicals are not fully responsible for the toxicity;
- more accurate prediction of effects;
- more adequate and less expensive monitoring.

Compartment	No. of tests	Investments	Response time	Labour time
Effluent				
toxicity tests	10	7 low 2 intermediate 1 high	6 short 3 intermediate 1 long	7 low 1 intermediate 2 high
mutagenicity tests	4	2 intermediate 2 high	3 short 1 intermediate 1 high	2 low 1 intermediate
degradation tests	3	2 low 1 intermediate	2 intermediate 1 long	2 low 1 intermediate
Surface water	see effluents	see effluents	see effluents	see effluents
Sediment				
toxicity tests	11	3 low 8 high	6 short 2 intermediate 3 long	7 low 1 intermediate 3 high
Early warning				
test set-up	3	1 low 1 intermediate 1 high	-	3 intermediate

Investments ($) : low = <2500; intermediate = 2500-5000; high = >5000.
Response time (days) : short = <2; intermediate = 2-10; long = >10.
Labour time (days) : low = <1; intermediate = 1-3; high = >3.

Table 2: Examples of ecotoxicological -laboratory- tests and some characteristics. All are well described in test protocols (i.e. OECD, ISO, ASTM etc.; source: UN/ECE, 1995).

Basically the WET-approach is part of the emission-based approach, because it is effluent-related. It is possible though that Whole Effluent Toxicity can also be of use for the water quality-based approach. The WET-approach after all, is aimed at minimising or limiting the impacts of waste water discharges on receiving water systems.
An area for future development therefore is the pairing of (whole) effluent toxicity tests or criteria with -biological- quality objectives in the receiving water ways.
Next to this, well-chosen toxicity tests may also indicate potential effects in the receiving water more rapidly than ecological monitoring methods (i.e. biological surveys).

At this moment, the implementation of these ecotoxicological tests is not a scientific problem but needs an intensive dialogue between ecotoxicologists and all parties involved in granting permits or consents to dischargers (Botterweg and Risselada, 1993). Experiences in the US with the introduction of the WET-approach show that implementation especially costs a lot of training, workshops, discussion etc., but then leads to clear results. Discharge permits in the US are now often tailored for individual effluents, including the characteristics of the receiving water.

THE ROLE OF ECOTOXICOLOGY

The development and the use of ecotoxicology for monitoring programmes and assessment policies has grown extensively during the last decade.
The relative importance of the available ecotoxicological tools differs according to the information needed to make decisions. In the case of an objective of emission reduction, e.g. priority substances, chemical information (still) is considered more important than ecotoxicological information (i.e. in this case bioassays) or ecological information (e.g. surveys of invertebrates). When on the other hand the objective is to assess the risks of pollutants or discharges for specific sites, ecological information (of the site itself) is of much more - relative - importance than the chemical specifications. In the case of quality indicators, which mostly are of importance to evaluate the present policy at a high aggregation level, the importance of both ecotoxicological and chemical information does not differ very much. Ecotoxicological data are also necessary both for the evaluation of generic quality standards and the need for a detailed site-specific risk assessment.

The above shows that the role of ecotoxicology can not be neglected. This was also clear from the overview of different monitoring strategies as presented by Tonkes et al (1995). In practically all countries and strategies which were reviewed, ecotoxicological tools or parameters played an important part.
It is concluded that ecotoxicological tools or parameters should always play their role in the designing process of monitoring programmes for complex mixtures.

THE FUTURE

Major areas of further development, which are of importance for the assessment of complex mixtures are (van de Guchte, 1995):
1. sensitive ecological measurement endpoints, having enough discriminatory power in areas with relatively low levels of contaminants;
2. field experimental exposures and methodological concepts to show the absence of effects due to chemicals, rather then their presence.

Especially in the second area lies a challenge for ecotoxicologists to develop new methodologies.

Progress in the implementation of whole effluent toxicity testing and in the development and acceptance of 'safe' water and sediment quality criteria, might result in realistic endpoints for effluent quality and for efforts in emission reduction programmes in future.

Based on the above, in the coming decades the water management policies will have to focus on (dis)proving effects and on cause-effect analysis next to risk estimates based on chemical analyses. In order to do this, methods or approaches such as Whole Effluent Toxicity (WET) and Whole Effluent Environmental Risk (WEER) will have to become a common part of water policy. Ecotoxicology plays its own, major, part in this.

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TOXICITY BASED APPROACH: LIMITATIONS, ADVANTAGES, HISTORY AND FUTURE

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ABSTRACT

Anthropogenic activity in the United States has resulted in the degradation of many natural resources. This degradation includes toxic receiving waters and sediments from effluent discharges. Since the 1980's the US Environmental Protection Agency has adopted toxicity-based regulations to prevent further degradation of our nation's waterways. Major factors in selecting toxicity tests for a battery of assays include appropriate end points, assay sensitivity, cost, level of standardization, and the overall goal of monitoring. This presentation summarizes six commonly used marine test organisms for regulatory and scientific purposes: Mysidopsis bahia (the mysid shrimp), Menidia berrylina (silverside), Arbacia punctulata (sea urchin) Champia parvula (algae), Ampelisca abdita (amphipod) and Vibrio fischeri (bacteria, formerly known as Photobacterium phosphoreum).

If toxicity is present in an effluent, dischargers may be required to remediate the effluent in order to be in compliance with their permits. Since remediation methods are relatively toxicant specific (i.e., the remediation of a metal requires different methods than remediation of an organic compound) in order to be cost-effective, it is necessary to know which compound is causing the toxicity. Toxicity Identification and Evaluation (TIE) methods are relatively simple, "low technology" procedures designed to characterize and identify toxicants in municipal and industrial effluents. Once a toxicant is identified and/or characterized, dischargers can then use cost-effective technologies to remove the toxicity. Toxicity tests and TIE methods are effective tools to diagnose and remove potential environmental toxicants, however, their results may be confounded by ionic imbalances in effluents, synergistic/antagonistic interactions, and naturally occurring toxicants such as ammonia or H_2S. TIE methods, theory and limitations will be discussed. While toxicity tests are often the most practical method to determine the toxicity of complex mixtures where thousands of toxicants may coexist, they are not used singularly. Toxicity tests may be combined with Water Quality Criteria (WQC) standards and field monitoring to ensure the overall integrity of a water system. Discharge permits are often tailored for individual effluents based on mixing rates, and physical parameters of the receiving waters. WQC levels can also be modified based upon local physio-chemical factors of the receiving water. While this approach has proven to be fairly effective in improving water quality in the last decade, the US EPA is moving towards regulations based on ecological risk assessment and ecological endpoints. With the goal of understanding the current state of ecological risk assessment, current limitations and advantages of this approach will be discussed. This presentation will cover current toxicity based US environmental regulations highlighting the problems, as well as the advantages. In addition, areas of uncertainty and current areas of research necessary to implement regulations based on ecological assessment will be summarized.

Toxicity Based Approach: Limitations, Advantages, History and Future.

INTRODUCTION

Anthropogenic activity has resulted in the degradation of many natural resources including inland and coastal waters, and sediments. In the 1960's, the United States began to establish water quality criteria which limited the amounts of specific chemicals in our nation's waterways. Despite these chemicalspecific regulations, degradation of waterways continued, evidenced by massive fish kills and increasing limits on commercial and recreational uses of our bays and rivers. Politically embarrassing episodes, such as when a portion of the Cuyahoga River in Ohio caught fire, coupled with increasing numbers of new chemicals being discharged, forced regulators to adopt an approach based on effluent toxicity to limit water pollution. In 1972, the Clean Water Act was amended to state that "discharges of toxic pollutants in toxic amounts be prohibited", and the US Environmental Protection Agency (EPA) was charged with developing definitions of "toxicity" and defining "toxic amounts". The US EPA has currently defined toxicity in terms of effluent effects on a suite of assays. The purpose of this paper is to give a brief overview of some of the existing marine tests, test requirements and problems unique to estuarine and marine environments.

TOXICITY TEST METHODS

Six test organisms commonly used to test marine samples in the US for regulatory and scientific purposes are described in this discussion. Of the six, four are accepted species for marine effluent testing: *Mysidopsis bahia* (mysid shrimp), *Menidia berrylina* (silverside), *Arbacia punctulata* (sea urchin) and *Champia parvula* (red alga). *Ampelisca abdita* (amphipod) is commonly accepted for marine sediment testing and *Vibrio fischeri* (a bacterium formerly known as *Photobacterium phosphoreum*) is a commonly used commercially available test organism (Table 1).

	USEPA Approved[1]	ASTM Approved[2]
ORGANISM		
C. parvula	X	X
A. punctulata	X	In sub-committee
M. bahia	X	X
M. beryllina	X	X
Bacterial-Liquid		X
Bacterial-Solid Phase		In sub-committee
A. abdita	X3	X

Table 1. Comparison of Levels of Standardization and Acceptance

1. US EPA 1988. Short-term method for estimating the chronic toxicity of effluents and receiving waters to marine and estuarine organisms. EPA 600/4-87/028 May 1988,
   Environmental Monitoring and Support Laboratory-Cincinnati, Office of Research and Development, US Environmental Protection Agency, Cincinnati, Ohio USA

2. ASTM, 1916 Race Street, Philadelphia, PA 19103 USA

3. US EPA 1994 Methods for assessing the toxicity of sediment associated contaminants with amphipods EPA 68/C1/00005. June 1994 Office of Research and Development. US Environmental Protection Agency, Cincinnati, Ohio USA

The mysid, fish and amphipod assays measure acute toxicity. The sea urchin and algae test measure fertilization and reproduction, respectively, and the bacterium test measures decrease in light output, or enzyme activity, relative to a control. Because of inherent differences among organisms, it is important to consider that no single assay can be sensitive to all compounds. A suite of assays is recommended in order to exploit the differing sensitivities and selectivities of various organisms and to ensure that compounds of concern are detected (Dutka and Kwan 1982, Samoiloff 1989, Luoma and Ho 1993).

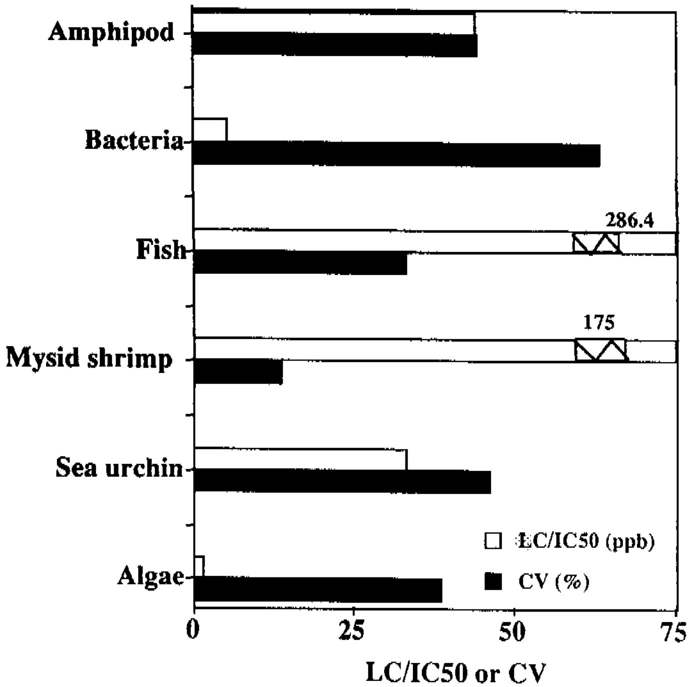

Figure 1: Comparson of LC/IC50S and CV for Copper See Table 2 for Literature references.

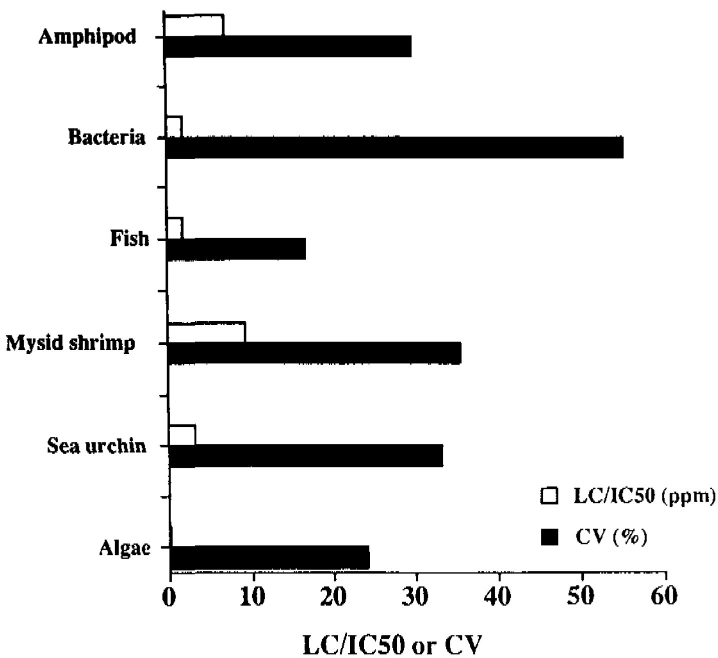

Figure 2: Comparson of LC/IC50S and CV for Copper See Table 2 for Literature references.

	Copper (ppb)		SDS (ppm)	
ORGANISM	LC or IC50	CV (%)	LC or IC50	CV (%)
C. parvula (N= 6)	1.4	38.6	0.3	24.1
A. punctulata (N= 5)	33.3	46.4	3.2	33.1
M. bahia (N= 5)	169.3	13.6	9.3	35.5
M. beryllina (N= 5)	286.4	33.2	1.8	16.7
Bacteria1	5.1(N= 4)	63.4	1.9 (N = 7)	55.2
A. abdita	44.13 (N=3)	44.3	7.12 (N=161)	29.8

Table 2. LC50s and COEFFICIENT OF VARIATION FOR SIX MARINE TEST ORGANISMS FOR COPPER and Sodium Dodecyl Sulfate (SDS)
All values obtained from round-robin tests [Morrison et. al. 1989] except Microtox values for copper and SDS and ampelisca values for SDS.

1. Adapted from (Kaiser and Palbrica 1991)
2. All tests performed at one laboratory over the course of three years. (Unpublished data from Science Application International Corporation, Narragansett, RI.)
3. Schlekatt (Unpublished data from Science Application International Corporation, Narragansett, RI.)

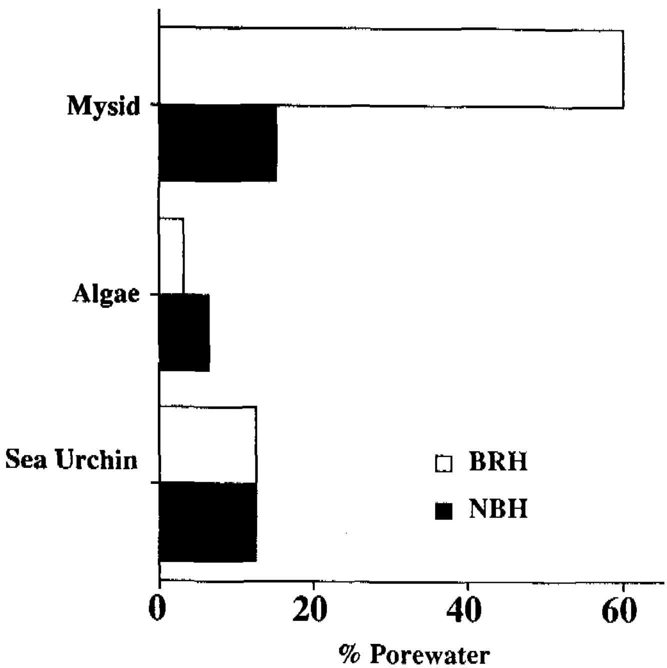

Figure 3: *Comparison of NOECS for three species from two marine estuaries:* Black Rock Harbor (BRH) and New Bedford Harbor (NBH) (Burgess *et.al.* 1993a)

Table 2 and Fig 1 -3 illustrate varying sensitivities of the different test organisms to single compounds and environmental samples. LC50's vary over a range of three magnitudes for copper and two magnitudes for SDS. Coefficients of variation (CVs) range from 14-63% with the majority less than 35%. The bacterium CVs are relatively high because they are literature values from tests performed in different laboratories without the oversight of round-robin guidance. Test duration and cost of assay (Table 3) are considerations particularly if the samples are to be used for determining the next series of steps in time-sensitive experiments, regulatory actions or if the sample load is very large. Of the tests considered, the bacterial test is the most rapid (less than one hour), and the amphipod test is the longest (10 days).

	Start-up costs	Maintenance costs/month in US dollars2	Cost (one dilution series/ one control)	Cost (10 samples/month one year)
ORGANISM				
C. parvula		650	500	20,000
A. punctulata	500	350	300	12,000
M. bahia (4 day)	350	1000	500	27,000
M. beryllina (4 day)	350	1000	500	27,000
Bacteria-Solid Phase	*20,000*		*30*	*25,000*
A. abdita (10 day)	(Field collected every two weeks)	350	1,000	28,000

Table 3. Comparison of Approximate Assay Costs1

1. Assuming utilities, saltwater systems, deionized water exist.
2. Cost of maintaining culture including food for organisms. Includes technician costs ($20.00/hr). Does not include data analysis.

OTHER APPROACHES AND LIMITATIONS OF THE TOXICITY BASED APPROACH

Limitations to environmental protection with the toxicity based approach include the exclusion of non-point sources from regulation. These sources include urban runoff containing PAHs from vehicle exhaust; agricultural runoff containing pesticides, herbicides and nutrients; and atmospheric deposition. Likewise, load allocation is often not considered when regulating different discharges into the same estuary, nor are there any examples of examining synergistic effects of one or more effluent in the same estuary. An unforeseen problem was toxicity in discharges due to their ionic balance. Produced water from oil fields, and effluents from desalinization plants for drinking water may be toxic, not due to anthropogenic compounds, but to ionic imbalances in their discharges. Because of the apparent salinity of these discharges, and the areas to which they discharge, these effluents are often tested with marine organisms. However, the physiological requirements of marine organisms are often different from the ionic balance that exists in discharge waters. In the case where addition of a chemical (e.g., more K or Mg) mitigates toxicity, the regulator must decide if adding substances to the discharge is warranted. Finally, in order for the regulator to set knowledgeable and scientifically defensible limits on effluent toxicity, flow rates, geochemistry and hydromorphology of an area must be known and taken into account. While toxicity tests are often the most practical method to determine the toxicity of complex mixtures in which thousands of potential toxicants may exist, they are not used singularly. Toxicity tests may be combined with Water Quality Criteria (WQC) standards and field monitoring to ensure the overall integrity of a water system. An example of this occurred in NY Harbor when WQC for copper were exceeded on numerous occasions)Thursby *et.al*, 1994). In response to these exceedences, toxicity tests were performed with site-specific waters from NY. Toxicity and chemistry test results indicated that NY Harbor waters were able to contain copper concentrations higher than WQC levels without eliciting a toxic response. The lack of toxicity is most likely attributed to high dissolved organic carbon (DOC) concentrations in the water which forms a non-bioavailable complex with the copper. As a result of this examination, WQC limits were increased for this specific site. The next evolution in the toxicity based approach was to develop methods to identify toxicants in effluents and sediments. Toxicity Identification and Evaluation (TIE) methods are relatively simple, "low technology" procedures designed to characterize and identify toxicants in municipal and industrial effluents (Norberg-King, *et.al.* 1991, Burgess *et. al.* 1993b). Once a toxicant is identified and/or characterized, dischargers can then use cost-effective technologies to remove the toxicity. Other uses of these TIE methods are to aid in regulatory decisions for dredged spoil disposal issues. The disposal options for dredged material contaminated with ammonia or hydrogen sulfide may be very different than disposal options for a sediment contaminated with PCBs or other bioaccumulative toxicants. In addition, since remediation methods are relatively toxicant specific (i.e., the remediation of a metal requires different methods than remediation of an organic compound) in order to be cost-effective, it is necessary to identify compound(s) causing toxicity prior to selecting a remediation method.

FUTURE DIRECTIONS FOR POLLUTION REGULATION

While the toxicity based approach has been fairly effective in improving water quality in the last decade, the US EPA is moving towards regulations based on ecological risk assessment and ecological endpoints. Ecological risk assessment allows us to organize our thinking by codifying a series of logical steps in approaching environmental problems. In addition, appropriate ecological endpoints allow us to incorporate the indirect effects of toxicity, for example, those manifested in changes in species interactions, habitat change and geochemistry of an area. Current limitations of this method are that ecological endpoints are not yet fully developed and are potentially costly to evaluate.

Pollution regulation is dynamic. New technology and resulting compounds bring the challenge of monitoring not only the specific compound, but its interactions with all other existing compounds. Conversely, new technologies may allow us to improve detection and monitoring methods, and ultimately our understanding of biological and chemical systems.

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CHLOROPHYLL-A MONITORING - AN IMPORTANT APPROACH TO INTEGRATED WATER MANAGEMENT IN RIVER BASINS

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ABSTRACT

The aim of the paper is to stress the necessity of the chlorophyll-a monitoring as a relevant and inomissible information source for the recognition of water ecosystem response to nutrient loading and level of eutrophication.
A high algal biomass limits not only water use but also gives rise to an important secondary loading of surface water bodies by organic matter.
Therefore the introduction of chlorophyll-a data into the routine activities of the decision-makers in water management is urgently advised to recognize and allocate priority measures for the decrease and/or prevention of nutrient load from catchment areas.

INTRODUCTION

Benefits of the biological monitoring and assessment of water quality by means of bioindicators have repeatedly been proved, especially in the cases of detecting a negative impact of organic pollution on water bodies and in the cases of biomass contamination through pollutants (Cairns, 1982; Klapwijk et al. 1995).
Recently, the successful introduction of modern industrial technologies and large-scale construction of municipal sewage treatment plants has lead to a decrease in organic and toxic pollution in water bodies of economically developed European countries. However, an urgent problem of quality of surface waters arises from the nutrient enrichment (eutrophication) and subsequent response of biotic components of the ecosystem i.e. the development of autotrophic organisms (algae, cyanobacteria, submerged plants). In this sense the chlorophyll-a represents a parameter allowing - on the basis of a simple chemical analysis - to assess functionally the situation in water bodies (Desortová, Fott and Strasškraba, 1977; ISO 10260, 1992).

RESULTS

EUTROPHICATION MONITORING IN SURFACE WATER BODIES.

It is obvious that the monitoring of concentrations and the budget of main nutrients (P, N) do not enable sufficiently the assessment of the eutrophication level as they do not include any response of the ecosystem.
Results of long term monitoring (e.g. Fig. 1) clearly indicate the increasing trend of phytoplankton biomass in stagnant water bodies accompanying the growth of phosphorus concentration which is a generally accepted limiting factor of autotrophic biomass in fresh waters.
Although the determination of chlorophyll-a has commonly been performed in research studies since the 1970s, until the 1990s it was not included among the routine monitoring regularly in all European countries (e.g. the former "socialist countries").
As yet the determination of chlorophyll-a has more frequently been applied in stagnant water

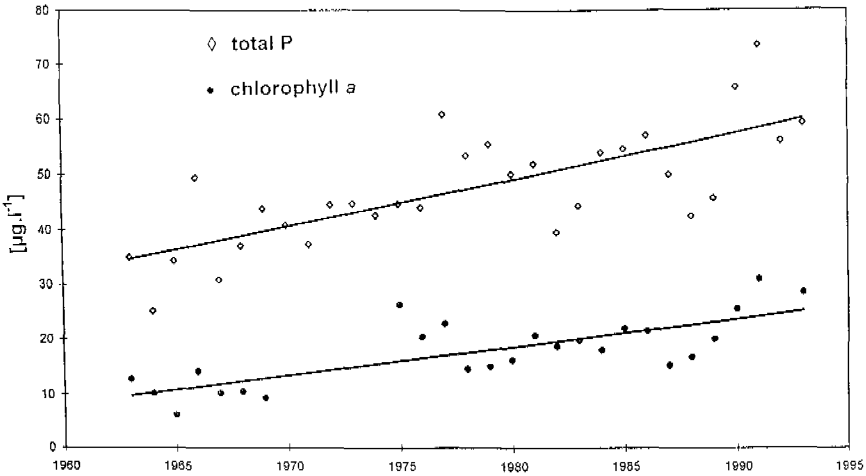

Figure 1: *Linear trends of the concentrations (annual means, 3-week interval sampling) of the chlorophyll-a and total-phosphorus in the Slapy Reservoir (the River Vltava, Czech Republic) during 1963-1993. /Data source - see Desortová & Punčochář (in print).*

bodies due to their common exploitation as drinking water sources. The available data from European rivers indicate that the phytoplank-ton density and chlorophyll-a concentration reach a very high level which is comparable with eutrophic stagnant waters during the summer period. Furthermore, the chlorophyll-a data have less frequently been discussed in connection with the stream water quality assessment and autochtonous enrichment of organic matter to the ecosystem. The importance and consequences of this Ñself-pollution" due to phytoplankton occurrence are evident especially in the lower part of the watercourses due to the development of algal biomass along the stream (Fig. 2).

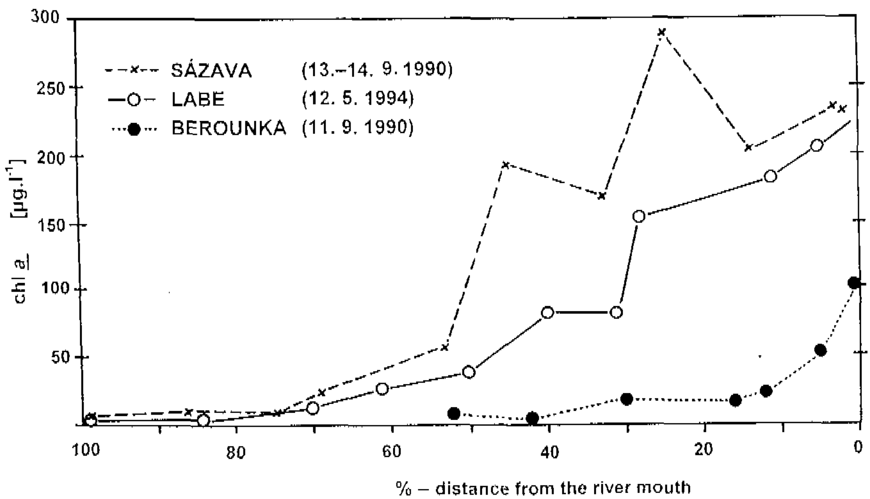

Figure 2: *Changes of the chlorophyll -a concentrations along the longitudinal profile of three Czech rivers in the vegetation season.*
The River S·zava (length 226 km, Q = 24.25m^3.s^{-1}), right tributary of the Vltava River).
The River Berounka (length 246.4 km, Q = 37.16m^3.s^{-1}) left tributary of the Vltava River).
The River Elbe (length 364.5 km to the state border, average flow-rate
Q = 313.8 m^3.s^{-1}

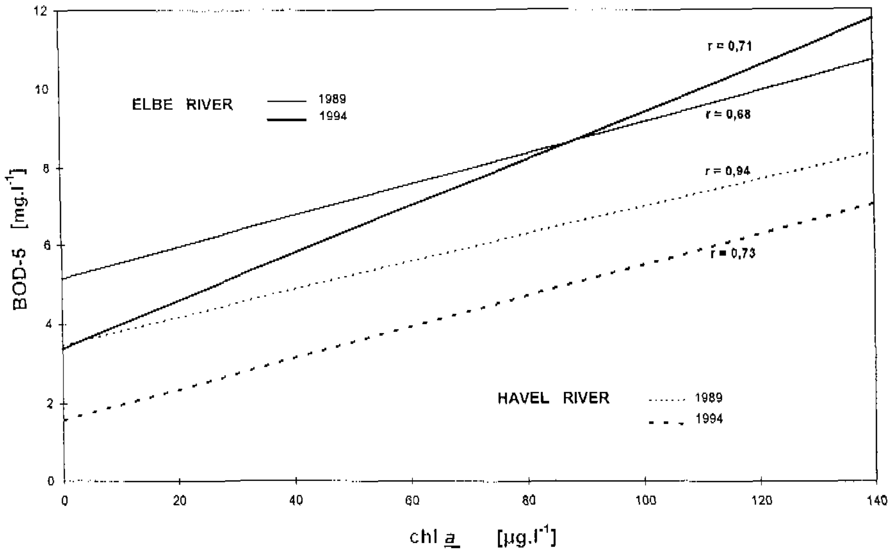

Figure 3: Correlations between chlorophyll a concentrations and BOD-5 in two rivers (the River Elbe near Magdeburg and the River Havel above its mouth) based on the comparison of the data from 1989 and 1994. The equations and number of measurements:

Elbe	1989	$y = 5.141 + 0.040$	(n = 45)
	1994	$y = 3.377 + 0.060$	(n = 28)
Havel	1989	$y = 3.464 + 0.035x$	(n = 12)
	1994	$y = 1.566 + 0.039x$	(n = 21)

The correlation between BOD-5 and chlorophyll-a documented in two European rivers (Fig. 3) obviously indicates the consequences of potamophytoplankton biomass for BOD values.
Thus the phytoplankton occurrence significantly decreases water quality assessments employing BOD as a parameter for organic loading. An appropriate interpretation of the actual enrichment of organic matter due to algal biomass is, therefore, questionable as the BOD after 5 days can consist of algal cell respiration rather than bacterial oxygen consumption for mineralization.
The similar character of the described regression lines indicates this process as a background of BOD-5 caused by allochtonous pollution discharged from point sources noticeably decreased during the period of five years in both rivers owing to the improvement in catchment areas (Dörr, 1995).
Nevertheless, an increasing importance of self-pollution of the river ecosystems due to the phytoplankton development can be expected during the vegetation season when the eutrophication is less controlled than organic pollution from point sources.

CHLOROPHYLL-A DATA IMPLEMENTATION

A shift has recently been evident towards an ecological assessment of the water ecosystem (Noordhuis et al., 1995) which is based on the monitoring of abiotic and biotic components, including their relations and functional characteristics in the sense of Agenda 21 of the Rio de Janeiro Conference.
The necessity of this approach increases especially under the conditions of a significant reduction of the pollution of waters from point sources. Thus the impact of diffuse sources increases, which is important namely for the input of macronutrients influencing the development of autotrophs in water ecosystems.
In spite of these facts the monitoring of chlorophyll-a and the necessity of phytoplankton analy-

sis for assessment of eutrophication levels in running water ecosystems have not yet been fully recognized in water management practice.
Thus the phytoplankton quality and quantity (followed by means of biomass evaluation) has not generally been accepted in routine surface-water monitoring networks of most countries. Surprisingly, the semi-quantitative analysis of potamoseston (containing frequently phytoplankton as the most common part) is still largely used for the evaluation of saprobity in spite of its insufficiency in relation to organic pollution assessment.
The chlorophyll-a determination offers an inexpensive source of information on the consequences of eutrophication in the water ecosystem. Nevertheless, the implementation of simultaneous phytoplankton analysis using a relevant methodology is advisable for a deeper understanding of the processes involved in the response of running water ecosystems to eutrophication.
Therefore it is urgently necessary to introduce the determination of chlorophyll a into the national monitoring programmes in all European countries in accordance with the fulfilment of the Helsinki Convention on the protection and use of transboundary watercourses and international lakes (1992). The procedure of chlorophyll-a determination is very simple, the basic prerequisite being the equipment of laboratories with an appropriate spectrophotometer (i.e. wave range up to 750 nm, 1 nm resolution, band width of 2 nm or less, sensitivity up to 0.001 absorbance units). It is therefore advisable to prefer this equipment in international supporting programmes, such as e.g. PHARE or the Danube Environmental Programme. Simultaneously it is imperative to harmonize the approach to assessment and classification of waters with regard to the data on chlorophyll-a, which has so far been applied rarely and rather in relation to the exploitation of drinking water resources.
The decrease in nutrient load and eutrophication level represents, however, a long-term process in which the setting of priority measures needs identification of the most beneficial and economically effective issues. Obviously, the decision process urgently needs the knowledge of ecosystem response (i.e. phytoplankton biomass development) as well as the demand for water use. This is evident particularly along the watercourses where the points of the main nutrient load and their consequences for the ecosystem behaviour are spatially separated.
Hence, the knowledge and use of chlorophyll-a data in the routine activities of the decision-makers are substantial and indispensable for integrated management in water courses and their catchment areas.

SUMMARY

The development of phytoplankton in water bodies (indicated by chlorophyll-a) represents a response of the ecosystems to eutrophication.
Simultaneously, the occurrence of algal biomass increases the BOD-5 data and has an influence upon water quality assessment.
Therefore the introduction of a routine monitoring of chlorophyll-a is necessary for an appropriate interpretation of water quality data.
The information on chlorophyll-a enables the decision-makers to recognize the causes of water quality changes as a prerequisite for setting and implementing appropriate measures in catchment management.

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DNA TOXICITY ASSESSMENT OF POLLUTED AND CLEAN WATERS

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ABSTRACT

The Bacillus subtilis rec-assay has been specially designed to evaluate DNA toxicity of various quality of waters (i.e., from polluted to clean) in only one and common scale. A variety of waters such as lake, river, rain, storm runoff, and waste waters have been tested by the assay. The assay results indicated that DNA toxicity was detected in most of them. For those types of waters, however, it is necessary to enrich samples ten to hundred times for the detection of DNA toxicity. The procedures developed are designed to concentrate hydrophobic micropollutants. The positive results of the assay for the variety of waters lead to the conclusion that many unknown micropollutants with DNA toxicity occur in public water bodies.

INTRODUCTION

An increase in number and amount of synthetic chemical utilization causes us to fear potential health hazards in the water environment. Those chemicals exert complex genotoxicity, which is chronic and difficult to evaluate, on biota including men through water comsumption and food chains.

The Bacillus subtilis rec-assay is a DNA repair test and it can detect wide spectra of DNA damages including intercalation, breakage of DNA molecules, and alkylation of DNA base (Matsui, 1984). Another important characteristic of the rec-assay, which is an advantage over other mutation bioassays (e.g., Ames methods) is its capability of evaluating DNA toxicity together with cytotoxicity. For example, the application of the Ames methods to the environmental samples having the cytotoxicity leads to death of the test bacteria, and no dose-response relationships can be obtained.

In this study, the Bacillus subtilis rec-assay was applied to various environmental water samples ranging from sewage to tap water. The direct DNA toxicity of these water samples was compared with one another in only one and common scale.

MATERIALS AND METHODS

STRAINS, MEDIA AND TESTING METHOD

In the rec-assay, *Bacillus subtilis* strain H17(Rec+) was used as a recombination-proficient strain. Its derivative strain M45(Rec-) was used as the recombination-deficient strain (Sadaie and Kada, 1976).

Firstly, both strains stored in Schaeffer's spore medium at 4°C are precultured in growth medium (Antibiotic Medium 3, DIFCO) at 37°C. The growth of bacteria is measured with an absorptiometer (BIORAD Model550) at the wavelength of 595 nm. Then, an aliquot of the logarithmic growth phase preculture is adjusted to a certain concentration : Rec+ is diluted to an absorbance of 0.001 (equivalent to 3×10^5 colony forming units/mL), while Rec- is adjusted to that of

0.002 (6 x 10^5 CFU/mL). For the direct DNA damaging tests, 30 µL of the diluted culture is mixed with 70 µL of the extracted and concentrated (those procedures are described below) test samples, which are diluted step-wise using 10 mM sodium phosphate buffer solution (pH 7.4), in each well of 96-wells microplate (Becton Dickinson Falcon 3072 Microtest III Tissue Culture Plate), and the mixture is preincubated for one hour at 37 °C. The microplate is shaken at 200 rpm on a stroke shaker {NR-3 (TAITEC Co.)} in a thermostatic chamber {BCP-120F (TAITEC Co.)}. For the detection of indirect DNA damages with metabolic activation, 10 µL of S9 mix is added prior to the preincubation. After the preincubation, 100 µL of growth medium is added and the plate is shaken at 37°C. When the absorbance of a control, unexposed to the test samples reaches 0.1 (3 x 107 CFU/mL) , the shaking is stopped and absorbance of all wells is measured.

DNA TOXICITY EVALUATION

For evaluating the DNA toxicity of a sample, a survival curve is drawn with logarithmic concentration on the abscissa and survival percentage on the ordinate. The area enclosed between the both survival curves of Rec+ and Rec- corresponds to DNA toxicity. In order to obtain the precise area between the two curves, we transformed the data mathematically to the Probit scale (Figure 1).

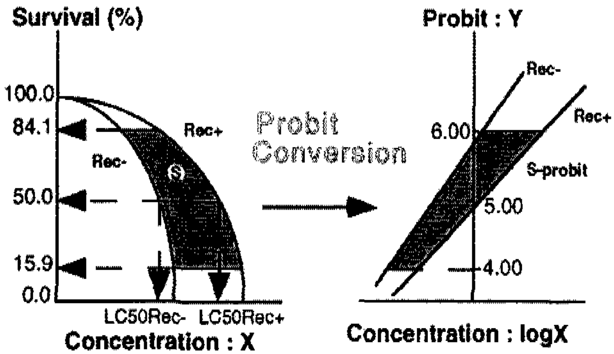

Figure 1: Survival lines for Rec+ and Rec- tested with a typical DNA damaging substance. Conversion to the Probit coordinate is shown on the right, where the S-probit is shown.

With this transformation, the two curves can be converted to two linear functions. Then the enclosed area between the two lines can be calculated by a simple integration. This integrated area, designated as S-probit, is a qualitative index to evaluate DNA toxicity. The rec-assay was applied to various organic and inorganic chemicals including genotoxic and cytotoxic substances and the criteria to assess the DNA toxicity of water samples were derived from those results (Matsui, 1988). The criteria consist of four ranges of DNA toxicity with S-probit values as follows:

> 0.593	: Strong DNA damaging potential (++)
0.200 - 0.592	: DNA damaging potential (+)
-0.123 - 0.199	: Non DNA damaging potential (-)
< - 0.124	: Reverse effect (r)

As mentioned above, the Bacillus subtilis rec-assay can detect both the DNA damage and cytotoxicity to the bacteria. Even though the integrated areas of different test samples are identical, the concentration ranges in which the survival curves are drawn could be different for each test sample. In order to consider the integrated area and the concentration range simultaneously, we further introduced Rec-volume. The value of Rec-volume is obtained by the following equation.

Rec-volume = S-probit /(the volume of sample water in liter applied to XAD-2 resin, by which 50% survival of Rec- was obtained). Therefore, the DNA toxicity becomes stronger, as the value of Rec-volume becomes larger (Matsui et al., 1989)

EXTRACTION AND CONCENTRATION OF WATER SAMPLES

In order to detect the DNA toxicity of environmental water samples by using the *Bacillus subtilis* rec-assay, a proper extraction and concentration procedure must be carried out.

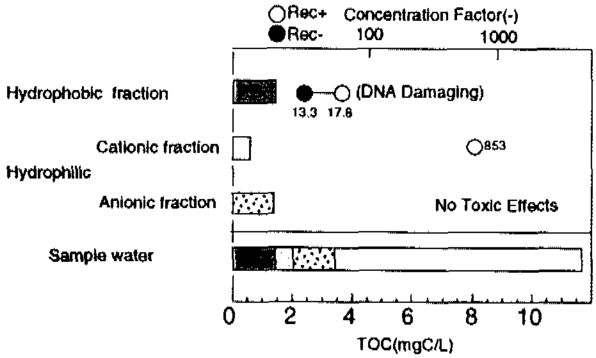

Figure 2: LC50 and TOC values obtained by an effluent from secondary sewage treatment and its three parts of fractionation. The fractionation is carried out using XAD-2, cation and anion exchange resins, corresponding to hydrophobic, hydrophilic (i.e., cationic and anionic) fractions, respectively.

Figure 2 shows the rec-assay results for the three fractionated samples of an effluent from secondary sewage treatment. Figure 2 indicates that the fraction extracted and concentrated by XAD-2 resins, shows DNA toxicity. Other fractionated samples, however, do not show DNA toxicity as well as cytotoxicity. In this research, therefore, XAD-2 resins, which can adsorb hydrophobic substances widely and effectively, were used for the extraction and concentration of water samples.

Water samples (18 L) were firstly filtered through glass wool filters followed by a paper filter (No.5A ADVANTEC). The filtrates were then passed through glass columns with the XAD-2 amberite resins (50 mL) under neutral pH condition at room temperature. The elution was conducted with ethanol and diethyl ether in this order. Distillation of eluted samples at 37°C under mild vacuum condition by a rotary evaporator was carried out, which allowed low-boiling point substances to vaporize. 10 mL of dimethyl sulfoxide (DMSO) was used for redissolving the distillation residues. With these procedures, the final concentration factor of samples became 1,800.

RESULTS AND DISCUSSION

EVALUATION OF DNA TOXICITY OF VARIOUS WATERS

Water samples were taken from several treatment facilities of various kinds. Those were effluents from sewage treatment plant, night soil treatment plant, community plant, hog urine treatment plant and cardboard factory, and leachates from landfills of municipal wastes and industrial wastes.

The rec-assay results are shown in Figure 3. Judging from the values of S-probit, all the samples except the effluent from community plant and the leachate from landfill of industrial wastes showed positive DNA toxicity. Among the water samples tested in this research, the effluent from

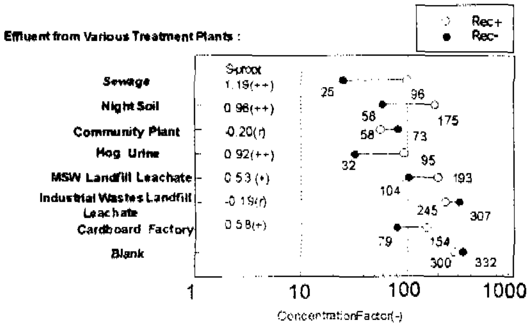

Figure 3: Bacillus subtilis rec-assay results {LC50 of Rec+ and Rec-, and S-probit (values)} of various water samples (1). The criteria of DNA toxicity of each sample is indicated in the parenthesis (++ : strong DNA damaging, + : DNA damaging, - : not DNA damaging, r : reverse effect).

a sewage treatment plant showed the strongest DNA toxicity. In the investigated sewage treatment plant, water samples were taken also before and after the primary treatment process in addition to the effluent of secondary treatment process, and were rec-assayed. The result showed that the influent sewage itself had DNA toxicity and water quality in terms of DNA toxicity went worse after the primary treatment due to the returned waste water from sludge treatment introduced before the primary treatment (Figure 4). After the secondary treatment, LC50 value of Rec- was not improved significantly while that of Rec+ was improved remarkably from 9.2 to 96.

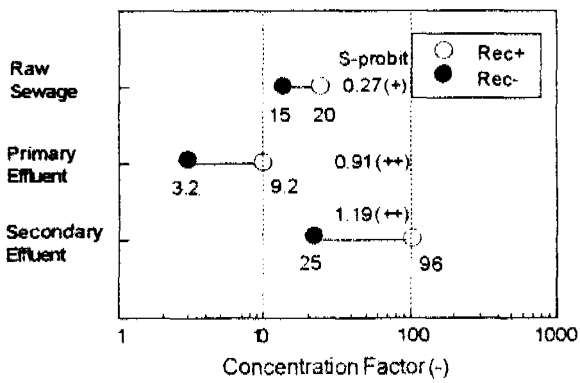

Figure 4: Bacillus subtilis rec-assay results {LC50 of Rec+ and Rec-, and S-probit (values)} of effluents of different stages from a sewage treatment plant.

A night soil treatment plant was also found to discharge DNA toxic substances. This result implies that a significant amount of DNA toxic substances may be discharged from human urine and excreta. It was found that the biological process such as activated sludge treatment is not effective to remove DNA toxic substances.

Tap water originated from the water in Lake Biwa, surface runoff and rain water were also extracted, concentrated and rec-assayed. As to surface runoff, extraction from suspended solid in the 18 L sample water was also conducted with diethyl ether. The distillation residues were redissolved in 10 mL DMSO. Figure 5 shows the rec-assay results for these samples.

The DNA toxicity was detected from surface runoff, surface runoff (SS) and rain water. The toxic level of runoffs was stronger than that of rain water. It is considered that the air pollution contri-

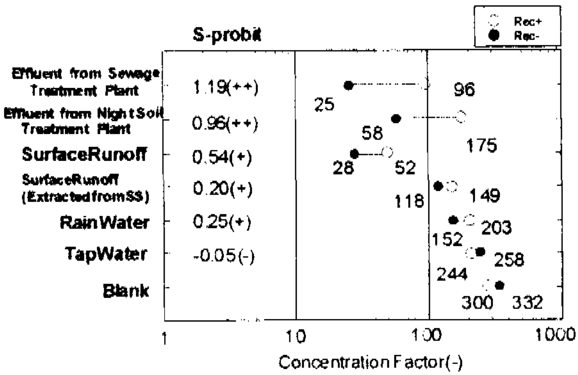

Figure 5: Bacillus subtilis rec-assay results {LC50 of Rec+ and Rec-, and S-probit (values)} of various water samples(2).

butes to the toxicity of rain water and the precipitation washes out the toxic substances contained in the surface soils. The DNA toxicity of the tap water was negative and its toxic level was only slightly stronger than that of blank, which is the distilled water sample extracted and concentrated by using the prescribed procedure.

REC-ASSAY RESULTS OF LAKE BIWA AND THE YODO RIVER SYSTEM

Lake Biwa and the Yodo river have a large basin area (7,281 km2) where about 14 million people live. Lake Biwa located upstream of the Yodo river is the largest natural reservoir in Japan of relatively good water quality for drinking purpose and other utilization in its downstream. In the middle of the basin, there are numerous urban areas including Kyoto city (1.2 million population), which discharge municipal and industrial wastewaters as well as agricultural runoffs. Since the big cities downstream such as Osaka and Kobe take municipal water sources from the Yodo river, the deterioration of water quality along Lake Biwa and the Yodo river has become one of the most important environmental issues in Japan.
In this research, for evaluating DNA toxicity in this lake and river system, thirteen points were selected along the lake and the river (Figure 6). The water samples were taken in November of 1990 and January, May, and June of 1991.

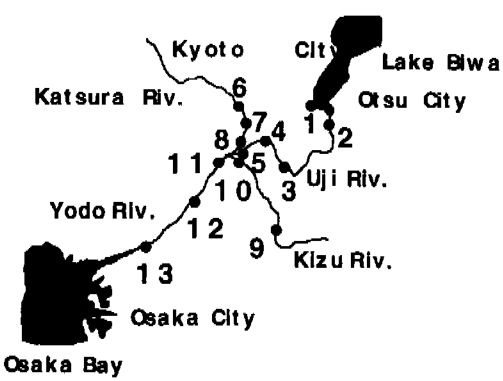

Figure 6: Location of sampling points along Lake Biwa and the Yodo River.

Figure 7 shows the results of rec-assay for these thirteen water samples. The rec-assay results of all thirteen samples shows large S-probit values (i.e., DNA toxicity). Several samples indicate very large DNA toxicity with the S-probit values equivalent to the typical positive DNA damaging substances such as MNNG (N-methyl-N'-nitrosoguanidine), MMC (Mitomycin-C) and 4NQO (4-nitroquinoline-1-oxide)(Matsui et al., 1989).

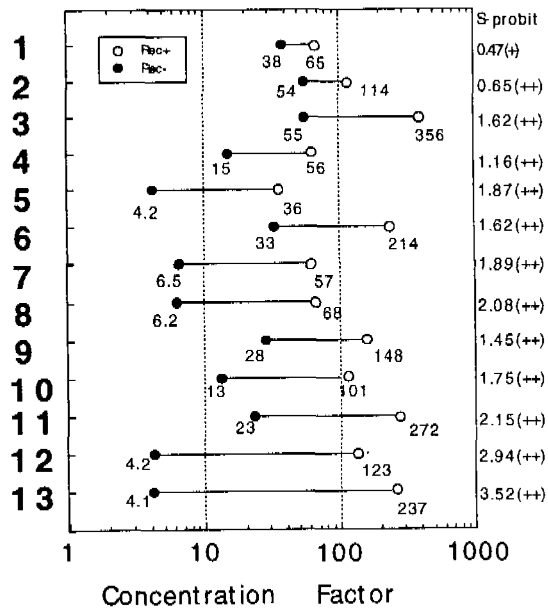

Figure 7: Bacillus subtilis rec-assay results {LC50 of Rec+ and Rec-, and S-probit (values)} of water samples of Lake Biwa and the Yodo river.

The rec-volume index can indicate how much volume of sample water is necessary to concentrate hydrophobic micropollutants and give us an information on the intensity of DNA damaging toxicity. There is an interesting relationship among sampling points No.6, 7 and 8 (Figure 8).

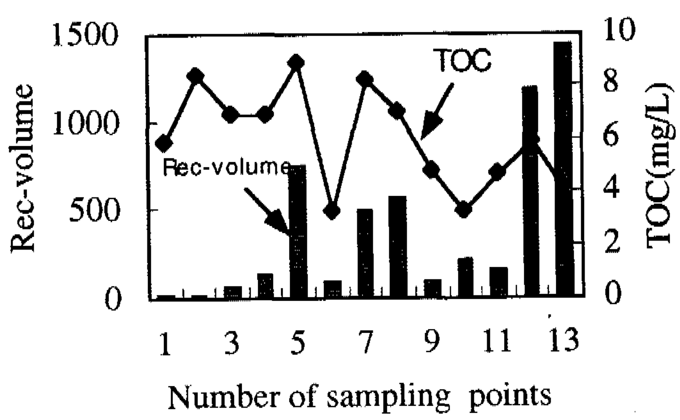

Figure 8: Rec-volume and TOC for the water samples from Lake Biwa and the Yodo river system.

The point No.6 shows slightly contaminated conditions, while No.7 and No.8 clearly indicate with the remarkable increase of rec-volume values. This is influenced by the discharge of treated sewage, industrial wastewater, untreated gray water and urban runoff from most areas of Kyoto city. After the confluence of three tributaries (Katsura river, Uji river and Kizu river), the rec-volume values decreased probably due to dilution by the water from the Kizu river. However, the rec-volume values sharply increased toward downstream again. When the rec-volume values were compared to TOC and other indices of conventional water quality, no close relationship was observed.

CONCLUSION

The *Bacillus subtilis* rec-assay which can detect DNA toxicity was applied to XAD-2 resin extractable fraction of various water samples. The DNA toxicity was detected in most of them, which means that the DNA toxic pollution extended widely in the water environment, and this type of pollution cannot be effectively removed by the current conventional secondary (biological) treatment process. The DNA toxicity was increasing toward downstream with the inflow of various effluents in the Yodo river system.

The DNA toxicity of these water samples can be attributed to numerous hydrophobic substances which are not identified yet. In order to protect the ecosystem, new wastewater technology is necessary to be developed.

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EARLY WARNING MONITORING IN NORTH RHINE-WESTPHALIA

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ABSTRACT

Water quality monitoring in North Rhine-Westphalia currently combines the ***long-term measurement programmes*** *(trend monitoring) which give information on water quality in order to protect natural waters as ecological systems as well as to safeguard its many uses and on the other hand the* ***system of intensified monitoring of natural waters*** *(INGO) which is carried out in order to detect irregular or illegal waste water discharges. The concept of early warning water quality monitoring was first put into practice in 1987, and there has continuously been further development since that time. Its purpose is the early detection of cases of significant water pollution and of shock loads following accidents, upsets or operational breakdowns in municipal and industrial sewage plants. The intensified monitoring system in North Rhine-Westphalia currently consists of 15 alarm measuring stations along the river Rhine and at its larger tributaries. The data provided by measuring stations allow the identification of polluters and enable the water works located along the river Rhine and the Ruhr area to receive water quality information in good time. The task for the next few years will be the development and installation of additional automatic equipment and analytical procedures adapted to the special pollution situation of the river areas which were monitored.*

INTRODUCTION

Water quality monitoring in North Rhine-Westphalia currently combines two monitoring systems with different objectives:

The graduated *long-term measurement programmes* give information on the water quality and its long-term changes. By using a very flexible measuring programme with three types of measuring stations, special local features of pollution and water utilisation are taken into account:

- The whole network consists of up to 3500 basic sampling sites where, depending on the water quality, biological and accompanying chemical analysis tests are carried out at least twice every five years.
- The trend measuring stations are situated in locations of particular relevance to water preservation. To select these locations, special uses requiring protection and exposed situations at junctions of catchment areas are taken into consideration. Due to detailed chemical, biological and radioactivity measuring programmes a comprehensive survey of the situation in the rivers under investigation is possible.
- Intensive measuring stations or sampling sites can be installed, where necessary, for a limited period to deal with local and regional problems, e.g. the need for or success of improvements.

Systematic monitoring of the quality of water and long-term changes in quality is carried out by the Regional Authorities in their area of responsibility and the State Environment Agency for tasks with spatial overriding importance. All state monitoring activities are harmonized by the water Quality Monitoring System of North Rhine-Westphalia, last amended in 1991 (Irmer & Vogt, 1996). By using a very flexible measuring programme, special local features of pollution and water utilisation are taken into account.

The *system of intensified monitoring of natural waters* (INGO) was installed in order to discover and follow up incidents of environmental damage and illegal discharge of pollutants, to inform drinking water treatment plants quickly and to identify sources of pollution.
After the heavy backlash to the effects of the fire in the storehouse of Sandoz AG in Basel on November 1, 1986, the Ministers in charge of the environmental protection in the countries bordering the river Rhine decided to reinforce the effort of upgrading the water quality (Goppel, 1990). The consequence of this decision in North Rhine-Westphalia was the installation of the intensified monitoring system (INGO). But before several questions had to be answered:

- Which parameters should be measured at how many sampling sites and how reliable should these data be?
- Which criteria are relevant for giving an alarm?
- How much of a time-lag is allowed between sampling and the availability of data for warning purposes?
- How can sources of pollution be detected?

PARAMETERS TO BE MEASURED

Due to investigations in the 80's information on daily changes of several physico-chemical parameters (mainly nutrients and heavy metals) is available. A special monitoring programme was carried out to get information on occurence and frequency of organic micropollutants from 1983 to 1984. The daily employment of GC, HPLC and GC/MS-analysis to river Rhine samples had confirmed the temporal fluctuations in the concentrations of several relevant compounds (Anna et al., 1985). A detailed analysis of accidental spikes of chemicals into the river Rhine in 1986 and an evaluation of analytical methods followed in 1987 (Friege et al., 1987). Due to this information a combination of substance and effect-orientated screening methods was selected for the monitoring system.
For screening purposes overview-analyses are carried out, from which any deviation from the "normal state" is easily recognizable. In these methods the water samples are analysed for organic pollutants, using gas chromatography (GC) and high pressure liquid chromatography (HPLC). The rough data obtained are checked for unusual values or constituents without applying complicated quality assurance methods (Brusske & Willemsen, 1990). The system is flexible to changes. The inventory of identified substances in river water grows with accidental spills. These substances are added to the standard, which includes the commonly occuring substances in our rivers. Dynamic biological tests (continuous-flow fish test, dynamic Daphnia assay, Dreissena-monitor) are carried out in the measuring stations using other side streams of the water sample. The water-organisms chosen indicate the composite effects of pollution loads and therefore supplement the chemical analysis by adding information on the actual effects of pollutants (Danwitz et al., 1992).

SAMPLING NETWORK

The intensified monitoring system requires a network of well equipped and staffed laboratories as well as measuring stations without staff. The monitoring system includes 15 alarmsampling stations with various capabilities along the Rhine and its larger tributaries, where they are located mostly at the confluences to monitor the impact of these rivers on the river Rhine. Fig. 1 gives an overview of the location of these stations.

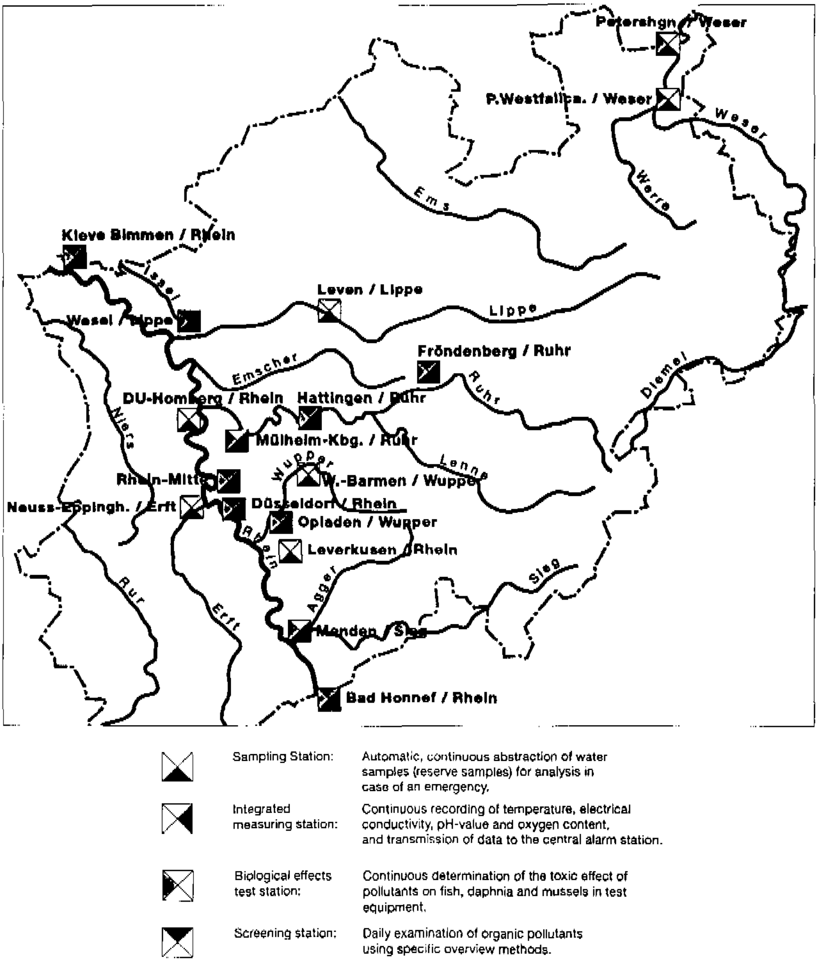

Figure 1: Early warning stations in NRW.

There is one water control station on the river Rhine at the entrance of the river to the state territory of North Rhine-Westphalia at Bad Honnef (km 640) consisting of a building with laboratories and staff. A second one is located downstream at Düsseldorf (km 744.5) below some municipal areas and industrial plants, such as the petrochemical industry in the area of Cologne and chemical industry as Bayer Leverkusen and Bayer Dormagen. The station is nearly 5 km from the main laboratory of the agency. A third station - the international water control station Kleve-Bimmen (km 865) - is located at the border to the Netherlands, so that the agency is able to control the quality of the water which is leaving state territory. This station also includes laboratories and staff.

During the last few years monitoring stations were also installed at the larger tributaries of the river Rhine, most of them at the confluences to control the influence of these rivers on the river Rhine. In this way the influence of the rivers Sieg, Wupper, Erft and Lippe is controlled. Additional stations only serve as sampling stations. They are used to follow known pollutions and to localize dischargers in case of accidents. At the river Ruhr the agency decided to control the water quality at three monitoring points near important water works.

This network of monitoring stations makes it possible to determine, whether a contamination of the river Rhine was dischanged directly into the river or if one of the tributaries is the source of pollution, as well as determining the section of the river where the event happened.

EQUIPMENT OF THE MONITORING STATIONS

There is a continuous flow of river water through each of the 15 stations. The basic module in each station is a continuously operating sampling apparatus for the collection and storage of reserve samples, so that possible pollution dischargers can be restricted and located if unusual incidents occur.

In addition physico-chemical parameters are continuously monitored in a side stream of the water sample using sensors (electrodes). The principal parameters used are pH-value, dissolved oxygen content, water temperature, electrical conductivity and turbidity. Supplementary on-line methods give data on pollution of the river Rhine by organic compounds (TOC) and salts several times a day. At the river Ruhr there are additional on-line measuring methods for NH_4^+ and Cr(VI).

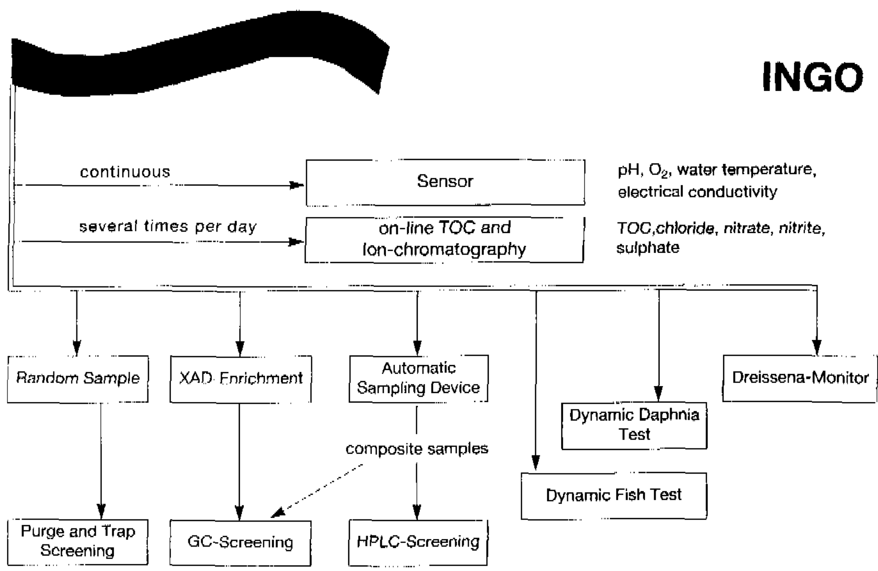

Figure 2: Equipment at the monitoring stations.

Furthermore automatic enrichment units for organic compounds (adsorber units for chromatographic screening analysis) and units for the determination of adverse effects on representative water-organisms (continuous-flow fish test, dynamic Daphnia test, Dreissena-monitor) were installed (Fig. 2).

The on-line data from the stations are passed on by remote data transmission to the alarm center of the State Environment Agency NRW in Düsseldorf. In case of an alarm a telefax is sent from the station to the alarm center in Düsseldorf, which gives an overview of all current parameters of that station.

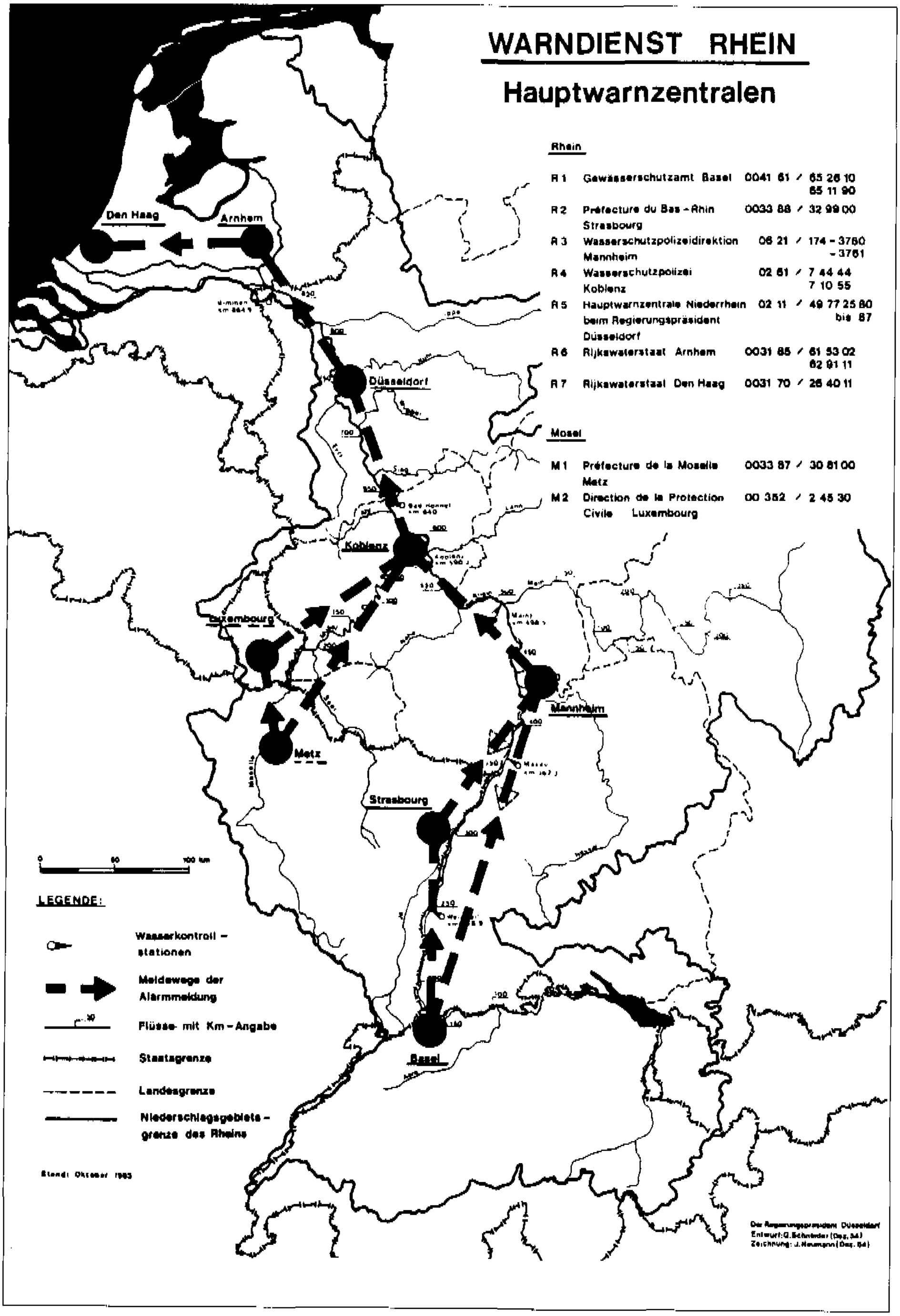

Figure 3: Organization of the international alarm and warning system "Rhine".

ALARM

The number of accidental spills of hazardous substances into the river Rhine and its tributaries and their effects on the water quality requires special action in case of such events. Therefore the International Commission for the Protection of the Rhine (ICPR) established the international alarm and warning system "Rhine" to transfer the reports on accidental spills to the neighbouring countries as well as to the national authorities concerned. The system consists of a number of warning posts along the river which receive special instructions in case of emergency. Thus all organizations concerned can be warned in the event of a sudden pollution upstream (Fig. 3).
In North Rhine-Westphalia the State Environment Agency is responsible for the analytical investigations in case of a sudden contamination and has to make an ecotoxicological hazard assessment for the North Rhine-Westphalian part of the river Rhine. This work is done with the help of the monitoring stations.

On the other hand the continuous online measuring methods - chemical and biological systems - in the monitoring stations with data transfer to the alarm center in Düsseldorf fulfill the demands of an early warning system.

If there are any significant deviations from the regular activities of organisms in the biomonitors or from the regular levels of online physico-chemical measurements, an alarm is triggered. The results are passed on to Düsseldorf automatically via remote data transmission and via telefax.

In case of e.g. organic pollutants any deviation from the normal state is established by comparing the results with standard chromatographically determined spectra, which represent characteristic "fingerprints" of the particular watercourse. Unusual or increased pollutant concentrations are identified and quantified using mass spectrometry (MS). Warning limits depend on analytical limits on the one hand and regulatory limits on the other hand (e.g. detection levels, regular concentrations in river water and demands of drinking water supplies) (Danwitz, 1991). When pollution exceeds previously agreed threshold levels, an alarm signal will be triggered. A list of concentration guide values for an information within the remit of the international alarm and warning system "Rhine" was agreed upon (Malle, 1994).

MEASUREMENT FREQUENCY

In order to be certain to hit waves of pollution it is essential that the frequency of analysis as well as the number of sampling sites is sufficient.
While on-line measurements and biological test methods run continuously, screening measurements are carried out at regular intervals on water samples and enrichments collected around the clock. The selection of measurement frequency was based on information on spatial and temporal distribution of micro-pollutants (river flow conditions), the time necessary for analysis, interpretation, validation, raising an alarm and the transit time for the water pollution from the sampling site to the important water supply intake points. Due to the dense network of sampling stations a frequency of daily analysis of 24 hours composite samples from 8 stations, carried out in the laboratories of the Rhine water control stations at Bad Honnef, Düsseldorf and Kleve-Bimmen, has proven to be sufficient.

SOURCES OF POLLUTION

The increased number of alarm measuring stations makes it possible to locate and restrict the sources of water pollution more efficiently.

Immission monitoring is supported by effluent early warning monitoring at the point of emission to the environment, covered by the industry itself. Most industrial effluent treatment plants along

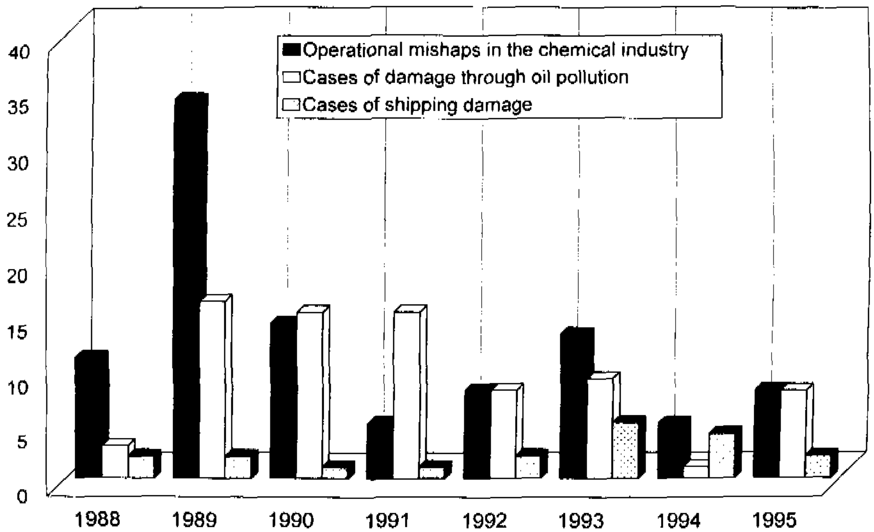

Figure 4: Alarm and warning system "Rhine": Pollution damage incidents reported by and to the State Environment Agency NRW.

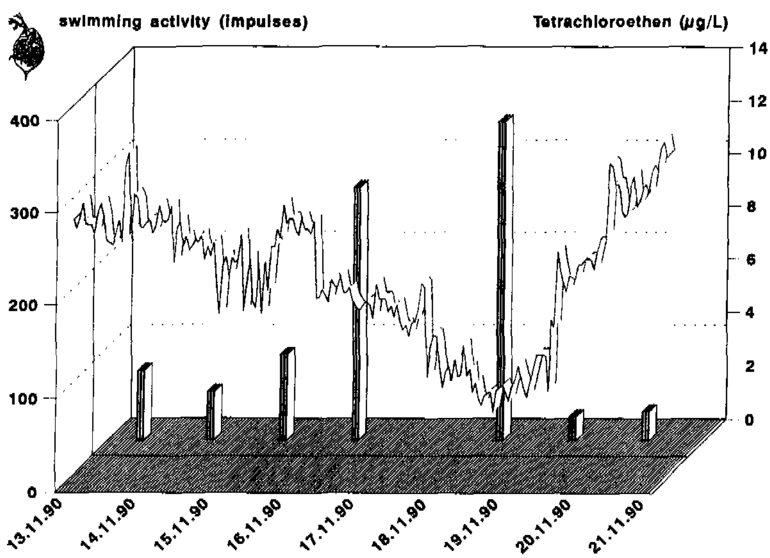

Figure 5: Decrease of swimming activity of Daphnia magna in Dynamic Daphnia assay during a spill of tetrachloroethene at the river Ruhr.

the river Rhine provide instrumentation for the detection of calamitous contaminations of the waste water with sampling frequencies agreed upon.

EXPERIENCES WITH THE SYSTEM OF INTENSIFIED MONITORING

The system of intensified monitoring in North Rhine-Westphalia was setup to safeguard the multi-functional use of water bodies (e.g drinking water supply, ecology) against sudden pollutants. The improvement of water quality of the river Rhine enabled more sensitive species to return to the river, but these organisms are also more sensitive against pollutants caused by accidental spills. An evaluation of the cases which were treated with the alarm and warning system "Rhine" from 1986 to 1996 shows that the number of alarms has decreased over the years (Fig. 4) and has reached a low level (Vogt & Lowis, 1996). The waves of pollution are documented in the annual water quality reports by State Environment Agency North Rhine-Westphalia.

The system of intensified monitoring proved to be a very useful instrument for the detection, identification, quantification and evaluation of sudden contaminations. In former times an accidental release of hazardous chemicals often had not been announced to the responsible authorities by the discharger. Today our monitoring system is able to detect most of the spills, the load of the hazardous substances discharged and consequently, most of the time, even the dischargers. Therefore, many dischargers nowadays routinely inform the authorities as soon as they become aware of an irregular discharge. In some cases they even inform press agencies because of the high public interest in these affaires. In the last four years nearly all registered cases of sudden contaminations of the river Rhine were announced by the dischargers. This allows fast and precise information for the drinking water suppliers.

Biomonitors are regarded as a valuable addition to chemical control. Besides their function as early warning systems they are used for an evaluation of results from chemical analysis. Only organisms can detect the impact of spilled chemicals and their interactions with other substances in the river. Effects, especially concerning bioavailability can only be estimated very roughly from analysed concentrations of chemicals in river water. Fig. 5 shows the decrease of swimming activity of daphnids in Dynamic Daphnia assay at Fröndenberg/Ruhr in November 1990. Parallel to this decrease Tetrachloroethene concentration, analysed in spot samples increased up to 15 µg/L. Tetrachloroethene, as the only irregular microcontaminant, would not have caused such effects, but we think this chemical can be taken as a marker of the contaminated water body, containing also non-identified substances. Results from biomonitoring can lead to additional investigations on the macrozoobenthos community of the river, e.g. in case of dead animals in the on-line testsystems.

CONCLUSIONS

The water quality monitoring systems in North Rhine Westphalia proved to fulfill their tasks. The long-term monitoring system, which has existed for more than 20 years, provides very detailed information on the improved water quality of the river Rhine and its tributaries over the time. The intensified monitoring system actually fulfills the demands of an early warning system. The monitoring programmes are continuously optimized within the constraints of a fixed resource and are adapted to new groups of substances and analytical possibilities.
The network with 15 alarm measuring stations along the river Rhine and at its larger tributaries, was completed in 1996 with two new stations in the middle parts of the rivers Rhine and Ruhr. They are closing the last gaps in the early warning system. In the future additional equipment and analytical procedures, adapted to special pollution situations, will be installed at some of the measuring stations.

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A METHOD FOR ANALYSING THE SPATIAL REPRESENTATIVENESS OF TIME SERIES GROUNDWATER HEAD OBSERVATIONS

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ABSTRACT

The spatial representativeness of groundwater head observations can not be considered to be a characteristic of a particular location. It can be different for each monitoring objective. In this paper the groundwater head is split into two components. The first component is the response of the groundwater head to precipitation excess, the second is the noise component accounting for all other influences. The representativeness of measurement series for both components is analyzed separately. First for each groundwater head series a Box Jenkins transfer/noise model was estimated to split the series in components. Then the transfer /noise models are extended spatially. The uncertainty associated with the spatial extension is dependent on the measurement locations. It provides insight into the spatial representativeness of the measurement series for each component according to different monitoring objectives. In a real-world example it is shown that for the same observation locations the spatial representativeness for the two components differs.

INTRODUCTION

An great variety of monitoring networks for groundwater heads have been designed. For each network some kind of management objective has been formulated. In most cases the management objectives are formulated in a qualitative way. Examples of management objectives are "to monitor the drawdown of the groundwater table due to a groundwater abstraction" and "to follow the behaviour of the groundwater head in space and time". Before the monitoring network can be designed the management objective have to be translated into a technical objective, expressing the information in a quantitative way. Technical objectives can be the assessment of a spatial average, the rate of change in time or a spatial interpolation of the groundwater head. Because there is always a limited amount of data, the evaluation of the technical objectives is subject to uncertainty. For monitoring network design we are looking for a relation between the uncertainty and the monitoring effort (network density and frequency). A basic assumption in network design is that an individual measurement is representative for the technical objective. In most design procedures for groundwater head monitoring networks the spatial representativeness of a groundwater head is considered to be an invariant characteristic of a particular location, related to the spatial variability of the groundwater head in space. However, the spatial representativeness is dependent on the technical objective.

In this paper the technical objectives are formulated in terms of response functions. The groundwater head is considered to be the sum of several components. To be able to split the "natural" and "man induced" variation the groundwater head is split into a component due to precipitation excess and a noise component. It is assumed that the precipitation component accounts for the natural variation, whereas the noise component consists of man-induced and random variation. In this paper the spatial representativeness of both components is quantified and conclusions are drawn related to the set up of the monitoring network.

SCALES AND MANAGEMENT OBJECTIVES

The groundwater head vanes in space and time at many different scales. In time we can define small scale as the day to day changes. intermediate scale can be the seasonal variation and the trend over ten years can be defined as large scale. Also in space several scales can be distinguished. These scales of the groundwater process are referred to as process scales. Here process scales are considered to be given characteristics of the groundwater system, that can not be influenced by the network designer. The second type of scale is the one on which the information is desired.

Groundwater management is always related to a characteristic volume in space and period in time. In general local water managers at an operational level are responsible for a relative small area and short periods of time. On the other hand national water managers at a strategic level are responsible for large areas and long periods of time. Each groundwater manager needs information at scales according to his responsibility. These scales are referred to as information scales. For the networks designer also the information scales are given. The third type of scale is the measurement scale. This scale is determined by the measurement device. the network density and the observation frequency. The aim of the network designer is to determine the optimal measurement scales in order to provide information on the appropriate information scale given the variety of process scales. The different scale types are illustrated with two examples.

In the first example the technical objective is to have an estimate of the spatial average of the concentration of a chemical component in a district of a city. The dominant process scale in space (< 50 m) is much smaller than the information scale (>1 km). In fact we consider the small scale variation of the process as "noise" around the spatial average. For a network design we need a relation between the uncertainty of the spatial average and the number of observation points. As long as the measurement scale is larger than the process scale (so the distance of the observation points is larger than 50 m) the network design is only dependent on the variation of the concentration and not on the spatial correlation of the process.

In the second example the technical objective is to have an interpolation of the groundwater head in an area of 10 km by 10 km. The information scale is in the order of a few kilometers. Let the dominant process scale be in the order of more than 15 km. The variation of the groundwater level appears as a trend in space, and the spatial correlation plays a very important role in the network design.

METHODOLOGY

The technical details of the methodology has been described in more detail in Van Geer (1996) and Van Geer and Zuur (accepted).

The objective is to analyze the spatial representativeness of the component of the precipitation excess and the noise component. Therefore these components should first be derived everywhere in the area, also at not measured locations. Secondly a relation should be formulated between the uncertainty in two components an arbitrary location and the monitoring network. The derivation of the components is performed by the following steps:

1. For each groundwater head series a Box-Jenkins transfer/noise model is fitted, with the precipitation excess as input series.
2. Estimation of the spatial structure of the transfer noise models.
3. Spatial extension of the structure of the white noise, the driving force of the noise component.

The result of these steps is that at each location in the area a transfer/noise model is defined. The relation between the uncertainty associated to these models and the monitoring network is analyzed during the steps 2 and 3. This relation provides quantitative insight into the spatial representativeness of the measurement location.

Single Box-Jenkins transfer noise model.

The starting point of the method is the transfer/noise modeling as described by Box and Jenkins (1976). We consider a simple form of this transfer/noise model with only one input series (precipitation excess) and a noise component, given by the following set of equations

$$h_{i,t} = h^*_{i,t} + n_{i,t} \qquad (1)$$

$$h^*_{i,t} = \delta_i h^*_{i,t-1} + \omega_i N_t \qquad (2)$$

$$[n_{i,t} - c_i] = \phi_i [n_{i,t-1} - c_i] + a_{i,t} \qquad (3)$$

where:

$h_{i,t}$ is the measured groundwater head at location i and time step t
$h^*_{i,t}$ is the component of the groundwater at location i and time step t attributable to the precipitation excess
$n_{i,t}$ is the noise component at location i and time step t
N_t is the precipitation excess in time step t
$a_{i,t}$ is the realisation of the white noise process at location i and time step t. $a_{i,t}$ is a zero mean process with variance $\sigma^2_{a,i}$
c_i is the expected value of $n_{i,t}$
δ_i is the autoregressive parameter of lag 1 in the transfer model of groundwater head series i
ω_i is the moving average parameter of lag 0 in the transfer model of groundwater head series i
ϕ_i is the autoregressive parameter of lag 1 in the transfer model of groundwater head series i

The precipitation excess is calculated from measurements of the precipitation and potential evapotranspiration, as:

$$N_t = P_t - 0.8^*E_{p,t} \qquad (4)$$

where

P_t is the precipitation in time step t
$E_{p,t}$ is the potential evapotranspiration in time step t according to the Penman formula in time step t

The models are assumed to be stationary. Although this assumption is a major restriction, our experience with many hundreds of groundwater head series is that this model applies to a large number of groundwater head series in The Netherlands, at least on the time scales we are interested in (5 to 40 years). A non stationary groundwater head series indicates a trend in time. A trend is nearly always caused by influences like abstraction of groundwater or a change in surface water management. In those cases we think it is preferable to model the trend by an extra input series (e.g. abstraction rates, or surface water levels) and avoid the use of differentiation in the time series model.

Estimation of the spatial structure of the transfer noise models.
The transfer/noise models given in equation (1) to (3) are defined at all measured locations. To extend the transfer/noise model to unmeasured locations, the parameters of the single transfer/noise models are considered to be spatially random processes continuous in space. This gives:

$$\delta_i = \delta(x_i, y_i) \quad \omega_i = \omega(x_i, y_i) \quad \phi_i = \phi(x_i, y_i) \qquad (5)$$

where (xi, yi) is an arbitrary location in the two-dimensional (x,y) space. Because the parameters are continuous spatial processes also the components of the transfer/noise models are defined continuously in space. In hydrological terms this means that the response of the groundwater head to precipitation excess varies continuously in space. The values of the parameters at unmeasured locations are obtained by spatial interpolation. We used Kriging interpolation. There-

fore we have to estimate a variogram model, expressing the spatial structure of each parameter separately. Using the variogram models a spatial interpolation can performed to any arbitrary location in space.

Spatial extension of the white noise structure

To examine the spatial representativeness of the measurements for the noise component, we *need an interpolation and the full covariance matrix of the white noise process.* Similar to the Parameters also the white noise process is assumed to be a spatial random field:

$$a_{i,t} = a_t(x_i, y_i) \qquad (6)$$

The spatial extension of the white noise structure is more complicated than the parameters, *because the white noise process is a function of time. To be able to extend the covariance matrix* to unmeasured locations, a model of the covariance structure has to be constructed. The general procedure for obtaining the noise covariance matrix, given the white noise series at the measured locations, consists of the following four steps:

Step 1. Consider the covariance between two points as the product of a correlation between those points and the corresponding standard deviations. This enables to estimate the spatial variance and correlation structure separately.

Step 2. Calculate the correlations between all measured locations and fit a correlation model (correlation between two points in space as a function of their distance apart). The model we adopted for the noise structure in this application is given in the next paragraph.

Step 3. Interpolate the standard deviations. The standard deviation of the noise series for each measured location is obtained from the results of the single output transfer/noise models. The standard deviations are independent of time and can be interpolated in the same way as the model parameters.

Step 4. For each element of the covariance matrix, multiply the appropriate correlation by the standard deviations.

APPLICATION

We applied *the multiple-output transfer/noise model to an area of 6 km by 10 km in the east of* The Netherlands, for which groundwater head data from 24 groundwater wells are available. We used data on head measurements relative to the Dutch ordnance level (NAP) from screens in one aquifer during the years 1986 to 1992. The measurement frequency is 2 times/month, giving 168 head data in each time series. We estimated a single transfer/noise model for all 24 groundwater head series, *using the precipitation excess as input.* The results of the single-output transfer/noise modeling are given in table 1.

x-coor km	y-coor km	δ -	ω cm/mm	ϕ -	σ_a^2 cm^2	c cm
237.66	467.44	0.84	0.38	0.49	186.44	-27.96
234.65	466.36	0.84	0.45	0.47	290.02	-32.67
237.84	472.38	0.89	0.31	0.5	173.43	-34.19
235.63	472.41	0.90	0.34	0.41	389.58	-36.49
233.40	471.72	0.97	0.21	0.76	45.22	-89.11
238.42	468.38	0.85	0.36	0.42	233.44	-28.08
234.33	467.33	0.87	0.39	0.59	184.39	-33.94
234.34	474.14	0.95	0.16	0.62	39.23	-39.8
234.74	473.44	0.95	0.23	0.78	52.33	-60.5
235.00	472.64	0.95	0.25	0.82	75.02	-62.26
235.18	471.89	0.9	0.19	0.47	134.46	-22.92
233.79	471.59	0.94	0.28	0.7	85.63	-58.74
235.64	473.54	0.86	0.26	0.44	96.42	-22.1
236.26	473.64	0.83	0.29	0.53	190.18	-19.25
235.35	472.96	0.92	0.29	0.7	88.86	-43.89
236.34	473.14	0.84	0.29	0.47	128.47	-21.67
235.97	472.73	0.87	0.31	0.58	120.43	-28.14
233.68	471.73	0.96	0.23	0.81	71.25	-80.81
235.11	473.62	0.89	0.26	0.47	96.5	-29
233.79	472.09	0.98	0.18	0.91	27.85	-116.04
236.34	472.33	0.86	0.30	0.55	146.46	-24.64
236.65	472.15	0.86	0.27	0.61	112.08	-18.99
236.35	471.79	0.81	0.11	0.33	59.45	-7.2
236.17	470.82	0.79	0.18	0.34	71.92	-9.95

Table 1. Parameters of the single-output transfer/noise models.

Table 1 indicates that the models from different locations show similarities. The autoregressive parameters (δ) of the transfer models for the precipitation excess are all around 0.9 and the moving average parameters (ω) are in the range of 0.25 to 0.40. This indicates that the groundwater head in different locations in the area show a similar response to the precipitation. The autoregressive parameter (ϕ) of the noise model seems to be more variable and the same goes for the white noise variance. The constants are all negative, indicating that the average groundwater head is below NAP. The parameters, the constant and the noise variance given in table 1 can be considered to be random spatial variables. and therefore, in order to interpolate them, we estimated variograms using the GEOEAS program (see figs. 1a to 1e).

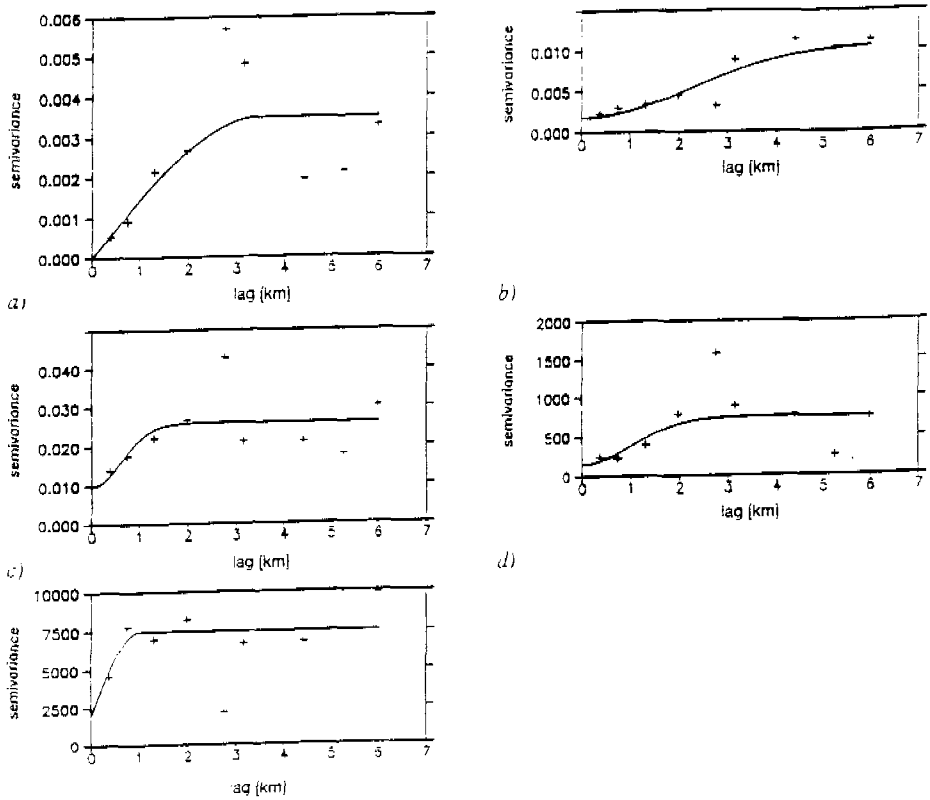


Figure 1. Variograms of the parameters δ (1a), ω (1b), ϕ (1c), constant (1d) and noise variance (1e).

Although the variograms in figure 1 show large scattering it can be seen that the parameters of the transfer model for the precipitation excess (δ and ω) do have a spatial correlation up to a distance of up to 4 km. The range of the parameter of the noise model (ϕ) is not more than about 1.5 km. This indicates that the response of the groundwater head to precipitation shows a larger spatial correlation than the noise term. The noise variance does not show a very clear spatial correlation either. As we expected, the constant C (fig. 1d) is correlated over a range of about 3 km, indicating that the mean elevation of the groundwater head is spatially correlated over this range. The values for the semivariance in the figures 1a and 1b are small by comparison with their absolute values. This indicates that the spatial variability of the models is quite small.

We used the variogram models for Kriging interpolation. The interpolated surfaces are given in figures 2a to 2e. As expected from the variograms, the interpolations of the parameters δ and ω are rather smooth. The pattern of the interpolation of the parameter ϕ (fig. 2c) shows steeper gradients. For the representativeness of individual measurement points for the response to precipitation excess, this indicates that:

1. The area shows a characteristic response to precipitation excess. Based on the measurement series an average transfer model for the precipitation component of the groundwater head can be defined accounting for the major part of the response at each arbitrary location.
2. The variation of the precipitation component around the spatial mean shows a spatial correlation up to 4 km (dominant process scale). This means that an individual measurement series contains information about this local variation up to a distance of 4 km. For a meaningful inter-

polation of the response of the groundwater head to precipitation excess the distance between the measurement locations (measurement scale) should not exceed 4 km.

3. In subareas where the distance between the measurement locations is larger, the best estimate of the response is the spatial average of the transfer model. In fact, in these areas an interpolation is not useful.

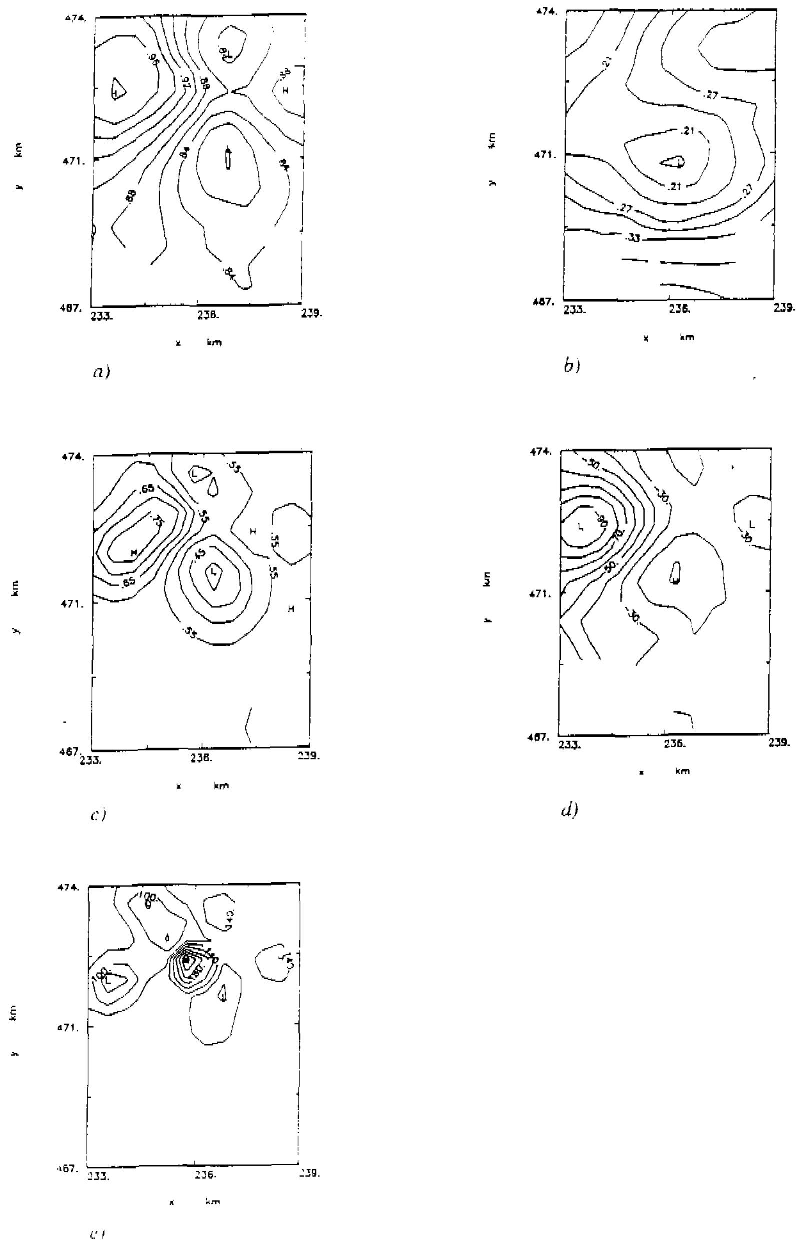
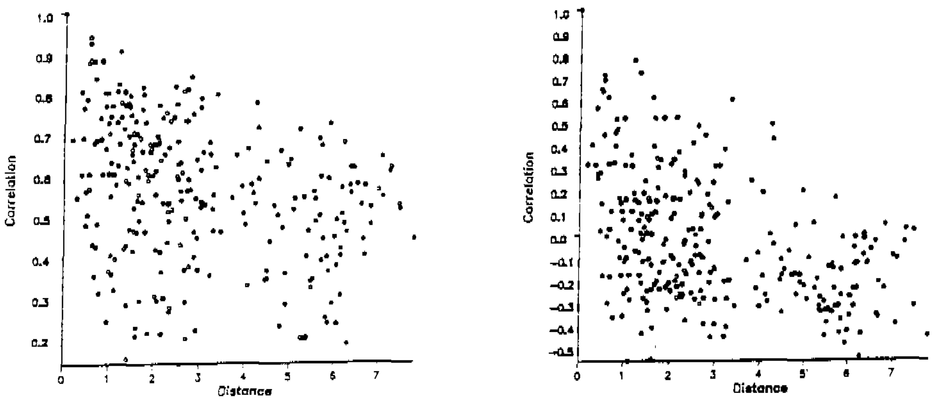


Figure 2. Kriging interpolations of the parameters, δ (2a), ω (2b), ϕ (2c) and noise variance (2e).

Before we consider the representativeness of the measurement locations for the noise component we discuss the model for the spatial correlation structure of the white noise. We calculated the correlations for each pair of noise series and plotted these correlations against the distance between the corresponding measurement locations (fig. 3a).



a) b)

Figure 3. Correlation between noise series versus distance (a) and the modified noise series versus distance (b).

From figure 3a it can be seen that the average correlation is around 0.6. Although the higher values occur at shorter distances there is no clear relationship between distance and correlation. The relative high value for the average correlation indicates that at a certain point in time the realisation of the White noise process everywhere in space tend to be either higher or lower than the time average. In other words if at a point in time the realization of the white noise series for one particular location is higher than the time average of that series there is a large probability that, at the same point is time, everywhere in the area the realization of the white noise process is higher than the corresponding time average. Unlike the temporal variation, the spatial variation does not show a clear spatial pattern.
Therefore we adopted the following noise model:

$$a_{i,t} = \bar{a}_t + \hat{a}_{i,t} \qquad (7)$$

where $\bar{a}_t$ is the spatial average of the noise at time t
$\hat{a}_{i,t}$ is a spatially and temporally independent noise term, which will be referred to as the modified noise term

At each time step the spatial average of the noise can be calculated from the measurements. Therefore it can be considered to be a known input.
The time variance of the spatial average of the noise is 62.75 cm^2. The correlations between modified noise series at the measurement locations as a function of the distance between those locations are given in figure 3b. This figure shows that the modified noise is spatially independent. We interpreted the modified noise variances using Kriging interpolation. The variogram and the interpolated variances of the modified noise show similar patterns as in figures 1e and 2e.

The range of the parameter of the noise model ϕ (fig. 1c) and the noise variance (fig. 1e) is small. These ranges are in the order of 1 km. The dominant process scales are much smaller than those of the response to the precipitation excess. A spatial interpolation of the noise model is only meaningful if the measurement scale is smaller than the process scale, and hence only in subareas where the distance between measurement locations is smaller than 1 km an interpolation of the noise model can be made. In areas where the measurement scale is larger than 1 km only the overall characteristics of the noise component can be estimated.

CONCLUSIONS

In the application the groundwater head was split into a precipitation excess component and a noise component. The dominant spatial process scales related to these two components are different. The precipitation excess component shows a larger spatial correlation than the noise component.

For the response to the precipitation excess a meaningful interpolation in space can be made with the existing network. The representativeness of measurements for interpolation of the response of the groundwater head to precipitation excess is in the order of 4 km. In other words one single measurement contains information for the interpolation of this response up to a distance of max. 4 km. If the measurement scale is larger only the overall spatial characteristics (like mean and variance) of the response can be given.

The noise component consists of two parts. This results in a non zero spatial average value for the white noise at each time step. It should be noted that the time average of the white noise is zero. At each measurement point in time the characteristics (spatial average and variance) can be calculated from the measurements.

For the second part of the process scale is smaller than the measurement scale. Therefore only the general characteristics (variance) can be given. However, the measurement series don't contain information for non measured locations at individual points in time.

The interpretation of the noise component from hydrological point of view is that at a particular point in time "random" variations around the response to the precipitation excess tend to have the same direction (higher or lower) for the entire area. Individual measurements are representative to estimate the spatial average of these random variations in time. However the spatial variation at a particular point in time have such a small spatial correlation that the measurement points contain no information about the spatial variation at a point in time for non-measured locations. The representativeness of the measurements for these spatial variations is much smaller than the distance between the distance between the measurement locations.

The general conclusion of this study is that representativeness of groundwater head measurements is not a fixed characteristic of a location, but it is variable according to the technical objective. Therefore for monitoring network design the spatial representativeness should be analyzed for each technical objective separately.

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WATER QUALITY MONITORING IN THE DANUBE RIVER BASIN

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ABSTRACT

The national water quality monitoring programmes in the Danubian countries has provided a vast amount of data on water quality, but comparison of the results has been difficult due to differences in sampling and analytical techniques, and the lack of adequate interlaboratory comparison procedures. Internationally co-ordinated water quality surveys in the Danube river basin started in 1988 as a result of the Bucharest Declaration. The Environmental Programme for the Danube River Basin (EPDRB) provides a framework to extend and upgrade the monitoring programme. The importance of: (a) harmonization of sampling and analytical methodologies to be used for obtaining comparable results by the different laboratories, and (b) quality assurance and analytical quality control has been recognized and developed. Special attention has been paid to the comparability of the results of nutrient analysis and the QualcoDanube performance testing exercises showed significant improvement. The "data rich - information poor syndrome" (DRIPS) is planned to be changed to "less data - more information" (LDMI) by developing new analytical techniques, such as for characterising the bioavailable, mobile fraction of sediment-associated pollutants.

BACKGROUND INFORMATION

Before 1985, water quality monitoring in the Danube river basin had been carried out independently in the different countries, in some cases as part of bilateral agreements. In 1985, the Bucharest Declaration was the first sign of a basin-wide international cooperation [1985]. Since that time several harmonized sample collections have been carried out in different projects covering the whole length, e.g., the Cousteau survey in 1991-92 [Costeau Equipe, 1993], or selected reaches, e.g., Austrian-Slovakian-Hungarian studies [3], of the Danube river. The survey results revealed the presence of pollutant, "hot-spots" along the Danube. The major pollutants, in addition to the inorganic nutrients, were found to be heavy metals and petroleum compounds. On the basis of several studies harmonization of the micropollutant monitoring for the Danube rive was recommended [Literathy Slaszlo, 1995].

The gaps in existing knowledge and the problems of the comparability of the monitoring results have been recognised. The Environmental Programme for the Danube River Basin (EPDRB), started in 1991, provides a framework to extend and upgrade the monitoring programme. One of the major tasks of the EPDRB was to set-up and operate a Trans-National Monitoring Network (TNMN, 1993) using accepted methodologies and appropriate quality control. In the framewok of the EPDRB, the mission statement of the Monitoring, Laboratory and Information Management Sub-Group (MLIM-SG) included the harmonization of the sampling and analytical methods for use in the TNMN. The comparability of the analytical results obtained in the different countries is ensured by quality control measures. The Laboratory Management Working Group (LMWG) of MLIM-SG is responsible for the harmonization of the analytical work and quality assurance for the TNMN along with the participation of Reference Laboratories designated in each country.

THE TRANS-NATIONAL MONITORING NETWORK

PARAMETERS TO BE MONITORIED

Selection of the pollutants as target compounds for pollution monitoring requires: (1) pollutant inventories, (2) water quality guidelines, criteria for healthy aquatic life and intended water uses, (3) results of preliminary surveys to identify potential polluting compounds, and (4) identification of unrecognised pollutants. Determinands to be monitoried in the water column and in the bottom sediment during the first phase of the Danube TNMN have been selected on the basis of (1) to (3) and are summarized in Table 1. It is planned that determinands for the pollution monitoring will be revised time-to-time on the basis of (4) and the improvement of the analytical procedures.

Determinands	Water Column	Bottom Sediment
1. General, physical and chemical	Temperature DO, pH, Conductivity Alkalinity Suspended solids	Particle size distribution Carbonates
2. Nutrients	NH_4-N, NO_2-N, NO_3-N Kjeldahl-, or Total-N PO_4-P, Total-P	 Kjeldahl-, or Total-N Total-P
3. Inorganic - Major ions - Elements	 Na, K, Ca, Mg Chloride, Sulphate Fe, Mn, Al Hg, Cd, Pb, Zn, Cu, Ni, Cr	 Fe, Mn, Al Hg, Cd, Pb, Zn, Cu, Ni, Cr
4. Organic - Non-specific, sum-parameters - Specific	 BOD_5, COD_{Cr}, COD_{Mn} DOC, AOX Phenol-index Anionactive Surfactants Petroleum Hydrocarbons Lindane, DDTs, Atrazine $CHCl_3$, CCl_4 C_2HCl_3, C_2Cl_4	 TOC, EOX Total Extractable Matter Petroleum Hydrocarbons Lindane, DDTs PAHs (Borneff 6) I PCBs (7 congeners)
5. Radioactivity	*Total-ß activity*	Caesium-137
6. Biological	Chlorophyll-a Phyto- and Zooplankton Macrophytes, Fish Saprobic/Biotic Index	 Macrozoobenthos Microzoobenthos Phytobenthon
7. Microbiological	Total and Faecal coliforms Faecal streptococci Salmonella	Clostridium

Table 1: Determinands to be monitored in the Danube TNMN - Phase 1.

MATRICES TO BE MONITORED

It is important to emphasize that different polluting compounds behave differently in the aquatic environment, where they are distributed in the abiotic and biotic compartments, or between the dissolved and solid phases within the abiotic compartment. Depending on their abundance in the different matrices, the sample collection should be extended to those matrices where the pollutant concentration levels are expected to be significant. In the case of toxic, persistent compounds, sediment is considered as an important compartment where these pollutants can accumulate.

Comparibility of the analysis of the sediment-associated pollutants in the suspended and bottom sediment can be achived by analysing the same grainsize fraction in both, suspended and bottom sediment. Due to dependence of the sedimentation rates on the particle size it is not likely that suspended solids will contain particles larger than 200 µm, but will include the clay particles, which are unlikely to settle to the bottom in a turbulent water body. It has been agreed that the less than 63 µm grain-size fraction of the bottom sediment will be used for pollution monitoring during Phase 1. of the TNMN. This grain-size fraction is obtained by wet sieving.

ANALYTICAL METHODOLOGIES

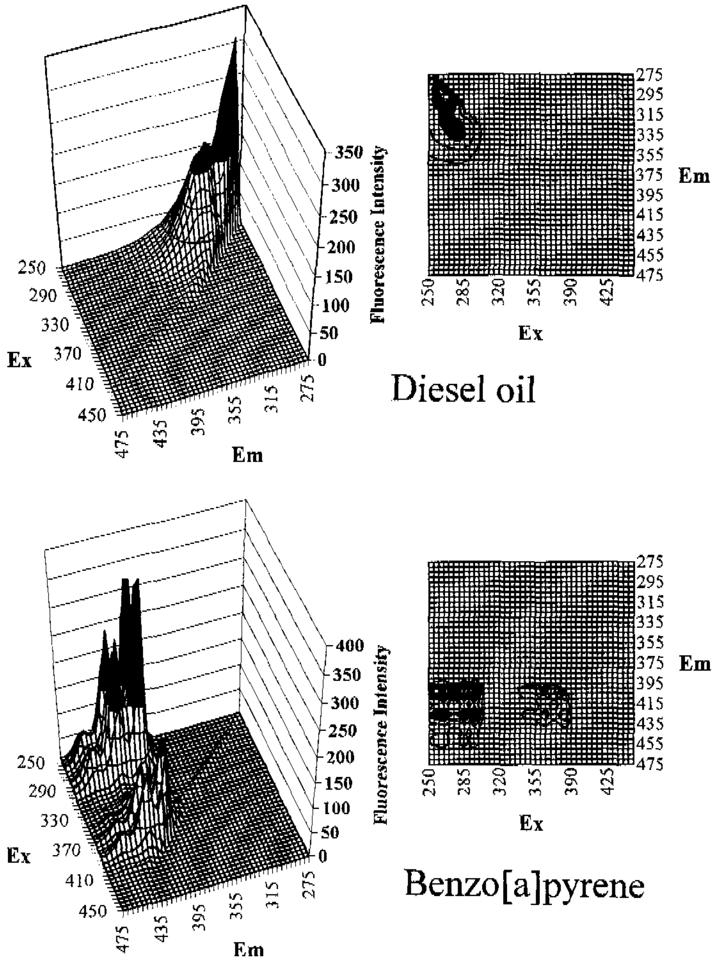
Internationally accepted analytical methods, e.g., ISO, are available for most of the determinands which should be monitored in the TNMN. Analytical problems exist in the case of determination of sediment-associated pollutants in general and of the oil (petroleum) pollutants in particular.

Characterisation of oil pollution is particularly important in the Danube basin because the largest amount of organic pollution in the catchment relates to petroleum products. Sources of the oil pollution includes refinery wastes, transportation of petroleum and petroleum products by vessels, pipelines, municipal wates, etc. Determination of the petroleum compounds is a problem for environmental analysts. This is because petroleum, as well as its refined products, is a complex mixture of different compounds, hydrocarbons and oxygen-, nitrogen-, sulphur- and metal-containing heteromolecules. Complexity of sources, processes and degradation mechanisms must be considered in order to chose the appropriate analytical approach for quality/pollution monitoring purposes. There is no single analytical method that can be use to characterise all petroleum components, or petroleum-related pollution. Selection of a particular analytical method is always a compromise between the feasibility of the analysis, e.g., instrumentation and available resources, and the degree of chemical detail, selectivity, sensivity and accuracy. The proposed approach for the Phase 1. of the TNMN includes screening tests using spectrophotometric techniques, and specific methods (e.g., chromatographic for PAHs). The spectrophotometric method selected is based on UV absorption measurement which allows detection of unsaturated, aromatic compounds and is calibrated to a Danube Reference Oil (DRO) standard. It would be desirable in the future to extend this measurement to fluorescence spectrophotometry because of its high sensivitity and of the qualitative information which could be achieved. Fig. 1. shows the 3-D and contour fluorescence diagrams of diesel oil and the benzo(a)pyrene; and Fig. 2. shows the contour diagram of two Danube sediment extracts diluted to the same UV absorption level in cyclohexane. The diagrams clearly show the qualitative difference between the extracted compounds.

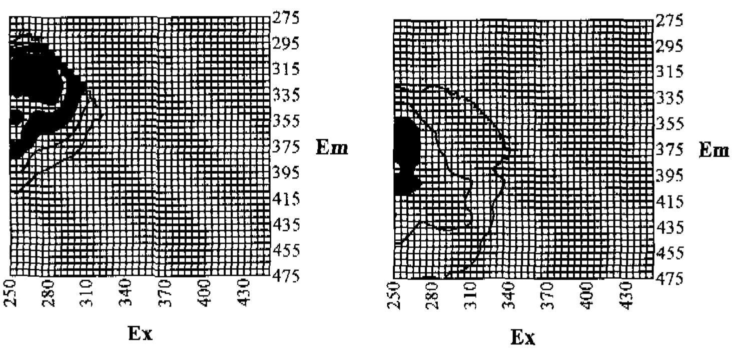
QUALITY ASSURANCE

Intra- and interlaboratory quality control has been used during the implementation of the TNMN which has been discussed and agreed in the LMWG. In addition to the preparation of Standard Operation al Procedures (SOPs), the approved MLIM-SG work programme has included recommendations for similar laboratory facilities; the provision of necessary analytical instrumentation in the laboratories (as a minimum for the National Reference Laboratories); the implementation of an integrated training programme; and the importance of proficiency testing carried out in interlaboratory comparison studies.

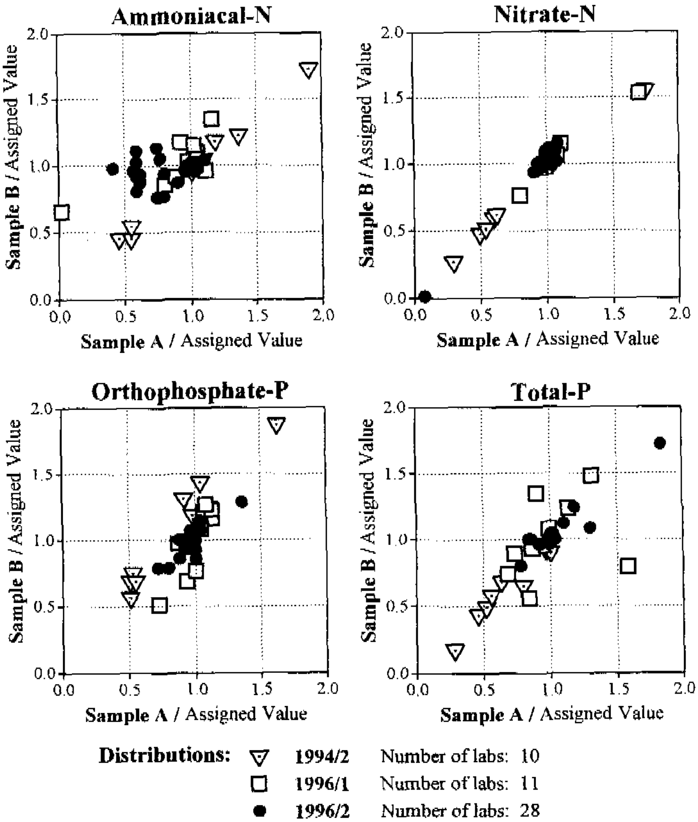
Such comparative studies included QualcoDanube, Aquacheck, IMEP-6 and EQUATE sample distributions. The QualcoDanube intercalibration programme started in 1993 with the participation of the laboratories involved in the Danube water quality monitoring in the framework of the Bucharest Declaration. In 1995, it was extended to the 11 National Reference Laboratories and in 1996 to another 18 national laboratories within the Danube river basin implementing the TNMN. After the third year the intercalibration results show significant improvement, as it is



Figuur 1: Fluorescence 3-D plot and contour diagram of diesel oil and benzo[a]pyrene, 1 mg/ml and 0.1 µg/ml in cyclohexane, respectively.



Figuur 2: *Fluorescence contour diagrams of Danube sediment extracts with different character at the same concentration with UV absorbance.*



Figuur 3: Quality improvement in nutrient analysis during the QualcoDanube intercalibration exercises.

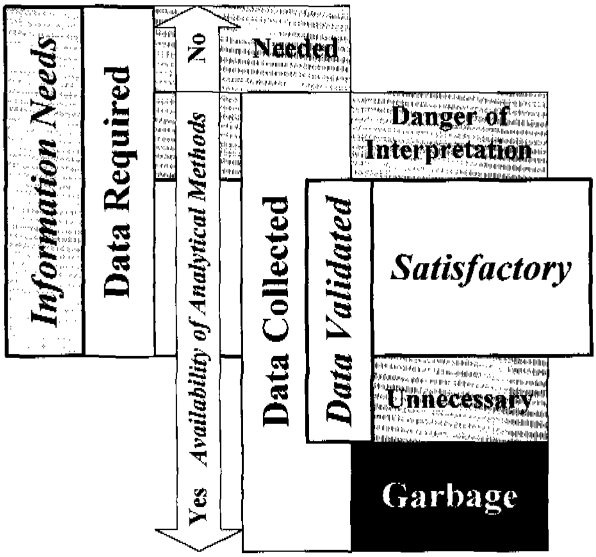
demonstrated in Fig. 3. (QualcoDanube data) [Vituki, 1996]. It is expected that the regularly organised intercalibration exercises will ensure adequate comparability of the water quality monitoring results in the Danube river basin in the future.

NEED FOR IMPROVEMENT: LESS DATA-MORE INFORMATION

The "data rich - information poor" monitoring usually involves determinands, target compounds, which are (1) from biogeochemical origin and controlled by natural processes, (2) known pollutants from industry, agriculture, household, or traffic/transportation, and (3) "easy to determine" - why not? In addition to the lack of needed data, the most important danger is the interpretation of the unvalidated data. This is demonstrated in Fig. 4.

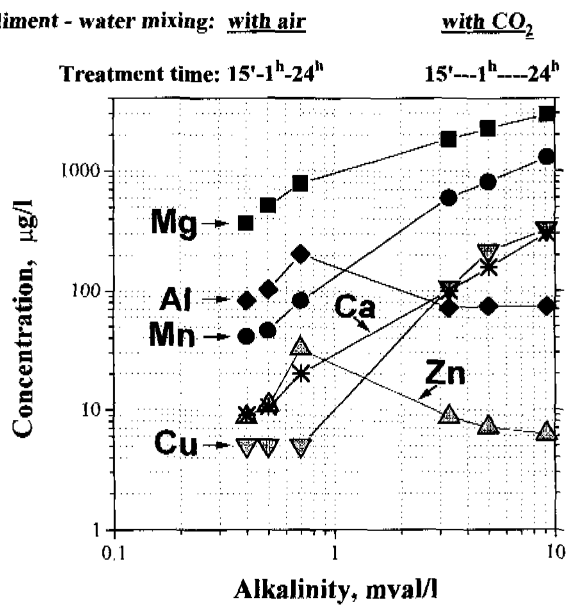
Because even a very simple determination is money and time consuming it is important to limit our efforts in monitoring to determinands and characteristics that provide significant information. This is needed to achieve a "less data - more information" monitoring. It is important to consider that the physical state and the chemical structure of the pollutants could be altered in the environment which affects their behaviour. On this basis, the pollutants can be categorised as follows:

a. primary polluting compounds which are emitted and discharged into the environment where they could be: (1) degraded into harmless end products, (2) transformed into persistant pollutants, or (3) persistent during the weathering processes;



Figuur 4: Data rich - information poor syndrome.

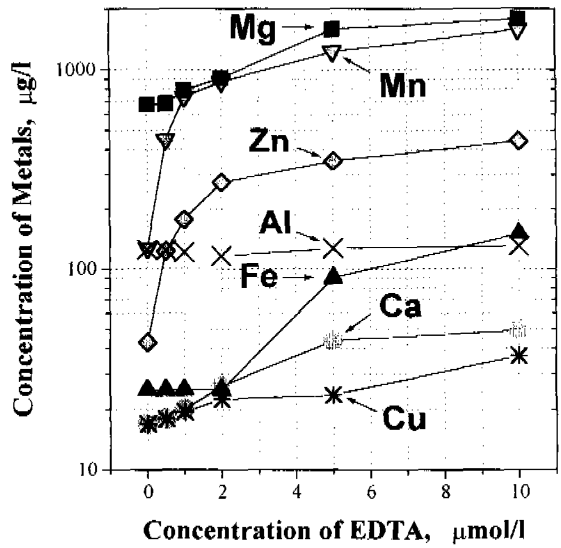
b. secondary polluting compounds that are breakdown and/or conversion products, which are produced during the environmental weathering of the primary polluting compounds. In most of the cases they are: (1) polar, water-soluble compounds which can easily migrate through bankside filtration, (2) compounds that might be more toxic than their parents, (3) compounds that may be at least temporarily resistant to further degradation, and (4) compounds that can affect the fate of other pollutants; and



Figuur 5: Leaching of metals from sediment during interaction with carbon dioxide.

c. tertiary polluting compounds such as primary and secondary pollutants and naturally occuring compounds which are present as particulate matter, accumulated and buried in the bottom sediment in inactive form, but that could be mobilised by primary and secondary polluting compounds, or by changes in environmental conditions.

Most of the monitoring programmes consider the primary pollutants as determinands. More effort is needed to achieve monitoring of the secondary pollutants and characterise the tertiary pollutants. In the case of heavy metals, for example, the total concentration is usually measured, however, the adverse effect mainly relates to the ionic (dissolved) fraction. The analytical problems related to the heavy metals include: (1) most of the heavy metals occur naturally and their background levels should be established, (2) they are associated mainly with particulate matter and their concentration depends on the grain-size and diluting minerals; and (3) their speciation may vary from place-to-place. The form of the anthropogenic input, and the pollution fate processes in the aquatic ecosystem will control their bioavailability and any harmful effects. Sample treatment techniques should be developed to characterize the fraction of the sediment associated pollutants, e.g., heavy metals, phosphates, and other compounds most liable for mobilization



Figuur 6: Leaching of metals from sediment during interaction with EDTA.

in the aquatic environment. The driving force for mobilization could be (a) the carbon dioxide evolved during microbial degradation of organic matter, (b) decrease of the redox-potential, and (c) presence of complexing agents either from pollution input, e.g., EDTA, NTA as primary pollutants, or from biological and photochemical reactions as breakdown products (secondary pollutants). Fig. 5. demonstrates the subsequent leaching of selected metals from sediment by carbon dioxide. The variation in alkalinity reflects the amount of hydrocarbonate as a product of the interaction between the sediment and carbon dioxide. Fig. 6. shows the leaching of metals with EDTA at different concentration levels. Both Figures demonstrate that the rate of leaching is different in the case of the different metals, and also varies with the amount or concentration, of the leaching agent. It might be feasible in the future to adopt a method which shows the mobilised fraction of the pollutants by a minute amount of leaching agents, which could be adjusted to their realistic concentration in the aquatic environment.

CONCLUSIONS

The solution of water pollution problems, and restoration of a healthy aquatic life in Danube-Black Sea system, *calls for international collaboration by all the countries in the catchment areas* of the River Danube and its major tributaries. This collaboration has been a key feature within the Bucharest Declaration, the Environmental Programme for the Danube River Basin, the Black Sea Convention [1994], and the Danube Convention, which is signed in 1994 and now under ratification. Implementation of these programmes requires reliable pollution assesment, identification of major pollutants, knowledge on their degradation fates and effects. The agreed TNMN programme, which includes chemical and biological monitoring, is the first step to achieve these goals by ensuring agreed sampling and analytical approaches and quality control. Significant quality improvement has been achieved to date as a result of the interlaboratory comparison exercises described in this paper. Improvement in the analytical techniques, and particularly in the sample preparation procedures, is needed in the future to obtain information rich monitoring in the Danube river basin at acceptable costs.

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REMSSBOT, A FACILITATING, TAILOR MADE ASSESSMENT TOOL TO IMPROVE ENVIRONMENTAL MANAGEMENT.

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ABSTRACT

The physical and administrative areas related to environmental problems may vary from a part of a region up to several countries. Challenging these problems, REMSSBOT (Regional Environmental Management Support Based on Telematics) will benefit from access to different information sources and aims at improving environmental information services. The innovative aspect of the Remssbot system is that administrations and environmental managers have access to each others data on an independent base while maintaining control over their own data. This concept results in several connected databases which are physically separated, but appear as one. The central element is a catalogue conform the guidelines of the Catalogue of Data Sources (CDS) of the European Environmental Agency. The system consists of different applications in order to use the information, these are called the demonstrator. The demonstrator is implemented at the participating administrations. Remssbot is supported by the European Commission within the framework of the Telematics Application Programme (DG XIII). Public administrations of the regions Piemonte (Italy), Attica (Greece) and Scheldt (Netherlands & Belgium) participate in this project. This paper briefly describes the general project and goes into detail on the demonstrator for the river Scheldt.

INTRODUCTION

Many local and regional environmental management boards, like those in the regions co-operating in Remssbot, today are faced with the challenges offered by the change from management by topic and zone to the integrated management of larger areas or even cross border management. Virtually all public administrations have some form of information technology (IT) systems supporting their activities, but these systems are mostly designed along the paradigms valid during the conception of these systems. This resulted in incompatible systems and in many cases incompatible information for each environmental topic as well as among management boards in different regions for one environmental topic.

The obvious way to overcome these obstacles would be to design and implement a new IT system which satisfies all user requirements in all local sites. Next to the complexity and costs involved in such an effort it would imply postponing all improvements in environmental management while the new system would be designed and implemented.

The Remssbot system seeks for an alternative method of helping the local and regional bodies with implementing integrated management by deploying a telematics solution. This solution, based on a catalogue conform to the guidelines of the Catalogue of Data Sources (CDS) of the European Environmental Agency, describes what information is available at what location and provides IT systems with the automated procedures to access the information. The supporting tools and building blocks allow users to navigate through the catalogue and explore information sources regardless of environmental topic and location.

The main goal of Remssbot is to contribute towards a better integrated environmental management in the future.
In the **Scheldt sub-project**, the ecological functioning of the river system, using the holistic river basin development approach is the main issue. This implies integrated planning for all essential elements in the entire river basin before implementing individual projects.

MAIN OBJECTIVES OF REMSSBOT:

- Stimulating integrated environmental management development as the central basis of policy plans, using natural borders instead of administrative determined frontiers.
- Sharing experiences, knowledge and measured data on environmental topics between administrations and their water managers concerned with an ecosystem. This can result in closer co-operation and understanding of public administrations concerning environmental information.
- Developing an environmental management support system which can be used for a wide variety of environmental topics and management levels.
- Facilitate the search for and access to specific environmental data by several well developed navigation systems.
- Increasing the velocity and quantity of environmental information available, through the use of Remssbot.
- Capacitating for the implementation of different types of integrated environmental information management through the use of a catalogue of data sources.
- Widening dissemination of all kind of public available environmental information.

PLANNING ACTIVITIES AND IMPLEMENTATION

The Remssbot project started January 1996 by making an inventory of the needs and requirements of the users of the demonstrators in the participating regions. One of the conditions of the European Commission is that a Remssbot system is operational before the end of 1997. A prototype of the system, a so-called 'Demonstrator' will be realised during the second half of 1996 and beginning of 1997. An upgrade of this demonstrator will be realised through verification and evaluation will be realised during the second half of 1997. In these processes the main activities focus on the interactions between users and technical specialists. The technical specialists train and support the users, while the user will give comments and advises to improve the functionality of the system. During the demonstration period (January 1997 - December 1997) the data generated through Remssbot in the Scheldt region focuses on water quality of the river Scheldt. The system developed will be able to handle a broader scope and quantity of data. However, it is not possible to realise this within the two years of construction and optimisation of the system.

REGIONAL FIELDS OF APPLICATION

Three regional administrations, each having different types of environmental problems, participate in the Remssbot project. These regions are:

- The region **Piemonte** in Italy. Here public officers of administrations involved in the management of the administrative and technical procedures of industrial plants will be supported.

These plants have information concerning the risk of accident, generated by specific industrial activities characterised by the storage and/or the processing of relevant quantities of hazardous substances,;

- The region **Attica** in Greece. In order to support the local environmental managers, several measurement points in the regional area will be integrated into a system to show conditions on air quality, solid waste disposal and bathing waters.
- The **Scheldt** region in the Netherlands and Flanders. In this region the Dutch and Flemish administrations related to water management in the Scheldt river, will be informed about the ecological functioning of the river system. The Scheldt region facilitates the integrated water management development, in which the river system is central, instead of manmade frontiers.

THE SCHELDT APPLICATION OF REMSSBOT

ENVIRONMENTAL PROBLEMS IN THE SCHELDT REGION

The source of the Scheldt river is situated in North-France, nearby Gouy-le-Catelet, just north of Saint-Quentin. The river, which is about 350 km long, flows through France, Wallonia, Flanders and the Netherlands and reaches the North Sea between Vlissingen and Breskens (Brabander, 1995). The estuarine part is situated downstream of Gent (length: 160 km); this part consists of a fresh water, a brackish and a salt water zone. The total surface of the Scheldt catchment is about 23.000 km2 divided among France (33%), Wallonia (19%), Brussels (1%), Flanders (43%) and the Netherlands (4%) (Santbergen, 1994) (see Figure 1). About 11 million people live in the catchment area.

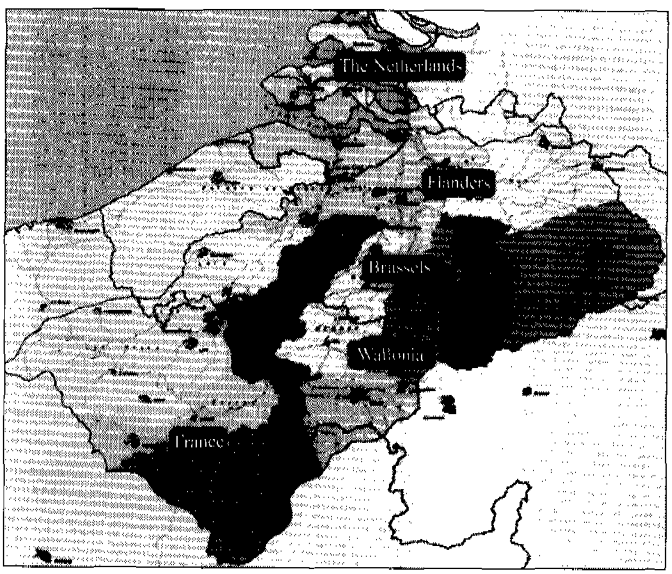

Figure 1: The Scheldt Catchment Area.

The river Scheldt is of great importance for a wide variety of uses like fishing, agriculture, shipping purposes, industry, drinking water purposes and recreation. For centuries these activities have benefitted from the river basin. The last decades however development of economical inte-

rests conflicts more and more with the ecological functioning of the system.
The high number of habitants, the high degree of industrialisation and the agricultural use of a large part of the area resulted in considerable pressure on the river ecosystem.
The ecological problems in the Scheldt basin are due partly to the enclosure of saltmarshes in agricultural use and more recently for a wider variety of industrial and port related, urban, safety and recreational purposes and the remaining saltmarshes decrease in area due to erosion processes (Meire, 1993). As a result, the river Scheldt is one of the most polluted river systems in Western Europe.

The river has its own specific problems:

- Pollution of surface water and water sediments. Generally, from source to mouth, the emphasis of the problems shifts from the surface water to the bottom sediment.

- In France and Belgium freshwater is diverted into canals, especially in dry periods. A yearly average of 65% of the freshwater from the Scheldt catchment area upstream of Gent is diverted to the North Sea, without reaching the estuary. Consequently, the estuary is threatened by salinisation. If in future more freshwater will be diverted, salt North Sea water will come further inland and valuable brackish and freshwater intertidal areas will disappear (International Scheldt Group, 1994; Holland and Smit, 1994).

- Men rebuild the river system by amputating river valleys and river banks. Saltmarshes and adjacent mud flats were enclosed mainly for agricultural use and more recently for a wider variety of industrial and port related, urban, safety and recreational purposes (Meire 1993). Gradual changes from water to land are disturbed by infrastructural works like for example dikes. As a result, man limits space for natural processes to take place.

- In the long term, estuary are doomed to fill up because of the inland transportation and sedimentation of sand and mud from the sea on the one hand, and downstream transportation and sedimentation of suspended matter by the river itself on the other hand. In the river Scheldt estuary, this process is accelerated by human activities as enclosures and embankments.

- Habitat degradation occurs in terms of size reduction and loss of diversity. The quality of habitats in the river Scheldt catchment area is endangered by pollution caused by industrial, agricultural and domestic waste, resulting in the disappearance of complete fish populations (Turkstra et al., 1995). In the Scheldt estuary, there is a shortage of freshwater and saltwater marshes, brackish mud flats and shallow water area.

INTEGRATED WATER MANAGEMENT APPROACH

During the last decades it became more clear that changes designed to serve sectoral interests have led to river's degradation.
Fortunately, there is a growing consciousness that the quality and the functioning of river systems is the joint responsibility of riparian states that share these systems. The idea of an integrated water system approach in order to make policy going beyond administrative boarders is growing. In the Scheldt sub-project, the river basin approach for integrated river basin management, is the basic concept.
The most important element in integrated water management is: **The water system is the centre.**
A water system is a geographically defined area, a hydrological unit containing all the necessary elements for the proper functioning of the system. This includes the ground and surface water, the soil, the banks or shores as well as the surrounding land and all factors related to the proper functioning of the whole river basin.

AN EXAMPLE HOW REMSSBOT CAN CONTRIBUTE TO INTEGRATED WATER MANAGEMENT:

The way in which Remssbot contributes to integrated water management in the Scheldt region could be for example: A water manager needs information about the concentrations of nitrogen for July 1996 in the Scheldt river. Remssbot, using the CDS, offers the possibility to select the subject (nitrogen), the place (the river Scheldt) and the time (July 1996). The result will be one map, table (or documents if available) with information from several databases.
The applications developed in the demonstrator will explain (see next pages) how the information is gathered and what the scope of the information is.

INTERNATIONAL POLICY

In the context of an integrated approach towards the problems in the water system, the International Commission on the Protection of the river Schelde (ICBS/CIPE) was initiated. France, Wallonia, Brussels and the Netherlands have signed a new water treaty for the river Scheldt in April 1994, followed by Flanders in January 1995.
The Commission and its working forces are in function since June 1995. This installation will give a strong impulse to the development of a common policy on water quality and ecology in the Scheldt catchment area.
An important task will be the improvement of communication among the participating states, thus avoiding misunderstanding and unnecessary disagreements (Zijlmans, 1995). In this context Remssbot can be an important tool and a step forward in the realisation of integrated water management. Although at this moment the ICBS/CIPE is not directly involved in the Remssbot project, the progress is followed by the member parties.

The organisations involved directly in the Remssbot project in the Scheldt region:

- Directorate-General for Public Works and Water Management, Zeeland Division, (The Netherlands),
- The Flemish Environmental Agency (Flanders),
- The National Institute for Coastal and Marine Management (The Netherlands),
- The National Institute for Inland Water Management and Wastewater Treatment (The Netherlands),
- Electronic Data Systems (EDS) International B.V. (Brussels and The Netherlands).

THE DEMONSTRATOR

As mentioned before, a prototype of the system, the so-called 'demonstrator' is realised during the second half of 1996 and beginning of 1997. The demonstrator is the first realisation of the Remssbot system at different locations using all components serving the water manager.
The Catalogue of Data Sources (CDS) is a list of available data from the different providers participating in Remssbot. These data will be used by the water managers, using several applications at the administration locations. The applications together with the CDS and the connected databases form the Remssbot-demonstrator. The water managers at different levels or locations can use one specific application more than another.

Within the **demonstrator** realised at different places, the next applications will be realised.
1. A **search information system** (SIS) application will give easy search and access facilities to

all kind of data and from different sources relevant for water policy. The information (wether documents, maps or data) is only available for professionals working at the participating administrations.

2. A **GIS application** (Schelde GIS), which will be developed, is mainly based on information about the most recent measured parameters from the different administrations. These data will be delivered by the different administrations co-operating in the project.
3. The Flemish Environmental Agency and the Scheldt Information Centre (SIC, formed by various parties, among them the Directorate-General for Public Works and Water Management, the National Institute for Coastal and Marine Management and the Province of Zeeland) will use Remssbot to connect their **WWW server.** This server informs all kind of organisations and public places (libraries, universities etc.) about several activities and regional projects of the administrations in the Scheldt region.

DIFFERENT LEVELS OF INFORMATION GATHERING

Next to the differences between the fields of application among the three regions, differences at the level of environmental management in which Remssbot functions occur.
The contents of the data made available by Remssbot have a very wide range. An policy maker may be interested in aggregated data in order to better interpret an investigation, a researcher can compare some investigations carried out by his colleagues, while a measuring service can compare equipment, methods and accuracy with their colleagues at the other side of the boarder. An important international body like the International Commission on the protection of the river Scheldt can use Remssbot to share their (unfinished) documents, minutes, appointments. This is realised by restricting access possibilities. The contents of the information varies from quite raw measurement values, experiment and investigation reports, up to policy papers.
The different users require this broad scope of contents , making clear that Remssbot can be applied to several levels of environmental management.
Different users have different interests, but although the type of data is varying highly, the concept does not change. This is the concept of what information is available on an item, where is the information and how does it become available.

TECHNICAL CONCEPT OF REMSSBOT

The main objective of Remssbot is sharing environmental information, not by building a centralised data warehouse, but by keeping the data at its original location and connecting these separate databases. This means Remssbot is in fact a logical data warehouse, but physically the data location remains as it is.

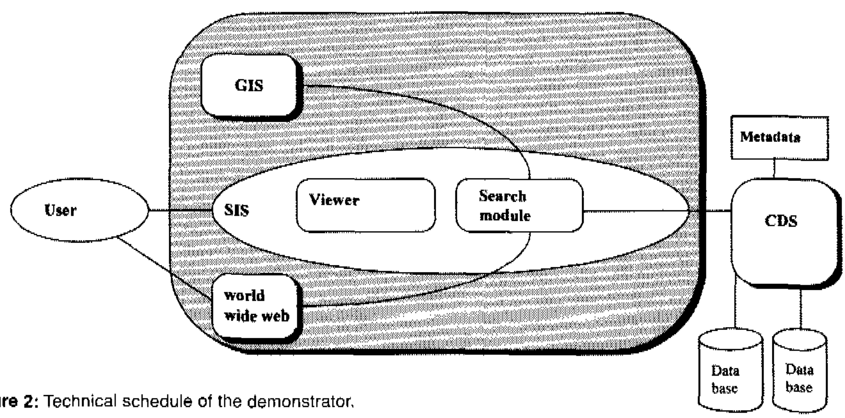

Figure 2: Technical schedule of the demonstrator.

The user should be able to identify what information is available and retrieve the information. There should be a transparent access to the information source at the regional data provider's location. Information may even need to be accessed in parallel from several information sources to serve the user's request. The user can find out where the information comes from, but it is not necessary to know in order to access the information.

The Remssbot system is based on a catalogue conform to the guidelines of the **Catalogue of Data Sources** (CDS) of the **European Environmental Agency** (EEA). The CDS comprises what information is available at what location and provides IT systems with the automated procedures to get access to actual information.

The system allows users to navigate through the catalogue and explore information sources regardless of environmental topic and location. The CDS provides an overview of all available data about the chosen subject (tables, documents, maps etc.), the so-called *metadata*. Access to the metadata is realised by means of keywords, a tree search system and a zooming system, based on geographic co-ordinates.
There should be an easy integration of the information, meaning that whenever possible, information should be presented in format that can be understood by the user. Especially when the information comes from different data providers, conversions to a standardised format may be needed.
In the first prototype of the demonstrator, the data providers will be the databases from the Dutch and Flemish water management administrations.
The data providers should not be hindered by Remssbot applications to maintain, enhance or improve their current systems and databases. The Remssbot solution should allow the data providers to change their data structure without having to co-ordinate closely with the Remssbot users. Any type of change should be allowed as long as the request for a Remssbot service can still be answered transparently for the application program.

CONCLUSIONS

The objective of Remssbot for the administrations and water managers co-operating in the Scheldt region project is to have better access to their own and each others data, on an independent base without losing the authority and control over their own data. This results in creating connected databases which are physically separated, but appear as one.

- The disposal of digital information and metadata can facilitate searching for documents and data using keywords, co-ordinates or a tree search system, even when this is within the department or organisation itself.

- The data level within Remssbot has a wide range: it varies from rude measurement values, experiment and investigation reports, up to policy papers.

- The environmental issues determining the contents of the system is varying within the three regional sub-projects. The users require this broad scope, in order to be able to apply Remssbot on several levels and topics of environmental management.

- The users have different interests, but although the type of data varies enormously, the concept remains the same. The concept embodies what data are available on an item, where are they and how do they become available.

The fact that Remssbot has a broad scope of contents and users is a challenge that can demonstrate the strength of the concept used by Remssbot. Having better knowledge of each others activities, such as investigations, measured values and methods used, can be facilitated using the Remssbot system. This creates more understanding for each others policy, prevents double investigations of the same problem and provides an accurate overview of water quality parameters in the river Scheldt catchment area.

The demonstrator can facilitate integrated water management development which places the river ecosystem as central basis of policy plans, instead of administrative manmade boundaries, such as state provincial frontiers.

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TAILORING OF A GROUNDWATER QUALITY MONITORING NETWORK FOR A WATER SUPPLY COMPANY

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ABSTRACT

In the Netherlands, groundwater is an important source for the production of drinking-water. Until recently WOB, the water supply company for the eastern part of the Province of Noord-Brabant, *has relied only upon this source. In the past groundwater has shown to be a very pure and reliable source, especially when it is extracted from confined aquifers. Unfortunately pollution has had its influence on the quality of unconfined groundwater. Pesticides, heavy metals and high concentrations of nitrate and sulphate are detected in the extraction wells in the groundwater protection area of* Vierlingsbeek. *With the use of a groundwater quality network the changes of the quality of the extracted water are monitored so if necessary corrective measures can be taken in time.*

INTRODUCTION

WOB is a regional water supply company, situated in the south eastern part of the Netherlands. Our company exploits twenty pumping stations at which groundwater is purified to drinking-water, which is distributed to over one million consumers in more than 450 thousand house-

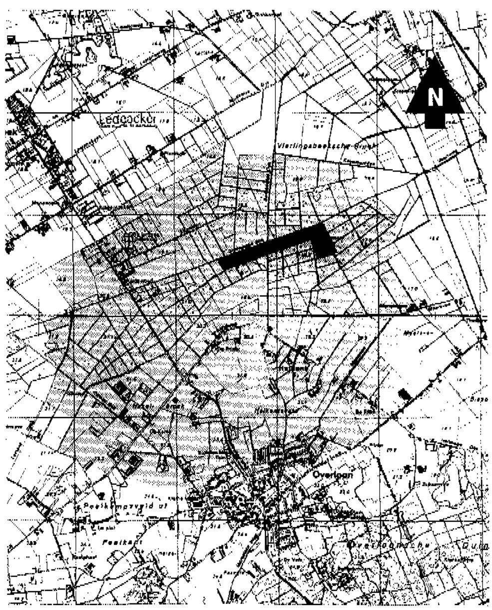

Figure 1: Groundwater protection area Vierlingsbeek

holds. In this region agriculture is very intensive, especially cattle breading, which reflects on the quality of unconfined groundwater. Pesticides used for crop protection and minerals from the large amounts of manure used for growth stimulation leach to the groundwater.
In the early eighties the concern for negative influences of pollutants on the environment increased. Groundwater, which used to be considered as the best possible source for drinking-water production, is now recognized to be under pressure from human activity. Ten of the groundwater extraction areas surrounding our pumping stations are vulnerable to contamination. An extraction area is considered to be vulnerable if a conservative and mobile pollutant at the surface can reach the wells in 25 years or less as predicted with models. The Provincial Government of *Noord-Brabant* has designated these vulnerable extraction sites as groundwater protection areas, like *Vierlingsbeek* (figure 1), in the Provincial Environmental Act. The act restricts and regulates activities which might endanger the groundwater quality.

NETWORK LEVELS

A quality monitoring network surrounding a groundwater extraction site consists of different levels, each giving another type of information. The quality measurements of the produced drinking-water, obligated by the Dutch Drinking-water Act, are considered to be the first level of the monitoring network. The Dutch Drinking-water Act also obliges monitoring of the extracted groundwater at drinking-water production sites, which can be seen as the second level of the network. The tertiary level is the actual groundwater quality monitoring network. For this level of the quality network special monitoring wells are installed and sampled throughout the extraction area.
Based on combination of the information from three network levels, predictions can be made on the quality changes that are to be expected in the near future. In this paper we will focus on the third level of the monitoring network, without ignoring the information produced by the first and second level.
The results from the chemical analysis of the water samples of the three network levels are stored in a groundwater database developed by IWACO Rotterdam. This database is frequently consulted when the groundwater quality is evaluated.

OBJECTIVES OF GROUNDWATER MONITORING

As Baggelaar (1996) has already pointed out in his contribution to this workshop, there are four objectives for groundwater quality monitoring by water supply companies:
1. to fulfil the legal monitoring necessity;
2. to underpin operational decisions in safeguarding the provision of good drinking-water, now and in the future;
3. to perform the signalizing function in the Dutch National Environmental Policy Plan;
4. to reassure customers.

The second mentioned objective of monitoring the groundwater is the most important. It requires knowledge on the quality of the groundwater will be extracted in the near future. This prediction is necessary to anticipate in time on quality changes which will affect the purification proces. To achieve this a groundwater network is designed to:
1. examine the groundwater quality, especially to examine the changes in time of the concentrations of pollutants;
2. study the chemical and biological processes which will affect the groundwater quality;
3. evaluate the results of groundwater quality management such as mineral and pesticide legislation and environmental friendly farming.

In practice the first two sub-objectives are closely related.
A special item in the monitoring system is formed by pesticide research. Pesticides are human

made and meant to harm specific species in the environment. Their characteristics and the low maximum allowable concentration (0,1 microgrammes per litre according to the Dutch Drinking-water Act) make pesticides a serious threat to groundwater quality in protection areas. Monitoring groundwater for detection of pesticides requires a special approach. Because of the high costs involving pesticide analysis, it is impossible to implement an extensive yearly research. Nevertheless it is important to know what kind of pesticides to expect. Pesticide research as part of the monitoring program is described in a special paragraph.

MONITORING PROGRAM

WOB has developed a general program for monitoring the groundwater quality in its ten groundwater protection areas. Even though the program is general in approach, it differs slightly per location since local situations are never the same. In this paper we will focus on the general program and use the *Vierlingsbeek* area as an example.
The monitoring program has two levels. The first level is the monitoring system the second level is the control system. The monitoring system has two lines of approach. In both the most important action is gathering information on (figure 2):
1. the extraction area and immediate surroundings;
2. the groundwater quality itself.

In the first approach the technical information about the pumping station and the groundwater protection area is gathered. This means information is collected on:
1. the pumping station (amount of groundwater extracted and the purification proces);
2. the hydrological situation (for instance the location of surface waters);
3. the geological situation (for instance the location of aquifers and confining layers);
4. soil and soil chemistry (what chemical reactions to expect between pollutants, soil and groundwater);
5. land use in the immediate surroundings, types of (sub)soil, et cetera.

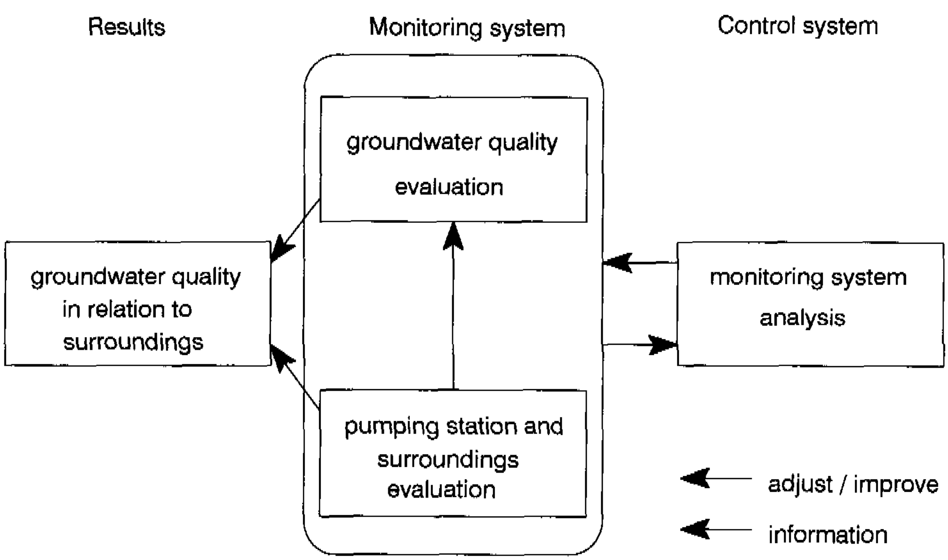

Figure 2: Monitoring program

Gathering this information can be considered as the foundation of the monitoring project. It provides the tools to describe groundwater quality in relation to the surrounding area and influence of human activity. In this way measures to protect the groundwater can be more effictivly be choosed. The information gathered in this step will not change quickly. Therefore it is not necessary to update this information more than every ten to fifteen years.
In the second information gathering approach of the monitoring system the object of study is the groundwater quality itself. This means results of the chemical analysis performed (with water samples of the three network levels) over the past years are studied and interpretated. The changes in concentration of certain components over time are described in relation to the reactiv parts of the soil. Together with the information gathered in the first approach, an estimation of the groundwater quality in the near future can be made. The results of this evaluation makes it possible to take a decision on further action. This can mean more research on a special subject, such as a specific chemical parameter or the decision to build a groundwater quality model for this area. This method of groundwater quality evaluation yields valuable information for the production of drinking-water. Its value depends for a great deal on the accuracy of the gathered information.
To assure the reliability of the results from the monitoring system an analys of that system is performed once in ten to fifteen years time. The chemical parameterset, the sampling frequency and the location of the monitoring wells are object of studie in this analysis. This means that when necessary the monitoring system is modernized.

DESIGN AND MANAGEMENT OF THE NETWORK

The *Vierlingsbeek* groundwater quality network was actually designed in steps. In the early eighties, the water extracted at the *Vierlingsbeek* area showed an increasing concentration of nitrate. This was the first indication of groundwater pollution by minerals due to excessive agricultural activity. To get more information the groundwater needed to be examined, so a tertiary groundwater quality network was designed and constructed. As it was the first groundwater quality network of our company, and - one of - the first in the Netherlands, the expertise of Kiwa Research and Development was used. In co-operation with Kiwa, the national research institute for water supply companies, intensive research was carried out. The experience gained with the *Vierlingsbeek* network is used for the development of other networks. In his manuscript and presentation Baggelaar (1996) has treated an overall strategy for groundwater quality monitoring by water supply companies in the Netherlands. The strategy he describes is now the base of our monitoring program (Baggelaar, 1992).
Designing the network involves projecting monitoring wells in the extraction area. The projection has to include all three dimensions, the position in the area - X and Y - as well as the position beneath the surface - Z -. Both standard monitoring wells and miniscreen monitoring wells are projected. Monitoring wells are used for concentration contours of a certain groundwater level - X and Y - where as miniscreen monitoring wells are used to produce concentration profiles - Z -. A yearly research program was designed, including standard as well as specific attention parameters. The research program is composed in close co-operation with our laboratory, *Stichting Waterleidinglaboratorium Zuid*. Initial sample sessions using broad chemical screening pointed out the specific attention parameters. Further attention was given to standard operating procedures for sampling and to the transport of samples as these form weak links in the monitoring chain.

INTERACTION BETWEEN MONITORING AND COMPUTER MODELLING

Groundwater is our source for drinking-water production so it is important to know what it contains besides H_2O. Changes in the quality of groundwater will result in quality changes of our product unless measures are taken. It is necessary to extrapolate the recent quality changes to the near future, so there is time to adapt the purification proces if necessary. Modelling is a use-

ful instrument to help with this extrapolation. To illustrate this the situation of pumping station *Vierlingsbeek* will be discussed.
Calculations of the use of liquid manure and fertilizer in the groundwater protection area pointed out that high concentrations of nitrate should reach the wells. The first results from the monitoring wells showed extreme nitrate concentrations in the most shallow groundwater (Verheijen, 1985). These measurements and the results from modelling contradicted with the modest nitrate concentrations in the extraction wells with screens at 8 and 28 metres below the surface, as conservative behaviour of nitrate was expected. The network was redesigned and samples of the subsoil were analyzed for reactive components, such as lime, organic matter and iron sulphides (pyrites) (Pool et al., 1987). In the soil samples from a greater depth a substantial amount of iron sulphides was found. Oxidation of iron sulphides by nitrate partially explains the decrease of nitrate and the increase of sulphate concentrations in groundwater at a certain level. In shallow groundwater the nitrate concentration is still high as this water is not influenced by iron sulphides. But still there was a gap between the calculated and the measured concentrations of nitrate in the extraction wells which could not be explained by oxidation of iron sulphides only. Therefore the groundwater quality model of the area was redesigned (Boukes, 1995). The chemical processes between the minerals in manure and the minerals in the subsoil were important parts in this model. The exact amount of iron sulphides in the subsoil available for reaction with nitrate had to be solved by fine tuning the model. Also the assumption that the total amount of iron sulphides would not change significantly in the considered time period was included in the model. The modeling of chemical processes was improved by adding denitrification by organic matter. Also pH-related processes were included in the model. This resulted in a successful calibration, using three keys to optimize five parameters (Boukes et al., 1996). Calculations with the model made it possible to understand and to quantify the denitrification by micro organisms and organic matter. The model was calibrated with measured values which resulted in calculated concentrations fitting the measurements for nitrate, sulphate, calcium, bicarbonate and carbon dioxide. Both pyrite oxidation and nitrate reduction by organic matter and micro organisms explain the decrease of nitrate concentration in groundwater.

MONITORING GROUNDWATER QUALITY MANAGEMENT

One of the lessons learned from the monitoring system *Vierlingsbeek* was the effectiveness of regulations on the groundwater quality. As a reaction to the alarming results of groundwater research and calculations of manure amounts applied on the surface, legislation was developed. Drastic measures were considered such as the forced transition of arable land into grassland in the entire groundwater protection area. Due to resistance of agricultural organizations and our company this suggestion was withdrawn.
In 1986 national legislation was installed to reduce manure application. A distinction was made between grassland and maize cultivation. In 1989 the Provincial Government installed stricter

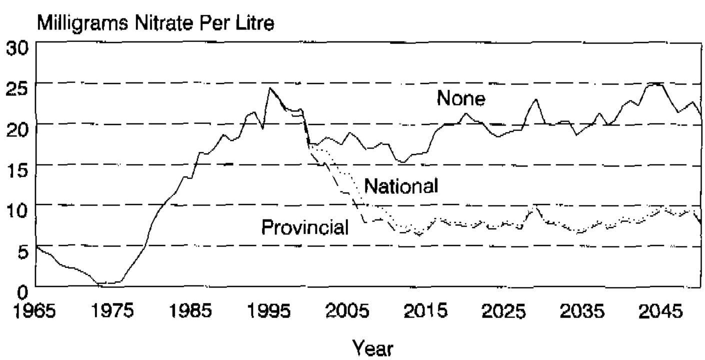

Figure 3: Effects of measures on nitrate concentrations

manure application levels for the Vierlingsbeek groundwater protection area: a 66 percent reduction for grassland, an 80 percent reduction for maize cultivation and a 44 percent reduction for other crops. A period of five years was proclaimed after which the effectiveness of the measures had to be evaluated. This short period was possible because the hydrological system of *Vierlingsbeek* responds fast. One third of the discharged water infiltrated less than 25 years ago. Due to the development of national legislation the amounts of manure which were allowed for application became more restricted over the years: a 50 percent reduction for grassland, a 69 percent reduction for maize cultivation and a 22 percent reduction for other crops. Evaluation of the Provincial regulations learned that now the national levels are strict enough to make extra regulations for groundwater protection areas redundant (figure 3). The very large increase of nitrate concentrations, which started in 1975, is stopped because of the measures taken. This is pointed out by measurements as well as modelling.
In the mean time other lessons were learned. Governments and water supply companies now do not put effort in developing new regulations anymore. The attention has shifted from curative measures or end of pipe solutions to preventive measures. Effective preventive measures are found in what is called *'stimulating policy'* and *'result rewarding'*. Projects are started in order to try to make agriculture more environmental friendly. In stead of punishing offenders, precursors are rewarded. Withdrawing land in extraction areas from their present use by acquiring it for ecological development or foresting is also an effective protective measure.

PESTICIDE MONITORING

Another lesson learned from the monitoring network *Vierlingsbeek* resulted in an effective method for pesticide research. In the *Vierlingsbeek* monitoring and discharge wells pesticides have been found (Janssen and Puijker, 1988)(Janssen et al., 1993). WOB has developed a method which is now an integral part of the monitoring strategy for all its groundwater protection areas. Pesticides found are specific attention parameters in the monitoring program.
Because pesticide analysis is expensive it is important to search as effective as possible. A selection has to be made of monitoring wells which give the most valuable information. Geohydrological modelling is used to determine where a monitoring well is actually situated in the flow pattern surrounding the extraction area. First only the unconfined monitoring wells are selected to gain the most recent information. If high concentrations are found and the pesticide is thought to have leached further down also deeper monitoring wells are sampled.
In addition a selection of possibly relevant pesticides has to be made. Pesticides which have already been detected in the region are automatically selected. First land use information is combined with knowledge about the specific pesticides applied on crops. The land use information is gathered from remote sensing, field recordings and the Dutch Central Bureau for Statistics. An inventory of specific pesticide use in the Province of Noord-Brabant is used to calculate the yearly dose (Ket, 1991). Pesticides which are selected are now screened for quantity and chemical features: if a pesticide is not likely to leach it is removed from the selection. The mobility and persistence of the pesticide is used in the Groundwater Ubiquity Score (GUS) (Gustafson, 1989):

$$GUS = \log(DT_{50}) \cdot (4 - \log(K_{oc}))$$

DT_{50}	time required for the first 50 percent of the applied pesticide to dissipate (day)
K_{oc}	soil/water partition coefficient based on soil organic carbon content (L/kg)

GUS values over 2.8 indicate a high probability that the pesticide will be a contaminant, whereas GUS values under 1.8 indicate a very low probability of it being a contaminant. Further narrowing of the number of pesticides is reached by determining whether there is a reliable analytical method. The detection limit has to meet half the maximum allowable concentration in the Dutch Drinking-water Act (0,05 microgrammes per litre).

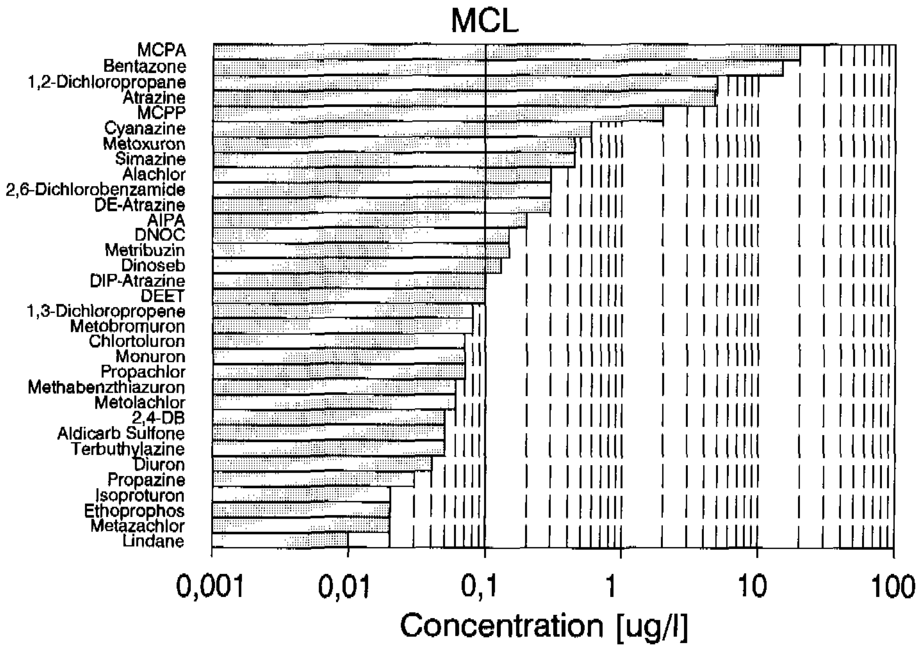

Figure 4: Results of pesticide research

From 1985 to 1994 a total of 93 monitoring wells in five groundwater monitoring networks have been sampled for pesticide research. A number of 70 different pesticides were searched, 33 of which were detected at least once in groundwater (figure 4). Sometimes the active component of the pesticide was found, in other cases metabolites or contaminants of a pesticide were detected. A total of 131 out of 210 positive analysis results equaled or exceeded the maximum allowable concentration according to the Dutch Drinking Water Act (0.1 microgrammes per litre). In groundwater samples taken at 25 meters below surface, bentazon, 1,2-dichloropropane and acetanilides were detected. Maximum MCPA and atrazine levels in groundwater exceded WHO guidelines for drinking-water quality of 2 microgrammes per litre (WHO, 1993).

USING THE RESULTS

As mentioned before, pesticides have been found in the groundwater quality network as well as in the extraction wells at the *Vierlingsbeek* pumping station. This information was used to adjust the purification process. In addition to the conventional rapid gravity filters, granulated activated carbon filters were installed. Bentazon, mecoprop and 1,2-dichloropropane have already been detected in the secondary network. Atrazine and simazine have been detected in the groundwater quality network and will probably soon appear in the raw water.
The groundwater network has also shown that the acidification led to increasing mobility of heavy metals and an increase of hardness of the groundwater. The attention for nitrate and sulphate problems at *Vierlingsbeek* have shifted towards cobalt, nickel and zinc (van Beek and van der Jagt, 1996).

CONCLUSIONS

Building a well functioning groundwater quality network takes a lot of effort but it will give a lot of answers as well. It does not stop at installing monitoring wells. Modelling is a useful tool for predicting the quality in the future. In the Budel groundwater quality research the fourth dimension - time - was introduced in modelling (Boukes, 1996). Using very little input data the model gave an accurate explanation for quality differences between individual wells.

ACKNOWLEDGEMENT

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USE OF NUMERICAL GROUNDWATER MODELS IN CONNECTION WITH DATA SAMPLING AND ASSESSMENT

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ABSTRACT

Numerical groundwater models have the purpose of simulating groundwater flow under particular site characteristics. Using basic geological and hydrological informations, models are able to produce a visualization of the groundwater flow under the prescribed conditions.
An optimal development of groundwater measurement systems by means of models will be demonstrated by two case studies.
In the first case study long-term groundwater flow simulation was based on model connected short-time monitoring of time-varying influence of a river.
The second case study refers to an adequate model simulation of the characteristics of river bank filtrate flowing to a row of wells supported by detailed monitoring.

INTRODUCTION - GENERAL REQUIREMENTS

For exploring and developing groundwater resources in respect to environment and with necessary protection against pollution far-reaching knowledge is required.
This concerns

- the present groundwater system with regard to
  - the geological structure
  - the geographical/morphological boundaries
  - the hydrological conditions

- the present interactions with regard to
  - the entire water balance
  - the interaction between groundwater and surface waters
  - the effects of human impacts on the investigation area

The following tasks arise:

- integrating water resources management and rural, urban and industrial development

- protection against groundwater pollution

- securing of ecological aspects

The solution of these tasks requires information as detailed as possible. The thorough use of these data and their evaluation are necessary to recognize the existing integrated groundwater system. Usually basic informations of the investigation area with regard to geological settings, water resources management and geographical conditions are available. These are based on existing monitoring systems.
In order to evaluate the dynamic system behaviour the mere interpretation of such information is

not sufficient. It is impossible to predict the system reaction solely on data interpretation.
Here the method of numerical groundwater model simulation offers the state of the art. This simulation is based on the available information of the area. It will only yield reliable results in connection with a sufficient and appropriate data sampling and a site specific evaluation. Accordingly a detailed and substantial monitoring-system is an essential basis for the reliable and successful use of numerical groundwater models.
The integrating property of these simulation methods, which correspond to the dynamic procedures in the groundwater area can be used to optimize a monitoring system.
Following this procedure, future system reactions can be predicted.

PROCEDURE OF WORKING

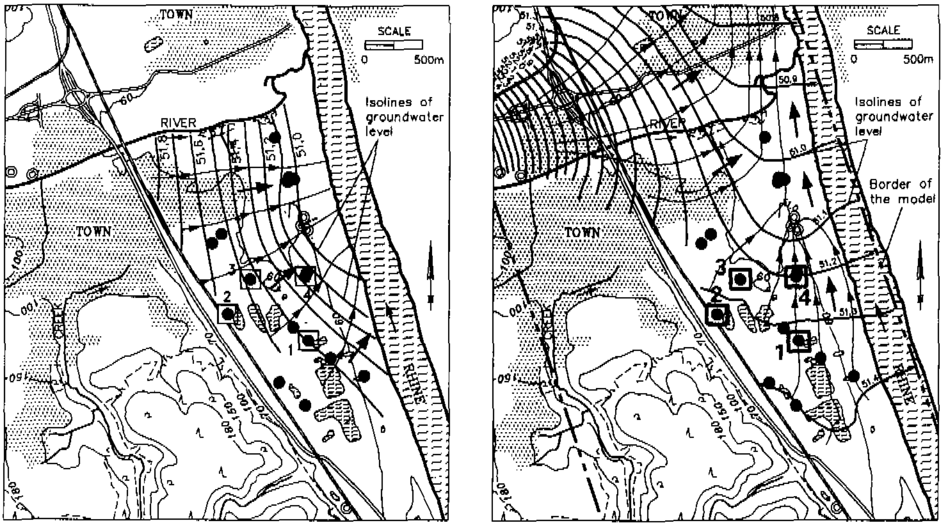
As a first step available informations must be gathered. Their evaluation results in an overview of the situation. In most cases the results are not sufficient for a reliable groundwater development.
At this state even an approximate groundwater model simulation provides a first overview of the possible groundwater flow. Of course, this must be supported by many assumptions of the boundary conditions.
The next step consists in description of the decisive lacks of information and the planning of an advanced monitoring system in order to get the necessary data. It has to fit the following requirements:

- collect groundwater levels in areas not yet covered
- check of hydrogeological conditions in different depths
- recognize groundwater quality over the investigation area in different depths with regard to
  - typical anorganic parameters
  - if necessary selected organic parameters

Connection of hydrological and hydrochemical information from these data allows conclusions concerning actual groundwater flow and transport conditions. These must be checked in a further extended model calculation.
Groundwater levels, flow paths and flow times available by model simulation can be assessed by plausibility tests due to more refined results of the model.
Very often this reveals the difference between the measured short-time groundwater situation and not measurable long-term states. An additional model simulation of hydrological short-time changes gives valuable results of locally different reactions to specific influences such as surface waters.
Beyond that long-term measurements including hydrochemical variations may allow an additional simulation of local groundwater flow characteristics. Thus an optimization of the monitoring system is possible. This way main and local characteristics of the groundwater system can be recognized and future situations can be simulated with a high degree of confidence.

CASE STUDIES

In the first case study a groundwater area influenced by the river Rhine demonstrates the application of the iterative process between monitoring and modeling.
In the investigation area near the river Rhine only restricted hydrological and geographical informations were available as a first basis for the planning. From some irregularly distributed boreholes four groundwater level data of one year existed only.
The first model simulation proved, that these data could not reflect the long-term groundwater situation.



Average of measured momentary states

Model calculation based on especially developed monitoring

◎ existing production wells

● observation wells

▣1 observation wells selected for short time monitoring

surface waters

flow path of groundwater

Figure 1: Evaluation of het long-term groundwater situation

Figure 1 shows in a surveying map the settings of the area. In the left part of the figure the average groundwater flow resulting from the sparse observation data is shown. The right part of the figure describes the general long-term groundwater flow situation resulting from a model simulation with regard to the boundaries and geological conditions. Considerable differences between the two figures are obvious.
Even under extreme variation of the boundary conditions of the area (geology, hydrology) the measurements could not reproduce the long-term situation within the scope of acceptable uncertainty.
Thus the measurement data could only represent short-time states of the groundwater situation varying with time. Moreover these facts illustrate the differences of temporary conditions and their local variation.
As a first step a time-dependent monitoring at different locations was essential in order to support and improve the model simulation. Because of the advanced state of the planning, a long-term observation was at this time not practicable. Thus, a monitoring system had to yield the entire information by measurement of short-time groundwater reactions at a few suitable locations. Due to the influence of the river Rhine on the groundwater level in the investigated area, from this monitoring system essential information about the real natural conditions could be expected.
The observation wells selected for the short-time monitoring are marked in figure 1. The varying water levels of the river Rhine and of the groundwater in two exemplary observation wells during the measurement period are shown in figure 2. Additionally the corresponding groundwater levels computed with the adjusted groundwater model are displayed. It can be seen, that the model approximately simulates the natural conditions. The other observation wells showed similar results.
By connection of an adjusted monitoring system with model simulation in this case a reliable description of the situation in the groundwater area could be achieved. This allowed calculations of groundwater reactions due to the planned water resources management.

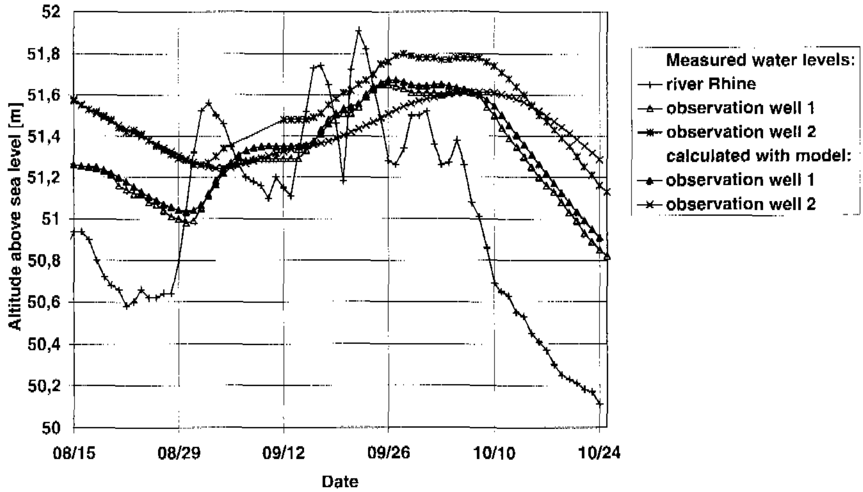


Figure 2: Comparison of measured and calculated water levels

In this case study the time-varying influence of the river Rhine was mainly responsible for the dynamics of groundwater. In other situations different effects may exist, that require additional aspects for an optimized monitoring.

In the second case study (Zipfel/Horalek, 1987), the rate of bank filtrate pumped by a row of wells near a river and the travel time of bank filtrate had to be determined. Because of the 60 m thick sandy layers of the aquifer, the variation of travel times and flow paths over the depth had to be discovered.

Based on informations about the hydrogeological conditions of the aquifer, the groundwater flow can be simulated with a spatial (three-dimensional) groundwater model. In the present case the variations of the sandy sediments over the depth of the aquifer could be determined only inaccurately from the available boreholes.

Therefore a special monitoring system with several observation wells situated between river and wells had been developed. This allowed sampling of data from different depths. First the piezometer heights of groundwater in different depths could be measured. The differences were small

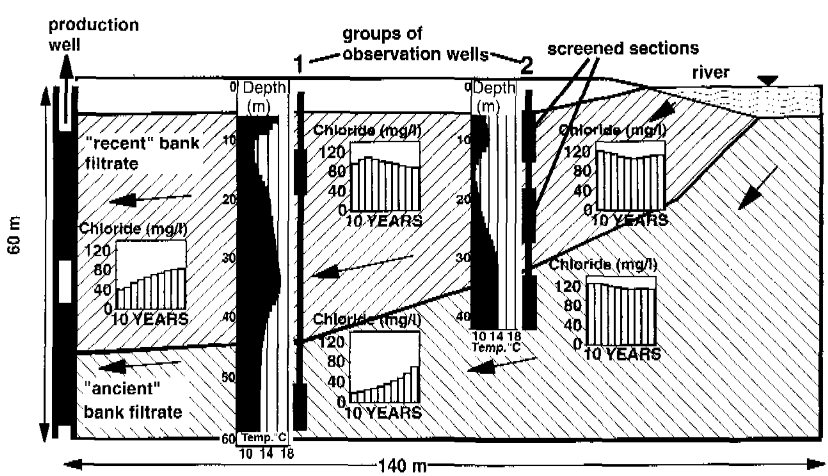


Figure 3: Concentrations of chloride in the groundwater (10 years) and temperature of het grondwater in het summer across depth

because the aquifer has no remarkable stratification.
Otherwise flow paths in different depths between the river and the row of wells could be achieved. Measuring the locally different hydrochemical conditions and the temperature gradient over the depth brought these results.
Thus, using these additional informations the distribution of the groundwater flow from the river to the wells could be reproduced by the groundwater model (Schöpfer, 1994).
Figure 3 explains the situation in a cross sectional view through the aquifer from the river to the row of wells (Schöpfer and Zipfel, 1995).
As an example the measured variation of the chloride concentration in the groundwater is shown over a period of 10 years. The chloride concentration in the river ranges between 100 und 140 mg/l. Because of the high bank surface no overland flow effects existed.
Additionally the measured distribution of the groundwater temperature in summer over the depth is displayed. In this time, when the river has water temperatures between 16 and 24 °C, lower temperatures occur especially in the groundwater near the water table.
In other months the distribution of temperature of the bank infiltrated groundwater changed accordingly.
The temperature distribution varying with time and depth has been discovered by the depth dependent monitoring system. Also the variations of several hydrochemical parameters in time have been registrated. On this basis a simulation of groundwater flow and transport was possible in connection with the hydrological and hydrogeological data.
The model simulation produced results of the amount of bank filtrate flowing to the row of wells and of its spatial and temporary variations. It was detected that two areas with different flow times from the river to the row of wells have developed. In the deeper part of the aquifer a substantially longer flow time of the bank filtrate is necessary ("ancient" bank filtrate) and accordingly the pumped water of the wells is mixed. By a specific monitoring system in connection with the groundwater model in this case the detailed determination of the flow conditions could be achieved. A more extensive use of the bank filtrate without quality problems has been possible.

CONCLUSION

Numerical groundwater models are excellent instruments for the simulation of flow and transport processes in groundwater. However, as operation basis they require reliable information on conditions in the investigated groundwater area. For this purpose substantial monitoring and its evaluation is absolutly necessary. Due to the integrating working of groundwater models an monitoring system optimally adjusted to different requirements can be developed. It can be applied with acceptable costs by a step-by-step approach in connection of model and data-sampling with regard to considerable lacks of information and knowledge. As demonstrated by case studies the monitoring system supports the best possible simulation of groundwater conditions by a gradually improved model.

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REDUCTION IN A NUMBER OF MONITORED COMPONENTS WITH CONCURRENT INCREASE IN THE GROUNDWATER MONITORING EFFICIENCY

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ABSTRACT

The operation of a groundwater monitoring system at certain sites as a means of solving two main problems:

- *the observation and control of the groundwater and soil conditions (this information is vitally important to administrative bodies which deal with environmental problems);*

- *the analysis of the technical conditions of the installation(observation of the design requirements for the technical regime of operation), this information is important to its owner.*

The first problem is a traditional one, while the second originates from the changes in attitude of the owners (or directors) of the sites with respect to monitoring. It has been suggested that these two problems can be solved by switching to a site-specific computation of the local monitoring network and the application of flexible regimes of observation, corresponding to a given stage of the sites functioning and/or situation.

PROBLEM DEFINITION

In order to solve the rather urgent current problem of how to protect the groundwater against pollution and/or depletion caused by the operation of economic installations, it is necessary to design local monitoring systems with corresponding programs of observations, elaborated on the basis of a modern concept and reflecting present-day realities. Monitoring of groundwater in definite sites is based upon the collection and analysis of a considerable number of samples. The monitoring methodologies which are available require data on scores of controllable components but, as a rule, they do not take any detailed features of the various sites into account (such as technical processes in the run, dimensions, environmental conditions in the region etc.). As a result, monitoring is very costly, while a considerable part of the data which is obtained on the groundwater quality remains unused. One of the reasons for this low level of efficiency is that monitoring systems are traditionally seen as systems which are to determine the state of the groundwater as a whole, without allowance for the characteristic, natural and technical features of the site, while the collected observation data has no practical significance for the owners and administrators of the site.

In Ukraine, and in some other countries, there is now a tendency to re-examine the principles of the local monitoring organisation. This is first of all due to economic problems, as the full-scale (in the traditional sense) monitoring program is too costly, while a large proportion of the collected data remains unused.

So there is a tendency to limit observation networks by developing network elaboration criteria, solutions for the problems concerning optimisation, and by accounting for migration characters of pollutants, etc. Moreover, the identification of pollution sources allows one to reduce the number of monitored components while simultaneously maintaining reliable control over the technical state of the site (or its parts). And this will enhance the interest of the owners of the sites (or the administrators) with respect to organising and operating a monitoring system for its groundwater.

POINT POLLUTION

The authors are presently completing a feasibility study concerning a reconstruction of the groundwater monitoring systems at two locations- the Zaporizka Nuclear power plant (the largest in Europe) and the Zmievska Thermal power plant (the largest in the north-east Ukraine).

The reconstruction of the Zaporizka NPP control network is being performed with allowance for, first of all, the technical features of operation, including the territorial location of main units which are the potential sources of pollution.
The potential sources of polution at the Zaporizka NPP include, first of all, underground pipelines which may leak in the event of accidents - when the plant is operating normally, there are no provisions for leakage. Pollutants may also enter the groundwater from other units in the event of an accident, spillage or leakage of polluted liquids. The observation points must be selected on the basis of a mathematical model of the site, which includes any possible technical disturbances and an emergency situation at all stages of operation and with an evaluation of the possible impacts on the groundwater, allowing for their natural degree of protection.

Experiences in local monitoring operations have demonstrated that it is impossible to obtain a continuous evaluation of the possible impacts on groundwater using the traditional set of six macro-components (Ca, Mg, Na, HCO_3, Cl, SO_4). Controlling the state of each separate installation on the NPP can only be achieved by identifying the characteristic components. The main monitored components were selected on the basis of qualitative analysis of liquids which are typical for an industrial process of a given unit (e.g., surfactants, radioactivity or temperature are typical for a Special housing and tritium is typical for sprinkling basins), or the composition of the fluids which are transported by a given underground pipeline (e.g. NH_4, NO_3 and surfactants in the case of municipal sewerage, and oil products or the composition of substances (bulk storage of reagents - NH_4 and K) in the case of industrial waste lines). In the traditional approach, the section of the table presented below (25 wells) should have contained 825 components (600 macro-components and 225 micro-components). Using the authors' method, it is possible to reduce this number to 194 (162 - macro-components and 32 - other).

In addition to the units presented in the table, there are between 2 to 4 underground pipelines in the monitoring zone of each well. The authors are planning to carry out an investigation in order to identify a specific source of pollution at the Zaporizka NPP site at the isotopic level. A specific test operation regime and the components to be monitored were selected for each well (or group of wells), as well as the components to be monitored. Such an approach is possible when detailed information on the site is available, or when one is only dealing with point pollution (real or expected one).

NON-POINT POLLUTION

For sites which are affected by non-point pollution, or in cases where the groundwater is polluted in an integrated way, the monitored components should be selected by applying mathematical statistic methods to the water quality observation data analysis. Zmievska TPP, like all thermal power stations, pollutes the air, the surface and the groundwater of a considerably large territory (interpretation of aerial photographs data reveals an area with a 30 km radius where diffusion pollution occurs). The authors have carried out investigations in order to identify and to determine priorities with regard to the sources and sites of groundwater pollution in the region of the Zmievska TPP site for a number of years now.

NN well	Well location	Monitored ingredients		
		monthly	quarterly	annually
1	R5	L,T	b-act.	OP,NO_3,A.A.
2	R5	L,T	b-act.	OP,NO_3,A.A.
3	R3	L,T	b-act.	OP,NO_3,A.A.
4	R2	L,T	b-act.,OP	A.A.
5	R1	L,T	b-act.,OP	A.A.
6	spr.bas.	L,T	b-act.,S,NO_3	A.A.,Tr.
7	SH2	L,T	b-act.,S,NO_3	A.A.,Tr.
8	SH2	L,T	b-act.,S,NO_3	A.A.,Tr.
10	spr.bas.	L,T	b-act.,Tr.	A.A.
11	spr.bas.	L,T	b-act.,Tr.	A.A.
12	spr.bas.	L,T	b-act.,Tr.	A.A.
13	spr.bas.	L,T	b-akt.,Tr.	A.A.
14	B6	L,T	b-act.	-
15	B6	L,T	b-act.,K,NH_4	-
16	MH5	L,T	b-act.	A.A.
17	B5	L,T	b-act.,NO_3,Tr.	A.A.
18	MH5	L,T	b-act.,NO_3	A.A.,OP
19	MH4	L,T	b-act.,OP	NO_3,Tr.
20	B3	L,T	b-act.,OP	NO_3,Tr.
21	B2	L,T	b-act.,OP	NO_3,Tr.
22	B1	L,T	b-act.,OP	NO_3,Tr.
24	SH	L,T	b-act.,NO_3,NH_4	A.A.
26	R1	L,T	b-act.	A.A.
29	R1	L,T	b-act.	A.A.,OP
30	B1	L,T	b-act.	-

Table 1: A section of the table with characteristics of the recommended groundwater monitoring system

In which: R - reactor unit; SH - supportive house; MH - main house; B - block; spr.bas - sprinkling basin; OP - oil products; S - surfactants; A.A. - abbreviated analysis; L,T - level and temperature ground waters ; b-act.- beta-activity

Analyses have elucidated that the pollution sources of the uppermost aquifer horizon include the inflow of components from piled-up aerial sediments of waste products, infiltration of polluted waters through the poorly screened bed of the ash dump, distortions in the natural exchange of waters in the cooling pond (Liman lake) and the irrigation system of adjacent lands. It was concluded from investigations that the groundwater of the territory surrounding Zmievska TPP is generally polluted by the whole range of technogenic sources of pollution.

On the basis of a comparison of the probabilities of priority pollutants being present in samples in various concentrations, the authors refined the procedure of sampling and compiled the list of substances to be monitored. The authors have subdivided the priority components into water hazard categories on the basis of the following criteria:

- relative frequency of the detection of the component in samples (in quantities above background value);
- the component toxicity level;

- average statistical MAC overtop for the component;
- *the component's persistence time.*

The component toxicity level was determined on the basis of its hazard class and MAC value. The persistence time here was determined as the component capacity for long-term migration in the aquifer. The maximum categories were determined, as could be expected, for Mn and Cd, which are widely distributed in concentrations above MAC.

Indices such as total salinity, nitrates and fluorine were also considered as high priority ones. In several cases, such an approach allowed for a reduction in the number of traditionally monitored substances, without having any detrimental effect on the aims of monitoring.

These investigations have made it possible to improve the long-term monitoring of groundwater to make it more aim-oriented and economically efficient, and also to prevent disputes when determining any guilty party in the event of the flooding of adjacent territories (Liman town) and the pollution of the soil and groundwater.

Here too, just like in Zaporizka NPP, the station administration has changed its attitude towards groundwater monitoring because the practical aspect of the obtained information became apparent.

There are plans to apply the results of the investigations at Zmievska TPP to the Zaporizka NPP region (a powerful TPP is situated in 3 km from NPP), in order to extend the monitoring network for the identification of regional sources of pollution. The Zaporizka NPP administration has invited the authors to perform this study. Preliminary investigations of the region's groundwater have shown that the main pollutants are generated at the thermal power station.

CONCLUSION

The recommended site-specific approach to groundwater monitoring (sampling frequency and monitored components) at Zaporizka NPP and Zmievska TPP allows one to control the state of all of the main units within the station site, to promptly detect technogenic impacts on the region's groundwater and to determine the sources of pollution.

Because of the site-specific approach to each unit and well, it was possible to optimise the monitoring network and to reduce the number of monitored components while simultaneously increasing the value of the information and, consequently, improving the monitoring results.

Together with an annual report - which is first of all intended for the environmental authorities, but also for the station administration - information on the state of groundwater, which is to be published monthly, is being prepared. This monthly information will allow one to exercise control over the technical state of underground parts of constructions and the main pipelines.

THE USE OF MODELS IN MONITORING STRATEGIES

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ABSTRACT

For decades monitoring networks have been designed either for long term monitoring or for special short term tasks. The use of water quality models can help the designer to optimise the network by showing where impacts can be expected and under which conditions. The authors have used the MIKE model system to optimise monitoring strategies for short term and long term monitoring tasks. The two cases described used the results from models to design a short term monitoring system for sampling data for an impact assessment and a long term system for monitoring the oxygen conditions in a stratified lake.

INTRODUCTION

During recent years VKI, Water Quality Institute, in co-operation with DHI , the Danish Hydraulic Institute, has developed computerised hydrodynamic and water quality models that are also capable of handling biological variables.

The models have mainly been used to predict changes in the water bodies under different impacts, i.e. from construction of bridges, discharge of waste water, regulations in rivers, etc. However, the use of mathematical models and carrying out sensitivity analyses for various system variables and parameters may help the designer determine important aspects of a monitoring strategy, such as list of variables to be monitored, frequency of monitoring and sampling locations.

This paper will demonstrate two cases, where models have been used to predict the future conditions in the water bodies and where models have also been used to help design the future monitoring systems for the water bodies.

THE MODELS

The models used are the MIKE-11 and the MIKE-12. Both models are one-directional models. The Mike-11 is a one-layer model, whereas the Mike-12 is a two-layer model.

The models are based on fully dynamic hydraulic models, where the advection-dispersion results are used in the biological modules for the calculation of water quality variables.

PHYSICAL DATA:

The models are set up according to the physical data for the river/water body including data on the slope, dams, impoundment, bottom sediment, Manning number, weirs, etc.

WATER QUALITY DATA:

The Mike system can model water quality on 6 levels where each level has some requirements regarding the amount of in-data. The first level only needs data on BOD and dissolved oxygen, whereas the sixth level needs data on suspended BOD, dissolved BOD, dissolved oxygen, NH_3-NH_4 and NO_2-NO_3.. Data on the water quality in rivers are often easy to obtain. However, data on sewage and storm water are more difficult to get and it is often necessary to generalise on the basis of a few data. A major problem working with sewage data is specially the lack of data on the oxygen contents in the storm water.

The loading is set up in separate time series for each source and these series are applied to all sources entering the model area, beginning with the loading entering the river at the up-stream point.

THE CASES

CASE 1: RIVER MØLLEÅ, DENMARK

BACKGROUND

River Mølleå has a length of approximately 10 km and drains the lake system of lake Furesø, lake Lyngby sø and lake Bagsværd sø. There is a difference in altitude between the upper and lower part of the river of nearly 20 meters. However, due to this large difference in altitude the river has been dammed with eight dams. The watershed is approximately 100 km^2. Most of the watershed is within the borders of the Copenhagen County.

River Mølleå and its surroundings have a very high value as a recreational area for the population of Copenhagen. However, the water quality is not considered acceptable, the main reason being impacts from storm water run-off from some 30 points of discharge.

The Mike-11 model was used to assess the impact from storm water run-off on the water quality in the river. The loading from the storm water run-off was calculated on the basis of default values and the model was used to assess the sensitivity of the size of the loading regarding oxygen depletion in the river.

Another objective was to determine the impact from a number of impounded dams on the water quality in parts of the river system, especially from the eutrophication levels in ponds.

Usually models have to be calibrated on the basis of existing data, but in this case the aim with the modelling was to emulate the situation and then use the results for the design of a monitoring network, that could supply data for a calibration of the model. The calibrated model will be used in an impact assessment of the point-source pollution in the river.

DATA

A Mike-11 model was set up for the river, including all the dams and the major storm water discharge points. The only available data were on the flow in the river.
It was decided to use a rough set-up, simulating two weeks of August. The base flow in August is usually low but also known to increase rapidly due to intense rain storms causing storm water discharge.

The flow in the river was calculated on the basis of several years of measurements, and calculations on the discharge from the storm water outlets were based on results from previous investigations. The outlets were calculated according to the loading and to the discharge time. A worst-case incident was constructed, with all outlets were active at the same time.

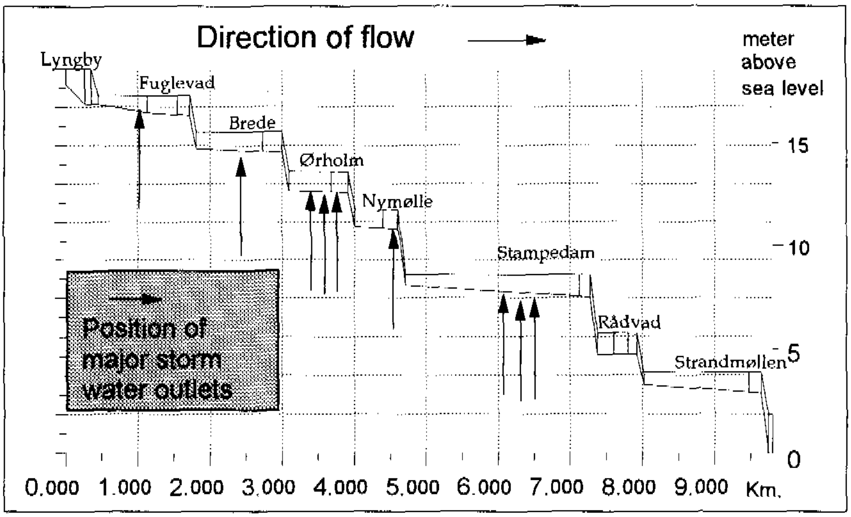

Figure 1. The model set-up of the River Mølleå.

RESULTS

The results from the model showed that the storm water discharges reduced the oxygen levels substantially. The passage of each dam increased the oxygen level, but not to a level comparable to the situation before the discharge situation. The oxygen level in the river just before the discharge event had a clear influence on the conditions under and after the event. If the event took place early in the morning, when the oxygen concentration was low due to respiration, the impact was larger in terms of time and space whereas it was shorter when the event occurred during the afternoon with high oxygen concentrations.

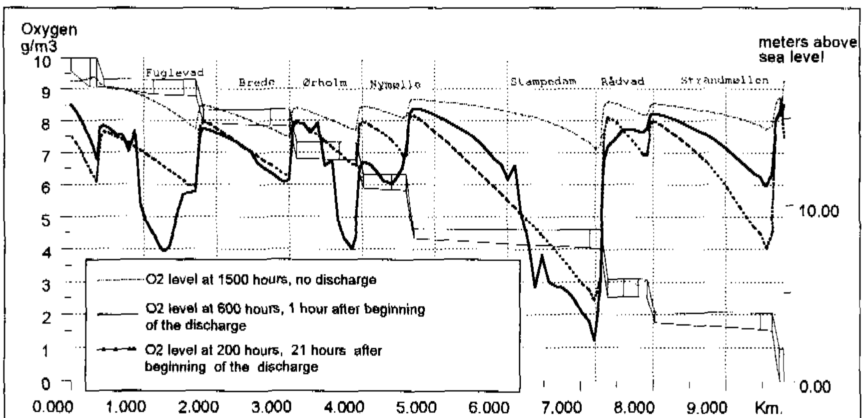

Figure 2. Oxygen concentrations prior to (at 15 o´clock), one hour after (at 6 0´clock) and 21 hours after the storm water discharge (2 o´clock the next day).

The sensitivity analysis showed that the oxygen levels did not change substantially when the loading during the discharge situation was increased by 50%.

The result from the modelling was then used to help design a monitoring system to encompass the main problems and to get only the necessary data for the future work, which includes an evaluation of the impacts from a proposed discharge from a sewage treatment plant. The results will also be used for an evaluation of the impact from each of the individual storm water outlets to set up a prioritised list according to their impact on the water quality in the river.

From the model a plot of an event simulating all active storm water outlets was used to determine the position of the sampling stations (Fig.2), and it was decided to sample in the pond at Ørholm and at an existing station at Stampedammen. Samples were also taken at the upstream dam at Lyngby.

With different modelling scenarios it was also shown that oxygen data on the discharge water was very important and it was decided to include measurements of the oxygen level in the storm water systems during a discharge situation.

CASE 2: LAKE KARLSGAARDE, DENMARK.

BACKGROUND

Lake Karlsgaarde in the southern part of Jutland is an artificially made hydropower lake constructed in the 1920'es (Fig. 3). The hydropower plant is now outdated and a rehabilitation of the river Varde feeding the lake has been proposed. The lake was heavily polluted with mercury 30 years ago and mercury is still found in substantial amounts in the lake sediment.

As a part of the rehabilitation project a 50 % reduction of the river flow to the hydropower lake has been proposed. This will have a substantial impact on the ecological conditions in the lake. The regional authority Ribe County decided to use a modelling approach to predict the changes mainly in the oxygen concentration in the stratified lake and moreover to use the results for managing the new situation with reduced flow to the lake. It was crucial to the authorities that the mercury in the lake sediment was not disturbed, for example by gas production due to anaerobic conditions in the hypolimnion.

DATA

In the summer of 1995 an intensive field investigation was carried out, including flow measurements, oxygen and temperature profiles and sampling of water quality variables. Meteorological data such as wind speed and direction were also recorded. The results showed clear stratification in the lake with an epilimnion and a hypolimnion layer. During the last two weeks of August the oxygen level in the hypolimnion changed due to low flow to the lake. The level dropped from 9 mg O_2/l to 1 mgO_2/l in 10 days and rose again to 8 mg/l within 2 days.

It was planned to make the proposed reduction of the flow during the summer to monitor the changes in the lake. However, the summer of 1995 was very dry resulting in a natural reduction of 50 % in the flow.

A Mike-12 two-layer model was set up to investigate the impact on the oxygen levels in the hypolimnion during reduced flow conditions.

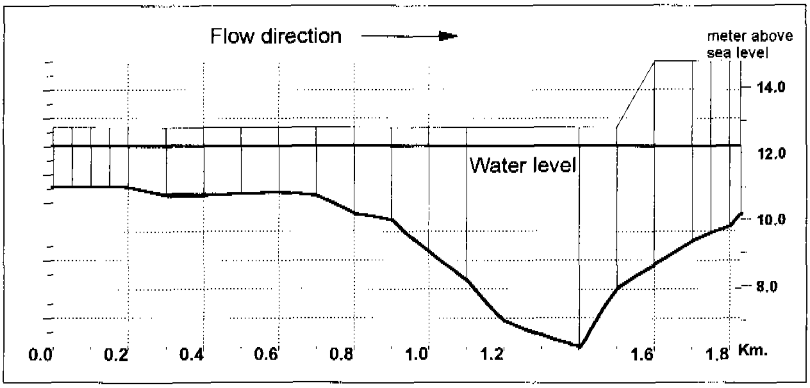

Figure 3. Profile of Lake Karlsgaarde with the hydropower station to the right.

Initially the model was calibrated on the field data. Based on the calibrated model, a number of scenarios with varying flow and wind conditions were run to generate a spectrum of results usable for the design of a monitoring system.

The aim of the monitoring system was to be able to detect changes in the oxygen levels and to establish a procedure for the management of the flow to the lake that would prevent the oxygen level of the bottom water from reaching anaerobic conditions facilitating gas production in the sediment and thus creating a possibility of resuspension of the mercury to the water.

Modelling of the different scenarios showed that critically low oxygen conditions at the bottom layer occurred, when the flow into the lake was reduced to less than 4 m3/s. It also showed that the wind had a substantial influence on the mixing of the two layers.

Based on the results from the modelling it would be possible to design a monitoring programme aiming at detecting low oxygen conditions in the bottom layers of the lake by concentrating the monitoring effort on the flow into the lake, rather than continuously monitoring the oxygen conditions in the lake. Construction changes to the water inlet to the lake ensuring a minimum flow of 4 m^3/s was also possible (Fig. 4).

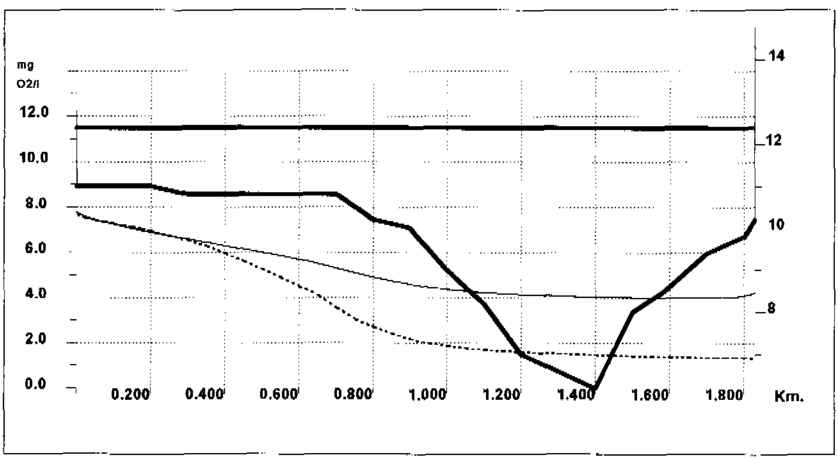

Figure 4. The oxygen condition at the bottom layer. The solid line represents the condition with a flow > 4 m^3/s into the lake and the dotted line represents the condition with a flow < 4 m^3/s.

CONCLUSION

The use of models as tools for designing monitoring programmes was a major advantage in the two cases presented. In River Mølleå the modelling showed where the main problems were in the river and the design of a short term monitoring system was limited to three sampling points supplemented by oxygen measurements at several stations.

In Lake Karlsgaarde the modelling showed that monitoring could actually be limited to flow measurements at the inlet to the lake or even simpler be limited to ensure a minimum flow to the lake, as it is a regulated, artificial lake.

The use of models as tools when assessing the impact from future point-source pollution will enable decision makers to optimise the design of a monitoring programme, as the results from the models will show the effects from the pollution at certain locations along a river.

The disadvantage of using models is the possibility of disregarding substantial problems which might have been detected if a traditional approach had been used for the design of a monitoring programme. However, such problems should eventually be recognised during the calibration phase of a model and thus lead to a redesign as is also the case when using the traditional approach.

METHODS AND MODELS FOR ASSESSMENT OF HYDROLOGICAL AND ECOLOGICAL CONDITIONS OF RIVER MOUTH REGIONS

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ABSTRACT

The paper deals with the complex of empirical, semi-empirical and semi-theoretical methods which have been specifically developed and verified for the computer processing and prediction of hydrological and ecological characteristics of aquatic systems in river mouth regions. The empirical computer processing methods include the following: the calculation of channel degradation or channel aggradation processes due to sea level changes, the calculation of the surface areas of inundated land in flood plains and deltas. Some methods were developed for the spatial calculation of hydrological characteristics such as water discharge distribution between delta branches, storm surge propagation, current velocities and salinity changes along the river plume at the near shore, and salt water intrusion into the river. The mathematical model for the temporal variability of dissolved non-conservative substance in the river mouth waters has also been developed. Hydrological and ecological conditions of semi-enclosed seas and sea bays, estuaries, lower reaches and deltas of rivers and near shore zones in Russia and other countries of the former USSR, *were selected as the subjects of investigation and verification of the above-mentioned methods. These methods are considered to play an important role in the monitoring of aquatic ecosystems.*

INTRODUCTION

The study of the hydrological and ecological characteristics of water objects is very important for the use and protection of water resources. Permanently monitoring the hydrological and ecological conditions of water objects is greatly significant in this respect. The main components of such a monitoring system usually include the observance, collection, accumulation, processing and distribution of data. However, when applying these monitoring systems, one may frequently encounter some serious difficulties connected with the following problems.

1. The conversion of easily measured characteristics (water levels, temperature, salinity, etc.) into characteristics which are more complex and seldom measured (water discharges, discharges of suspended and dissolved matter, components of ion composition, etc.), poses a problem.
2. The observation data is usually obtained at separate points and needs to be extended in space - from the separate points (gauges, stations, cross-sections, etc.) to the whole water system (volumes, surface areas, etc.).
3. The observation data is discrete and usually needs to be interpolated and extrapolated. The prediction of changes in hydrological and ecological characteristics obtained from the current observation data is the most important and actual.

Because of the above-mentioned reasons, a comprehensive and effective water monitoring method has to include a system of computer processing and prediction of the spatial and temporal variability of characteristics in addition to the observations. This is especially important for water systems which have a complicated hydrology and a very variable and vulnerable ecology

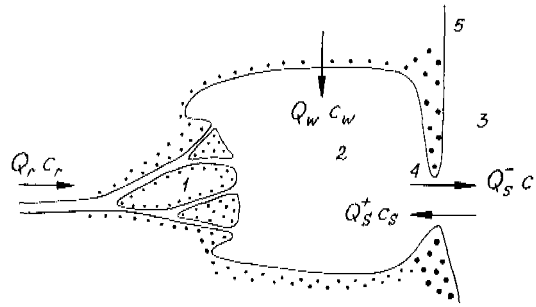

Figure 1. Scheme of a river mouth: 1 - delta, 2 - estuary, 3 - open near shore zone of the sea, 4 - straight between estuary and the sea, 5 - barrier bar (spit). Q_r - river water discharge, Q_w - sewage discharge, Q_s^+ and Q_s^- - water exchange between estuary and the sea, c_r, c_w, c_s and c - concentrations of the given dissolved substance in river, sewage, sea and estuarine water.

and which are subjected to a significant degree of anthropogenic influences. Such water systems include river mouth regions (Figure 1). River mouths are unique zones of river and sea (lake) interaction. They have very changeable water and sediment regimes, large spatial gradients of characteristics, and they pose multiple problems in studies and in the organisation and operation of monitoring systems.

Many river mouth regions in Russia represent very important systems with respect to ecology and economy. Water management and the protection of the natural resources of these systems are actual problems in Russia (Mikhailov et al., 1986). The most important and vulnerable river mouths include: the Volga delta and the Northern part of the Caspian Sea with a very changeable water level, the deltas of the Don and Kuban Rivers with artificially decreased water flow, the Ob, Yenisey, Lena, and Yana deltas, which are of important relevance to navigation, the Neva mouth with high water pollution, etc.

EMPIRICAL COMPUTER PROCESSING METHODS

There are many empirical methods which allow us to compute hydrological and ecological characteristics which are difficult to measure on the basis of the observation data characteristics which are easily measured. The latter characteristics include the following: water levels (water stages), water temperature and conductivity, sometimes water discharges. The empirical computer processing methods are based on the system of empirical relationships between different characteristics.

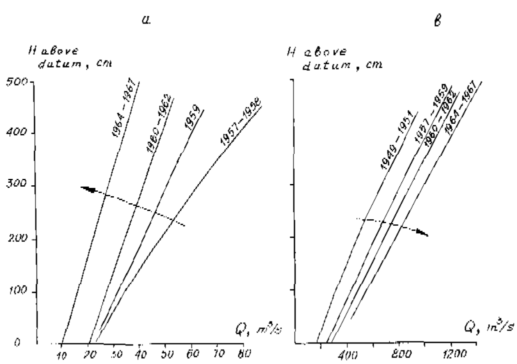

Figure 2. Stage - discharge relationships for two channels in the Danube delta: filling up (aggradation) the Sredniy branch (a) and eroding (degrading) the Bystriy branch (b).

RELATIONSHIPS BETWEEN WATER LEVELS AND DISCHARGES

These relationships, often called "stage - discharge relations" or "stage - discharge curves", are the main means to compute river runoff. In addition, these curves permit us to compute the distribution and redistribution of water discharges between delta branches, and to assess the channel degradation or aggradation processes in deltas. For example, the process of water discharge redistribution between the channels in the Danube delta is shown in Figure 2.
"Stage - discharge curves" were used for the assessment of the rate of channel degradation and aggradation in the delta of the Sulak River in reaction to variations in the Caspian Sea level (Figure.3). The main reason for the changes in the level of the Caspian Sea is that this large

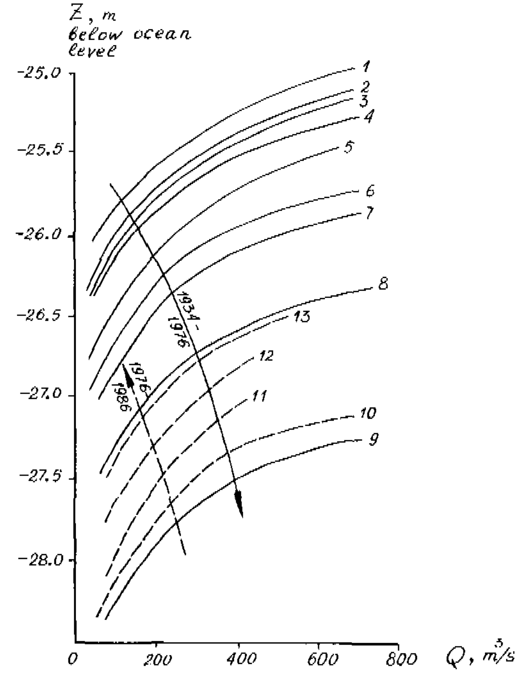

Figure 3. Stage-discharge relationships for the Sulak River near point Sulak in different years:
1 -1934, 2 - 1935, 3 - 1936, 4 - 1937, 5 - 1938, 6 - 1939, 7 - 1940, 8 - 1954, 9 - 1976, 10 - 1980, 11 - 1982, 12 - 1984, 13 - 1986.

enclosed lake comprises a high variability of water balance components. During the period 1900-1977, the input components (river water flow and precipitation) exceeded the water loss (evaporation) and the sea level dropped 3,4 m. The situation then changed and during the period 1977-1996, the water balance of the Caspian Sea became positive due to an increase in the river water flow. As from 1977, the level of the Caspian Sea began to rise. Up until 1996, the level rose a total of 2.3 m. During the period 1934-1976, the level of the Caspian Sea dropped 2.7 m, but the water level at the Sulak station in the delta (6 km from the sea) dropped only 2.3 m. This event was connected with the erosion induced by the sea level drop and a slight compensating advance of the delta into the sea. During the period 1977-1986, the sea level rise reached 1.1 m. In the Sulak River, it varied from 0.9 to 1.0 m at different water discharges. During the periods of water level instability, the elevation of the river bottom changed more or less in accordance with the water level variations (Mikhailov, 1993). Therefore Figure 3 demonstrates not only water level changes, but also the channel degradation (erosion) and aggradation (sediment accumulation) processes.

RELATIONSHIPS BETWEEN WATER LEVELS AND INUNDATION AREAS

The data concerning inundation areas in flood plains or deltas is very important, both in an economical and ecological sense. But this data can only be obtained sporadically through aerial or satellite photography. Therefore, the relationships between water levels and limited inundation data are often used for calculations. An example of such a relationship is presented in for the Volga delta in Figure 4.

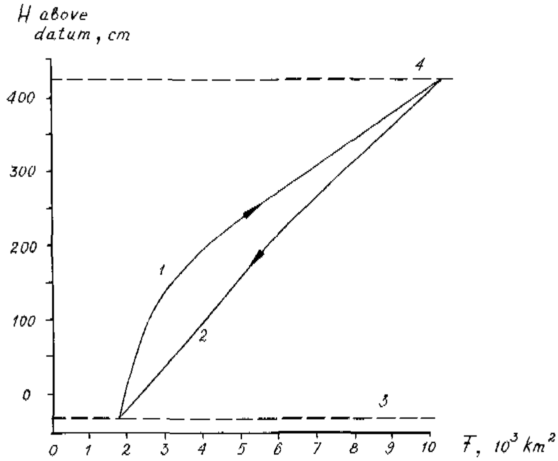

Figure 4. Relationship between the water level and the inundation area for the Volga delta. Water level *H* is observed at Astrakhan station. 1 - rising curve, 2 - falling curve, 3 - initial water level for delta flooding, 4 - maximum observed water level (423 cm, 1926).

RELATIONSHIPS BETWEEN WATER CONDUCTIVITY AND SALINITY

Water salinity (S ‰) is one of the most ecologically important characteristics of aquatic ecosystems in river mouth regions. The water salinity is usually obtained from measurements of the chloride-ion content (chlorinity) Cl ‰ or water conductivity R. The water salinity can then be calculated as follows:

$S=a\mathrm{Cl}+b$ (1)

or

$S=f(R,T)$ (2).

In the previous case, the parameters a and b are different for various seas, sea bays, estuaries, etc. For example, the following equations are used in Russia: $S=1.7984\mathrm{Cl}+0.1856$ for the Black Sea, $S=1.792\mathrm{Cl}+0{,}230$ for an open part of the Sea of Azov, $S=1.664\mathrm{Cl}+0.0294\mathrm{Cl}^2+0.263$ for the freshened near shore zones of this sea, $S=2.36\mathrm{Cl}+0.14$ for the Northern part of the Caspian Sea, etc. (Yuschack, 1973). Chlorinity Cl can be very easily estimated using the titration procedure. In the previous case, S can be calculated from the specific conductivity data which is obtained by means of a salinity meter (salinity bridge), taking into account the water temperature data.

RELATIONSHIPS BETWEEN SALINITY AND SALT COMPOSITION

As it is a conservative characteristic, the water salinity can be used for the computer processing of other conservative salt components, including pollutants. In these cases, two types of empirical equations are used for the estimation of any substance c concentration:

$c = -k_1 S + k_2$ (3)
and
$c = k_3 S + k_4$ (4)

for river-borne and sea-borne substances respectively. Here, k_1, k_2, k_3, and k_4 are empirical coefficients. For non-conservative substances, these relationships are more complicated.

RELATIONSHIPS BETWEEN WATER AND SEDIMENT DISCHARGES

The following empirical relations are used in river mouth regions: $R_s = k_1' Q^m$ and $R_b = k_2' Q^n$ in which R_s and R_b correspond to the suspended and bed sediment discharges. Here, k', m and n are empirical coefficients, whereas m=2...3, n=3...5 (Mikhailov et al., 1986).

METHODS OF SPATIAL CALCULATIONS

TASK OF CALCULATIONS

The complex of hydrological and ecological characteristics is usually observed at separate points (gauge stations, cross-sections, mooring buoys, etc.) in river, lake and sea water in river mouth regions. These observations include measurements of the water level, water and sediment discharges, current velocities, the concentration of dissolved and suspended matter, water temperature, biochemical oxygen demand, pH value, etc. In some cases, these characteristics can in part be calculated for the same observation points using the empirical methods mentioned above. Some methods have been worked out for the purpose of extending this measured or calculated data in space.

HYDRAULICAL METHODS OF WATER DISCHARGE DISTRIBUTION IN DELTAS

These methods are based on the equations of river hydraulics.
For example, in the simple case of a two-branch channel system and steady water flow, the author (Mikhailov, 1971) worked out methods based on the following set of the equations:

$$\left.\begin{aligned} \frac{Q_1}{Q_2} &= \frac{B_1}{B_2}\left(\frac{h_1}{h_2}\right)^{5/3} \frac{n_2}{n_1}\left(\frac{l_2}{l_2}\right)^{1/2}, \\ Q_1 + Q_2 &= Q_0 \end{aligned}\right\} \quad (5)$$

in which Q_0, Q_1 and Q_2 are the water discharges in the joined channel and two branches (1 and 2), and B, l, h and n correspond to the channel width and length, mean depth and Manning's roughness coefficient. Indexes 1 and 2 refer to branches 1 and 2 respectively. An iteration method of calculation was worked out for more complex delta channel networks (Yuschack, 1973). The set of the equations above (5) permits us to calculate the water discharge distribution between two branches, and the water discharge redistribution in the event that B, h, n and / or l change. For example, this method was used by the author for the assessment of the water discharge redistribution between branches of the Danube delta in reaction to the artificial deepening and straightening of some delta channels (Mikhailov, 1971; Yuschack, 1973).

METHOD OF COMPUTER PROCESSING OF STORM SURGE PROPAGATION

In addition to the rigorous hydraulic methods of computer processing of tide or storm surge propagation into rivers (McDowell and O'Connor, 1977), there are more simple semi-empirical computation methods. One of them is based on the semi-empirical exponential equation of storm surge attenuation along the river, and was developed by the author (Mikhailov, 1971):

$$\frac{\triangle \boldsymbol{H_x}}{\triangle \boldsymbol{H_0}} = \boldsymbol{exp}\,(\boldsymbol{-kx}), \quad (6)$$

in which DH_x and DH_0 are the heights of the storm surges at the distance x upstream and at the near shore; k is the parameter depending on the river discharge. This method was applied by the author for the Don and Danube Rivers (Yuschack, 1973; Mikhailov et al., 1986; Mikhailov, Morozov, 1994).

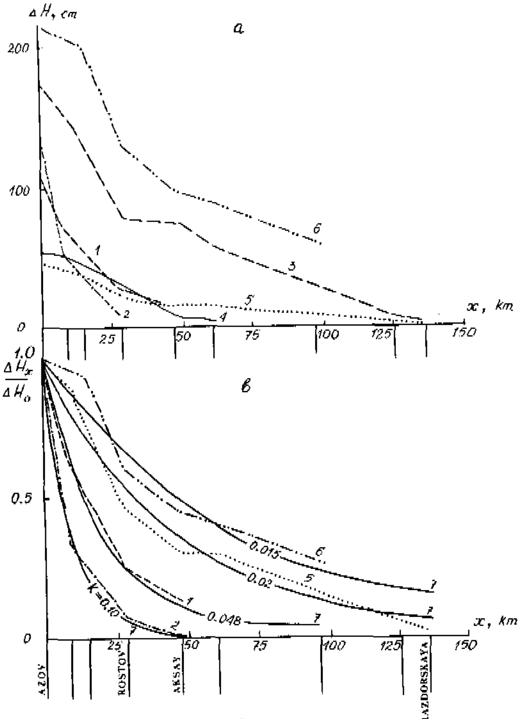

Figure 5. Curves of storm surge attenuation along the Don River: a - storm surge ranges in cm above steady water level; b - storm surge ranges in non-dimensional form.
Storm surges: 1 - March 17, 1941 (river water discharge outside storm surge extend at Razdorskaya is 2620 m^3/s); 2 - May 8, 1941 (6830 m^3/s); 3 - July 22, 1941 (581 m^3/s); 4 - April 8, 1947 (2800 m^3/s); 5 - August 31, 1950 (189 m^3/s), 6 - November 1, 1970 (695 m^3/s).
Bold lines (7) represent exponential approximations in accordance with the equation (6) with values of exponents k (damping decrements).
Vertical lines on the axis of abscissa indicate points of observations (gauge stations).

In the first case, parameter k was equal to $1.21x10^5xQ+0.11$, in which Q is the water discharge of the Don River before the storm surge. The examples of storm surge penetration into the Don River are shown in Figure5.
The length of the storm surge penetration into the river can be calculated by the equation

$$l = \frac{\ln(\triangle H/a)}{k}$$

in which a is the maximum recognisable water level upsurge in the river, for example 0,05 m. Equations such as (6) can be also used for the calculation of changes in the tide amplitude along the tidal reach of the river.

METHOD OF COMPUTER PROCESSING OF CURRENT VELOCITIES AND SALINITY SPATIAL DISTRIBUTION

On the basis of the equations for water dynamics at river mouths (Mikhailov, 1971), the equation for computer processing of current velocities along the river plume at the near shore can be obtained in the non-dimensional form:

$$\frac{V_x}{V_o} = \exp\left(\frac{kx}{h_o}\right), \quad (7)$$

in which V_0 and V_x are averaged over the cross-section water velocities in an initial (coastal) cross-section and at the distance x seaward, h_0 is the depth in the initial cross-section, k is the empirical coefficient that equals 0,002...0,004 (Fig.6).

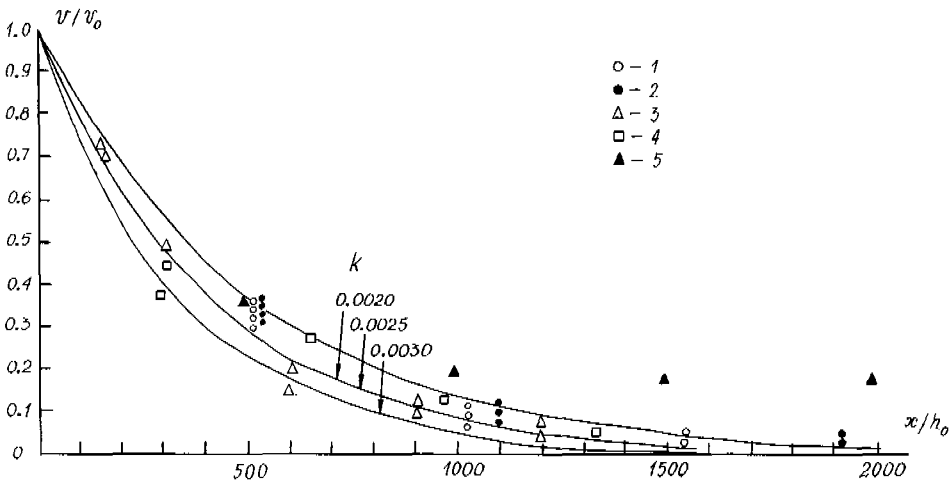

Figure 6. Current speed attenuation along the river plume at the near shore zone. Mouths of the Rivers: 1 - Kuban, 2 - Kura, 3 - Danube, 4 - Amudarya (deep near shore zone), 5 - Volga (shallow near shore zone). k is the empirical coefficient in the equation (7).

On the basis of the water-salt balance of the plume, we can obtain the equation of salinity distribution at the near shore zone:

$$S_x = S_s - \exp\left(\frac{kx}{h_o}\right)(S_s - S_r), \quad (8)$$

in which S_s, S_r and S_x - correspond to the salinity of sea, river and mixed water at the distance x from the initial (coastal) cross-section.

These methods were verified for the river plumes at the Danube, Kuban, Sulak, Amudarya, Kura. All rivers in the former USSR (Mikhailov et al., 1986).

METHOD OF COMPUTER PROCESSING OF SALT WATER INTRUSION INTO RIVERS

This very ecologically important event can be computed on the basis of the equation of advection-dispersion or the concept of densimetric velocity, $V_\rho = \sqrt{\frac{\Delta\rho}{\rho_m} gh}$, in which Dr is the difference between the sea and river water densities ($Dr=r_s-r_r$), r_m is the mean water density ($r_m=(r_s+r_r)/2$), *h* is the channel depth, *g* is the acceleration of gravity (McDowell and O'Connor, 1977; Mikhailov et al., 1986; Van der Tuin, 1991).
The concept of densimetric velocity is suitable for non-tidal river mouths and produces very good results. The main equation is as follows:

$$\frac{l_s}{h} = a(Fr_\rho)^b, \quad (9)$$

in which l_s is the length of the salt water wedge, Fr_r is the densimetric Froude number, which is equa to V/V_r, in which V and V_r correspond to the river (at the top of the salt wedge) and densimetric velocities. *The critical value of* Fr_ρ *for the onset of salt water intrusion into the channel is* equal to 1.
The above-mentioned method was verified by the author for the non-tidal mouths of the Danube, West Dvina (Daugava) and Yana Rivers (Van der Tuin, 1991; Mikhailov and Morozov, 1994). The following values of a and b were obtained for artificially deepened channels in the Danube delta: *a=270, b=2.04*. This method can be useful in the assessment of the influence channel dredging on the salt water intrusion into rivers.

METHODS OF PREDICTION

An analysis of the temporal variations in the hydrological and ecological characteristics and their prediction are the key problems in the assessment of variability and vulnerability of aquatic (eco)systems.
As an example, this problem can be considered for the changes in concentration of any dissolved non-conservative substance in the river mouth waters (Fig.1).
Let us consider the differential equation of mass balance for the non-conservative substance:

$$\frac{dm_c}{dt} = Q_r c_r + Q_w c_w + Q_s^+ c_s - Q_s c + AV \quad BVc. \quad (10)$$

Here, dm_c is the change in the mass of a given substance. It can be replaced by Vdc, in which c is the mean concentration of the substance in the waters, and V is its volume. Qr and Qw correspond to the discharges of river water and wastes (sewage), respectively. Q_s^+ and Q_s^- represent the water exchange in the strait which connects the waters in question and the sea: Q_s^- is the water flux from the water system to the sea and Q_s^+ is the opposite water flux from the sea to the water system. c_r, c_w and c_s are the concentrations of the substance in river, wastes (sewage) and sea water, respectively. It is assumed that the substance in water outflow from the water system has the same concentration c. The terms AV and BVc represent the addition and removal of the substance due to production and destruction processes, and are proportional to the water volume and the concentration c. A is the production rate [kg/(s m³)], B is the destruction coefficient [1/s]. The solution of the equation (10) is

$$c = \left(c_0 - \frac{\Sigma Q_i c_i + AV}{Q_s^- + BV}\right) exp\left(-\frac{Q_s^- + BV}{V} t\right) + \frac{\Sigma Q_i c_i + AV}{Q_s^- + BV}. \quad (11)$$

Here, c_0 is the initial concentration; $\Sigma Q_i c_i = Q_r c_r + Q_w c_w + Q_s{}^+ c_s$

The equation (11) permits us to compute or predict the changes in the concentration c in reaction to changes in the river flow, waste discharges, exchange of water with the sea, sinking or addition of the substance, etc.

Furthermore, the equation (11) also allows us to obtain the formula for the time period of changes of the substance concentration from c_0 to c_t :

$$t = \frac{V}{Q_s^- + BV} \ln \frac{C_0 (Q_s^- + BV) - \Sigma Q_i c_i - AV}{C_t (Q_s^- + BV) - \Sigma Q_i c_i - AV} \quad (12)$$

For c_t, one may choose any concentration, including the ecologically "critical" concentration. In this case, the corresponding value of t is the period of time in which the aquatic system is resistant to the given substance.

The above-mentioned mathematical model is universal and can be used for any non-conservative or conservative dissolved substance. The first include phosphorus, some trace metals and pollutants, gases, phytoplankton and others, the second includes water salinity, for example. This model can be applied in the case of both an increase or decrease in some substance content. The author used the model to predict the water quality in a number of Russian river mouth regions.

The application of the above-mentioned methodology for water salinity changes in semi-enclosed river mouth bays provides the following equations for the case of negligible fresh water salinity:

a. for the desalinisation stage (initial salinity S_0 is more than salinity of river water and sea salt water does not penetrate into the bay):

$$S = S_0 \exp\left(\frac{Q_r}{V} t \right), \quad (13)$$

$$t = \frac{V}{Q_r} \ln \frac{S_0}{S_t}; \quad (14)$$

b. for the salinisation stage (initial salinity S_0 is less than sea one, sea water penetrates into the bay):

$$S = S_0 + \frac{Q_s^+ S_s}{V} t, \quad (15)$$

$$t = \frac{(S_t - S_0) V}{Q_s^+ S_s}. \quad (16)$$

Here, S_t is the reference salinity, which is chosen in accordance with the research task. One can see that desalinisation occurs in accordance with the exponential law, but salinisation follows the linear law.

CONCLUSIONS

The complex of empirical, semi-empirical and theoretical methods and models was worked out as an essential part of the assessment of river mouth water resources. These methods permit us to extend observation data at separate points in space and to compute the spatial and temporal variability of hydrological and ecological characteristics. Some of the methods and models men-

tioned above can be widely used as an essential part of the monitoring system. A monitoring system of this kind has to consist of four units.
The first is the observation unit, which includes the complex of hydrological, hydro-chemical, ecological and meteorological observations at the permanent network of hydro-meteorological stations, and a data bank.
The second is the assessment and computer processing unit based on the complex of the empirical and semi-empirical methods extending monitoring data from the separate points to the whole water system. Also, some methods allow us to compute unknown characteristics, or characteristics which are difficult to measure, from known or easily measured characteristics. The third is the prediction unit, of which the purpose is to forecast changes in hydrological and ecological characteristics using simple tendency methods or mathematical models like those that were considered above in this paper. Finally, the goal of the fourth unit is extending the data and results of the previous units to the regional administration and water management or protection organisations. At present, this monitoring system is widely working in Russia, for example in the Volga delta (Mikhailov, 1994). The above-mentioned methods were used for the study of river mouths in Russia and to establish a monitoring system.

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MONITORING AND ASSESSMENT IN RIVERS BASED ON TWO-DIMENSIONAL MODELS

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ABSTRACT

Many data concerning water quality and quantity are collected in rivers in order to understand the complicated physical processes in rivers, to be informed about their physical and chemical state, and to detect in time the occurence of disasters and undesired development. As illustrated by this paper two-dimensional mathematical models are a reliable tool to assist engineers in providing these data.

It is the objective of this paper to demonstrate what insight in the flow physics and hydromechanics of rivers can be provided by these models and how this information can lead to a reduction in monitoring. The theoretical ideas are illustrated by the following examples:

- *monitoring of water surface and flooded area during flood events*
- *monitoring of the annual distribution of water depth and flow velocities on floodplains for the assessment of habitat development*
- *monitoring of pollution spreading on flood-plains due to point waste-water emission*

Despite its rationalization effect on monitoring systems, mathematical models still need a considerable amount of field data. The boundary conditions must be provided by monitoring systems and the empirical parameters of the model have to be approved by a model calibration in the real flow situation. In the conclusion of the paper some general guidelines are given on the design and operation of monitoring systems supported by two-dimensional mathematical models.

INTRODUCTION

River valleys cut through our countries like lifelines. Uninfluenced by mankind they develop aquatic, amphibic and terrestric habitats with unique diversity of flora and fauna. On the other side due to their fertile soils and good suitability for trade routes, they are preferred areas of settlement. With the rise of industrialization and increase of population the conflict of interest between mankind and nature was intensified.

Today anthropogenic changes in river basins are only permitted if they are without negative influence on the environment. The ecological state of a river system is largely dependant upon the hydrodynamical, morphological, and hydrochemical state of that river system.

Therefore the effect of anthropogenic changes can only be estimated if the complex physical and chemical processes in rivers are well understood. Consequently, at least the following parameters need to be recorded in river systems:

- water table and boundaries of flooded area
- time-dependent discharge along the river reach
- velocity distribution within the cross section of river and flood-plain for various flow situations ranging from low flow to flood
- deposition rate of suspended and bed load especially in stillwater areas, e.g between groins, at naturally shaped river banks, in secondary channels and on flood plains
- rate of erosion and deposition at the river bed and banks
- concentrations of chemical parameters like oxygen, nitrate, phosphate and sulphate, heavy metal etc.
- composition of bed and bank material, vegetation on bank and flood-plain

In Central Europe with its intensive use of water resources, the installation and operation of a monitoring system as mentioned above is a very large effort. Thus for many German agencies it is a major concern to reduce this monitoring effort without lack of information.

For more than twenty years multi-dimensional hydrodynamic flow models have been developed and applied to study the flow in rivers (King, 1976; Rodi, 1980; Pasche, 1984, Stein, 1990; Wenka, 1992). In the first decade these models were mainly restricted to scientific application. However, they have considerably been improved and the necessary computer performance can even be provided on personal computers. Nowadays they are accepted as a reliable and fast applicable tool for surveying flow-related problems in rivers (Pasche, 1989; Zielke et al., 1988; Delft Hydraulics 1992; Hall 1987). However, the potential for monitoring purposes has nearly been undiscovered, yet. Thus it is the objective of this paper to demonstrate in what way two-dimensional flow models can be used for monitoring purposes and to what extent they can reduce the monitoring effort or give insight into the flow physics which cannot be provided by monitoring systems with affordable effort.

Despite being an advanced technique a 2d-flow-model cannot be used as a black box tool without being aware that the technique relies on a model which is based on many simplifications of the real world and thus need expertise to set up the numerical grid, to evaluate the model parameters and finally to establish the limits of application. Therefore a theoretical excursion on 2d-flow models is preceding this paper in which the governing equations, introduced assumptions and numerical specialities which are needed to get realistic results, are presented.

Even with flow models those monitoring systems do not become useless as for the calibration of mathematical models still a considerable amount of field data are needed, e.g. topography, roughness conditions, discharge and parameters of the flow. But the data to be recorded differs considerably from systems without model support. Therefore at the end some general guidelines are given in the conclusion of this paper on the design and operation of monitoring systems which are supported by flow models.

THEORETICAL BASIS OF TWO-DIMENSIONAL FLOW MODELS

The governing equations of two-dimensional flow models are based on the depth averaged-shallow water equations which assume hydrostatic pressure over the water depth and neglect verti-

cal components of the flow velocity (Vreugdenhil/Wijbenga, 1982). Applying tensor notation and Einstein's summation convention they can be written in the following compact form:
continuity equation:

$$\frac{\partial h}{\partial t} + \frac{\partial hu_i}{\partial x_i} = 0$$

momentum equation:

$$\frac{\partial(h\bar{u}_i)}{\partial t} + \bar{u}_j h \frac{\partial \bar{u}_i}{\partial x_j} =$$

$$-gh\frac{\partial(a_0 + h)}{\partial x_i} + \frac{\partial}{\partial x_j}\frac{1}{\rho}\int_{a_0}^{a_0+h} \tau_{ij} dx_3 + \frac{1}{\rho}\tau_{o,i}$$

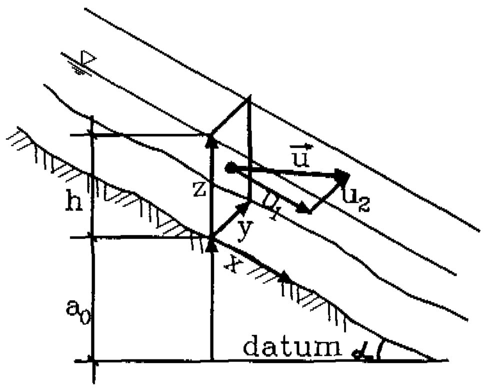


Figure 1: Definition sketch

with i,j = 1,2 (longitudinal (x) and lateral (y)-axis), h = water depth in [m], $\bar{u}_i$ = dept-haveraged velocity components in [m/s] (u_1 = longitudinal and $\bar{u}_2$ = lateral velocity component), a_o = height of the bed in [m], g = gravity in [m/s²], ρ = specific weight of water in [Ns²/m⁴], t = time in [s], $\tau_{o,i}$ = bed shear stress in [N/m²], τ_{ij} = interial shear stresses in [N/m²]

In this equation the influence of wind shear stress and Coriolis acceleration is neglected as they are of minor importance for the flow in inland rivers. Despite of these assumptions the equations are not directly solvable. Further assumptions have to be made on the bed shear stress $\tau_{o,j}$ and the interial shear stresses τ_{ij}:

- bed shear stress τ_{oi}

There exists a dependency between the flow velocity and the bed shear stress which is governed by the quadratic resistance law.

$$\tau_{oi} = \rho \cdot \frac{\lambda}{8} \cdot \bar{u}_i \cdot \sqrt{\bar{u}_i^2 = u_j^2} / \cos \alpha_i$$

with i,j = 1,2 $i \neq j$, λ = friction factor in [-], $\alpha_i = \arctan\left(\frac{\partial a_{oi}}{\partial x_i}\right)$

The friction factor λ is dependent on the structure of the bed roughness and the flow depths. For its quantification many relationships exist. The most popular is the Mannings-Formula. Due to its better physical basis in Germany the Darcy-Weisbach equation is nowadays preferred (DVWK, 1991). Both formulas describe the friction effect due to bed and bank material in the river and on the flood-plain. However in natural rivers the flow is also affected by vegetation on the bank and flood-plain. In the case of non-submerged vegetation a resistance law developed by Pasche (1984) is recommended for it relates the friction factor λ to directly determinable geometric vegetation parameters (diameter, longitudinal and lateral distance of non-submerged vegetation).

For submerged vegetation like grass the resistance is further influenced by the stiffness of the material. Kouwen (1990) set up a resistance formula, which takes this into account and evaluates the friction factor of submerged grass in dependence on vegetation height and stiffness. Despite these improved methods to evaluate the bed shear stress in rivers the empirical coefficients should be verified by observation for the high degree of irregularity especially in natural rivers makes geometric parameter evaluation uncertain.

- shear stress tensor τ_{ij}

The *interial shear stresses* τ_{ij} *are composed of molecular, turbulent and dispersive stresses* and are modelled on the basis of Newton's law:

$$\tau_{ij} = \left[\underbrace{\upsilon_m \left(\frac{\partial \bar{u}_i}{\partial x_j} + \frac{\partial \bar{u}_j}{\partial x_i} \right)}_{\text{molecular}} - \underbrace{\overline{u'_i u'_j}}_{\text{turbulent}} - \underbrace{\int_{a_0}^{a_0+h} (u - \bar{u})_i (u - u)_j dz}_{\text{dispersive}} \right] = (\upsilon_m + \upsilon_t + \upsilon_d) \left(\frac{\partial \bar{u}_i}{\partial x_j} + \frac{\partial \bar{u}_j}{\partial x_i} \right)$$

The evaluation of the *molecular viscosity* υ_m *is straightforward because it is only dependant on* the state of the material. While the turbulent viscosity υ_t considers the effect of generation, transport and decay of vortices, the dispersive viscosity υ_d is due to the depth averaging of the governing equations and considers the deviation of the vertical velocity profile from the logarithmic law due to secondary currents. Especially in natural rivers the turbulent and dispersive viscosity is several orders higher than the molecular viscosity (Stein, 1990). Thus a reliable evaluation of these parameters is of great importance. However their estimation is difficult because they are dependant on the state of the flow, and despite intensive research this dependancy is not fully understood especially in flow situations where the dispersion is dominating (e.g in meandering rivers and compound channel flow). Rouvé/Schröder(1993) show that *in these flow situations* even the most sophisticated turbulence models, like the k -model in which the eddy viscosity υ_t is related to the turbulent kinetic energy k and the rate of dissipation ϵ, don't provide better results than the simple approach of constant eddy viscosity. But experience is necessary for the evaluation of the eddy viscosity parameters. Otherwise they must be validated by velocity measurements.

MODELLING OF WATER QUALITY AND SEDIMENT TRANSPORT

Two-dimensional flow models can be extended to simulate the water quality and material transport in rivers by adding an additional transport equation to the set of governing equations which is called the *advection/diffusion-equation:*

$$\frac{\partial c}{\partial t} + ui \cdot \frac{\partial c}{\partial x_i} = \frac{1}{h} \cdot \frac{\partial}{\partial x_i} \left(h \cdot E_{ij} \cdot \frac{\partial c}{\partial x_j} \right) \pm S$$

with i,j=1,2; c = depth-averaged material concentration, E_{ij} = longitudinal and lateral dispersion coefficient.

By this equation the transport of material is divided in advective transport, diffusive and dispersive mixing. The rate of reaction of material due to chemical or biological processes is considered by the source/sink-terms. For the quantification of this term mathematical relations are introduced which describe the physical processes of settling, mobilisation, chemical and biological reaction. An extensive summary of the state-of-the-art in water quality modelling is given in Zielke et al. (1996).

Similar to the turbulent and dispersive viscosity parameters the dispersion coefficients are no real constants but dependent on the flow structure, with the longitudinal dispersion one order larger than the one perpendicular to the main flow. For their evaluation it is referred to Fisher et al,1979. However, it is recommended to verify these parameters by measurements.

In most cases the mixing process and flow structure are not influencing each other that the flow equations and the advection/diffusion-equation can be solved independantly. But in case of density stratification the flow equations and the advection/diffusion-equation have to be solved simultaneously.

NUMERICAL ASPECTS

The shallow-water equations and the advection/diffusion-equation are not directly solvable for real flow situations. Therefore numerical solution methods have to be applied by which the flow domain is divided into simple geometric elements, e.g. triangle, quadrangle (see fig.2). For these elements, the differential equations can be transformed into a set of algebraic equations which can be solved for each element. The Finite-Element-Technique, the Finite-Difference technique and the Finite-Volume technique are used in this connection. With respect to the discretisation of the flow domain the Finite-Element and Finite-Volume technique are most flexible which is most important for modelling the flow in rivers. Thus, they are the preferred numerical technique although the necessary computer resources are much higher than for the Finite-Difference technique.

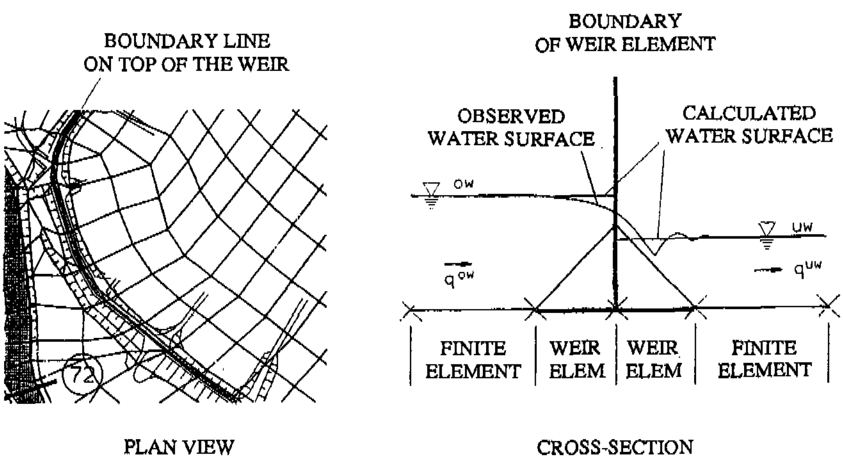


Figure 2: Discretization of a weir by a 'weir boundary element'

For the closure of the set of equations boundary conditions have to be defined. This includes the definition of the topography at each grid node. At the boundary of the flow domain the following three different boundary conditions are usually introduced:

- upstream boundary: specific discharge at each node
- downstream boundary: water depth at each node
- longitudinal boundaries: slip condition with no velocity normal to the boundary

As the water depth is not known at the beginning of the calculation, the longitudinal boundary of the flow domain is not found before the end of the simulation. Thus, the moving of the boundaries has to be reproduced by the numerical method. For engineering purposes often the simple method of eliminating dry elements and including wet elements in the system matrix is applied. But usually the boundary elements are partially wet. The full elimination of these elements can cause an inaccurate flow simulation especially for coarse grids or in areas of high lateral velocity gradients. Adaptive grids which exactly follow the wetted boundary would solve this problem. However, due to the topographic irregularity which is especially in natural rivers dominating the development of an algorithm for adaptive grids is not straightforward and so far has not been satisfactorily accomplished. Therefore the authors took up an idea of King/Roig (1988) which introduced a marsh element" for partially wetted elements. This element is not eliminated while

partially wetted, but a marsh flow is assumed for the dry part of the element which is governed by the two-dimensional Darcy-equation.
For more details see King/Roig (1988) and BCE (1996).

Often the flow in rivers is controlled by structures like weirs, bridges and conduits. Due to pressure flow, change of the flow regime and steep water surface gradients within the structure, no hydrostatic pressure distribution over the water depths occurs and the vertical velocity component is not negligible. The application of the flow equations can lead to a considerable underestimation of the upstream water table of the flow structures. Deviations of more than 20% were observed by the authors.

By combining a 2d-flow model with a 3d-flow model in which the flow area around the structure is described by the three-dimensional Navier-Stokes-equation, the model quality can be improved. However, in general for engineering problems such a coupled 2d/3d-flow model would be too expensive. But often no flow details around the structure apart from the flow resistance are wanted. Therefore the authors developed special boundary lines and elements which covers the whole control structure or obstruction. The discharge through these boundary lines and elements is defined as a function of the upstream and downstream water table, and the shape of the obstruction and control structure respectively. For many standard gates and weir shapes these functions are known. Only for special structures these function must be derived by physical models or field observation.

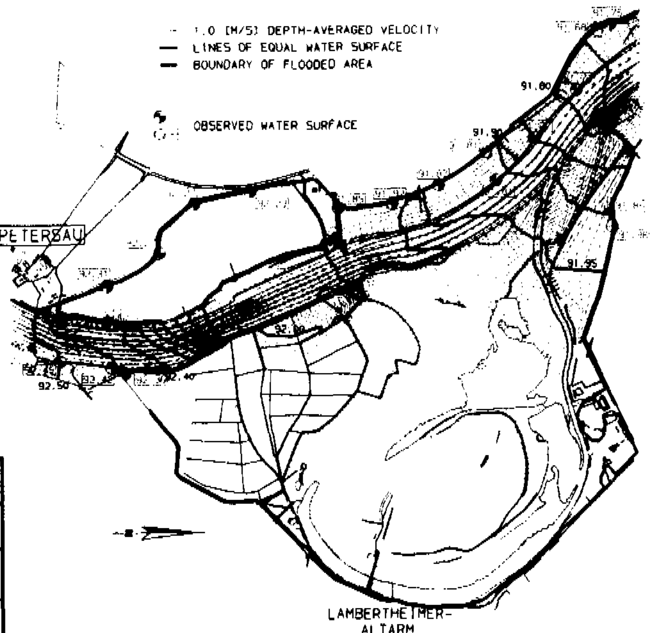
APPLICATION OF 2D-FLOW AND WATER QUALITY MODELS

2d-flow and water-quality models cover the whole flow domain and thus give the spacial distribution of all state parameters in dependence of predefined boundary conditions. In the ideal case a combined application of a monitoring system together with the 2d-flow model requires only the monitoring of the state parameters along the boundary of the flow domain. However, this presumes a precise simulation within the whole model area. Due to the uncertainty in the evaluation of the empirical coefficients, the underlying model assumptions and the mapping of the continuous terrain into a grid of discrete points, the calculated values can considerably differ from reality. Consequently, flow models need to be calibrated and verified on the basis of recorded flow parameters inside the flow domain. Once a model is calibrated, the monitoring effort can often be cut back, being only a fraction of the monitoring effect without the support of 2d-flow models.

Out of more than ten years experience of 2d-flow modelling three illustrative examples are subsequently presented in order to demonstrate how a monitoring system can benefit from 2d-flow models.

FIRST EXAMPLE: RECORDING OF FLOOD STAGES AND FLOODED AREAS

In order to estimate the flood risk and to dimension flood protection facilities like dikes and walls, the water surface and the extension of flooded areas are recorded along many rivers. For example at the river Rhine, in a distance of 500 meters, the water surface on both sides of the river have been recorded along the German reach of the river Rhine during flood events. Even areal pictures have been taken at extreme flood events to document the boundaries of the area affected by the flood. In the meantime, extensive data records have been available. They have been statistically analyzed to estimate the risk of flood and to check the risk of flooding for the dikes. Despite this good data basis, still uncertainty existed in the evaluation of the flood risk of the dikes. Due to the complicated flow situation in the upper river Rhine where the river still passes extreme meander bends and flows through secondary channels and wide flood plains during flood, leading to complex two-dimensional and even three-dimensional flow structures, the recorded flood stages cannot be extrapolated with certainty to the design flood. Consequently, the government of the German states Rheinland-Pfalz and Baden-Württemberg decided to apply a 2d-flow model to simulate the design flood for the river Rhine between the French border and the city of Mainz. A section of this model is presented in fig. 5 and shows the river Rhine and its flood-plains northwards of Ludwigshafen. On the basis of the recorded water surface of the flood event in 1988, the empirical parameters of the model were calibrated.



	river	flood plains
k_s	0,1 m	0,5 - 0,7 m
d_p a_x , a_y		0,5 - 5,0 m 3,0 - 15,0 m
v_w	1,0	1,2 m²/s

Figure 3: Flow velocity and water surface of the river Rhine at Ludwigshafen

With the empirical parameters of fig. 3 the model reproduced the observed water surface with a standard deviation of = 5.2 cm and a confidence range about ±11 cm (tab.1). Thus the model quality stayed within the confidence range of the observed flood stages. The empirical parameters adapted physically realistic values showing that the calibration was not a pure curve-fitting procedure but lead to a physically well-founded model. As the transport characteristic of suspended and bed load does not considerably change during flood, the model, calibrated at a twenty-year flood, could be applied without any change of paramters to the design flood, which is a 200 year-flood. By the model results the uncertainty in the design of the flood protection facilities could be removed. In the future, on the basis of this model, the monitoring of water stages can be considerably reduced. On the other side, areas could be detected by the model where the monitoring net of water stages should be refined in order to analyze the reasons for the observed deviations of recorded and calculated water surface.

WATER TABLE

	left bank			right bank		
Rhine-kilometer [km]	observed [NN+m]	calculated [NN+m]	difference [cm]	observed [NN+m]	calculated [NN+m]	difference [cm]
402.00	96.58	96.55	-3	96.52	96.55	3
403.00	96.42	96.41	-1	96.45	96.38	-7
404.00	96.25	96.20	-5	96.33	96.24	-9
405.00	96.10	96.12	2	96.10	96.13	3
406.00	96.00	96.01	1	95.96	96.03	7
407.00	95.94	95.95	1	95.90	95.95	5
408.00	95.79	95.79	0	95.83	95.81	-2
409.00	95.74	95.68	-6	95.76	95.75	-1
410.00	95.63	95.57	-6	95.63	95.59	-4
411.00	95.60	95.49	-11	95.46	95.46	0
412.00	95.34	95.27	-6	95.35	95.28	-7
414.00	94.99	94.97	-2	95.13	95.04	-9
415.00	94.87	94.83	-4	94.96	94.92	-4
416.00	94.81	94.73	-8	94.86	94.80	-6
417.00	94.72	94.65	-7	94.68	94.69	1
418.00	94.54	94.59	5	94.56	94.61	5
420.00	94.37	94.38	1	94.34	94.34	0
421.00	94.27	94.29	2	94.25	94.24	-1
422.00	94.08	94.10	2	94.21	94.12	-9
423.00	93.91	93.90	1	94.11	94.00	11
424.00	93.84	93.81	-3	93.84	93.88	4
425.00	93.68	93.74	6	93.79	93.78	-1
426.00	93.64	93.70	-6	93.61	93.67	6
427.00	93.52	93.59	7	93.54	93.57	3
428.00	93.42	93.50	8	93.42	93.52	10
429.00	93.36	93.36	0	93.36	93.35	-1

Table 1: Comparison between observed and calculated water tables of the flood event 1988 at the upper river Rhine

SECOND EXAMPLE: BIO-MONITORING

In many districts of Germany bio-monitoring programs are set up to improve the picture of the ecological state of rivers. Due to a strong correlation between the flow physics and the ecological state, biologists and ecologists are asking for detailled information on the flow to understand the present habitat structure, to estimate the future develoment and to design restoration methods. The desired information on the flow covers in general the following parameters:

- the annual fluctuation of the water surface and local flow velocities for a mean year
- the probability of flooding for the area adjacent to the river
- the annual duration of flooding for every section on the flood-plain
- the event-dependant duration of flooding for every section of the flood-plain

In general, the effort to record these parameters along the whole river reach cannot be raised and 2d-flow models turned out to be more suitable for this task. This will be demonstrated by an example of bio-monitoring at the river Lippe which is located northwards of the Ruhr Area. The investigated 15 km long reach of this river is characterized by a meandering main channel with uniform cross-section and cobble-stones at the banks and bed. Only little vegetation is to be found alongside the river. The topography of the flood-plains varies considerably with many permanent water bodies on the flood-plain.
At the beginning, the intention was to monitor all necessary flow parameters so that several

measuring devices were installed on the flood-plain. However, soon it became evident that the necessary effort to take data could not be provided and only at few locations the water surface was recorded. This information was sufficient to calibrate the 2d-flow model.

Beside the depth-averaged mean velocities and the water surface distribution in the flow domain, for the first time the annual duration of flooding for the whole flood-plain could be derived. In a first step the water stages and flow velocities were calculated for different flow situations ranging from mean flow to 250-year flood. On this basis, for each grid node stage-discharge and stage-velocity relationships were set up (fig. 3). By these relationships the recorded annual discharge duration line could be converted into annual flood duration, days-stage and days-flow-velocity lines for each grid node. This point information was transferred in lines of e.g. equal flood duration. Visualizing this result in a map, a good impression on the spacial distribution of annual flood duration is obtained (fig. 4). The same technique was applied to estimate the duration of

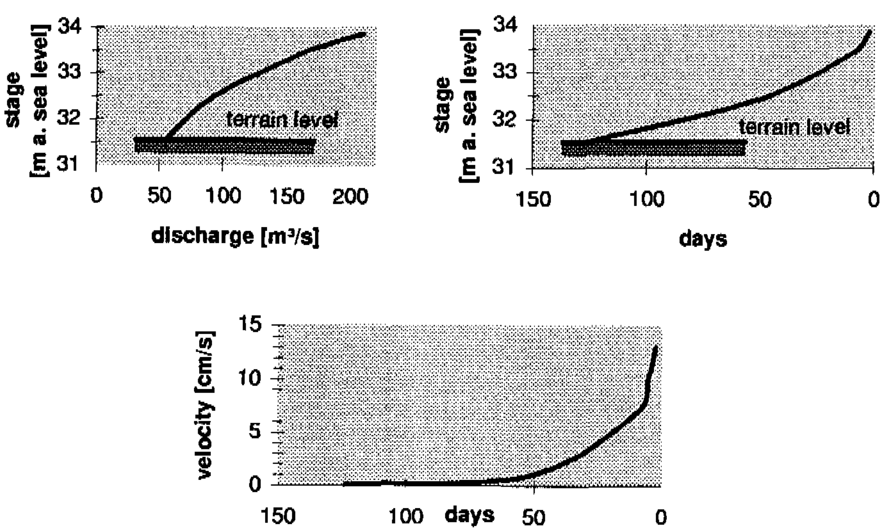


Figure 4: Stage-discharge relationship, velocity-days-, and stage-days-distribution for a grid node on the flood-plain and for a mean year

flooding for single flood events. While the annual flood duration map is used by biologists and ecologists to understand the present habitat conditions, the flood event-related duration map can serve for the assessment of habitat damage caused by a single flood event.

THIRD EXAMPLE: WATER QUALITY MONITORING

The possible deposition of suspended material in stillwater areas (e.g. between groins, at natural-shaped river banks, in secondary channels and on flood-plains) has a considerable influence on the ecological state of a river. Neither the influence on the self-cleaning effect by deposition nor its possible remobilisation by erosion is fully understood, yet. Measurements of deposited suspended solid and its chemical composition would be a method to fill this gap in knowledge. However, this monitoring is difficult, time-consuming and will only give information on selected points in the flow domain. Hydrodynamic water quality models would be a more reliable and less expensive tool. This will be made clear by a model application at a 10 km reach of the upper river Rhine which is a section of the model area of the first example. In this river reach the water quality is affected by the discharge of a regional waste-water treatment plant

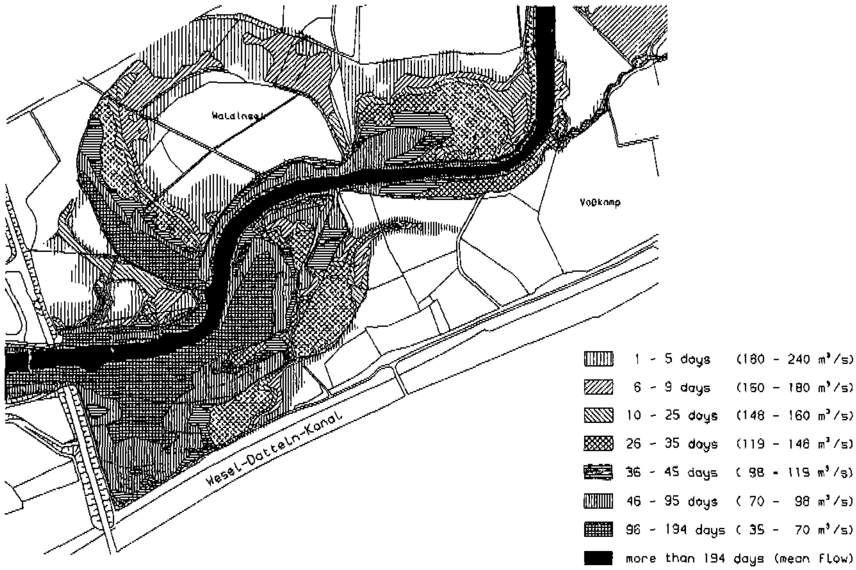


Figure 5: Calculated annual duration of flooding

with maximum discharges of 15 m³/s. For the management of the flood-plains the government required information on the transport and mixing of the discharged waste-water especially during flood, when the total discharge is between 3000 and 6000 m³/s. The monitoring of material concentration rates on the flood-plain was given up for being too expensive. Therefore, the 2d-water quality model described in chapter 2, was applied supplementary to a 2d-flow calculation.

The dispersion coefficient $E = e_{T/l} \cdot u^* \cdot h$ of the advection/diffusion equation was taken from Elder (1959) who recommends the following values:
$e_T = 0.23$ and $e_l = 5.93$

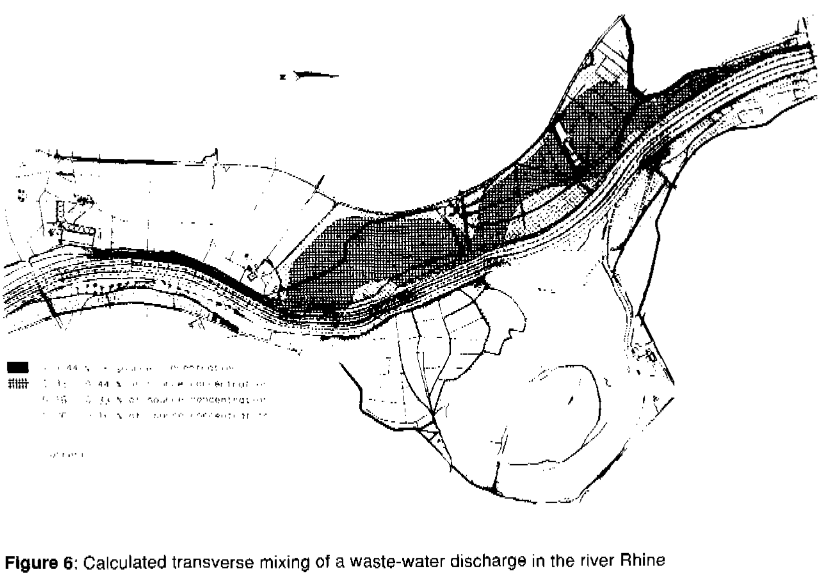


Figure 6: Calculated transverse mixing of a waste-water discharge in the river Rhine

With respect to the longitudinal dispersion this value is in good aggreement with observations of Fisher et al (1979). However, for lateral dispersion in rivers Fisher et al., (1979) recommend higher values ranging between e_T=0.3 and 0.8. But the low value of Elder was taken, for. Grimm-Strele, 1983 measured lateral dispersion coefficients between e_T=0.15 and 0.30 in the river Rhine close to this model section.The reason for this low lateral disperion is to be seen in a rather regular bed size, smooth banks, nearly no groins, and a low degree of meandering.

The reaction of material was not considered in the study so that the source/sink term in the advection/diffusion-equation could be neglected. On the basis of the water quality simulation it could be shown that due to little lateral mixing a material discharge on the left bank will only slowly spread over the whole cross-section. Only at the downstream end of the investigated flow domain the discharged material reached the other side of the river and thus did not effect the flood-plain on this side of the river. However, on the side of the discharge the flood-plain receives most of the discharged material and thus would be considerably affected by deposition of transported material.

CONCLUSION

2-dimensional flow and water quality models can be evaluated as a reliable tool for monitoring purposes. However, these models must fulfill at least the following quality standards:

- solution of the complete shallow-water equation
- evaluation of the turbulent shear stresses by the Boussinesq approach and assumption
- of constant eddy viscosity
- the flow resistance at structures must be considered by special boundary elements
- partly wetted boundary elements need special treatment.

Especially for natural rivers with vegetation on the flood-plain and bank the Darcy-Weisbach friction factor is advantegous compared to the Manning's parameter, for relationships exist by which the friction factor can be evaluated on the basis of directly determinable geometric parameters. More sophisticated aproaches to evaluate the eddy viscosity like the k -turbulence model or algrebraic equations which correlate the water depth and bed shear stress to the eddy viscosity were not found to improve the prediction of the model.

As 2d-flow models rely on empirical parameters, a calibration of the model by observation is a must. When only the flow is modelled, the measurement of the water depth at several locations within the flow domain is sufficient. Different flood events are necessary to show the depedance of the empirical parameters on the water depth.Velocity measurements would improve the understanding about the dependance of the eddy viscosity on the flow situation. In case of water quality modelling the dispersion coefficients are difficult to evaluate especially for flood conditions so that they should be quantified by measurements. Moreover, our knowledge of many sediment and material transport processes is insufficient so that the empirical relationships need to be checked by concentration measurements.

Once a model is calibrated it can considerably reduce the effort of monitoring. This was proved for the following monitoring tasks:

- flood stages in river and on flood-plain

- flooded area
- duration of flood-plain flooding for each single flood event and for the whole year
- distribution of a point-source pollution in the near and far field of the river.

The possibilities of applying 2d-flow models for monitoring purposes are by far not fully utilized. Especially in navigatable rivers the geometry is still regularly surveyed to detect areas of erosion and sedimentation. Although the modelling of these processes is much more complicated than the modelling of the flow, today's morphologic models are qualified enough to cut down this monitoring effort. One-dimensional water quality models are already applied in rivers to assist monitoring (IKSR/KHR 1991). Especially where two-dimensional flow effects are dominating the 1d-models are difficult to calibrate and need special extensions to take care on the material-trapping effect (Pasche/Lippert,1995). Obviously coupled 1d/2d-water quality models would be more advantegeous. Therefore, their application is recommended and future activity of model based monitoring should aim at the devolopment of such water quality models.

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PERFORMANCE EVALUATION OF WATER QUALITY INFORMATION SYSTEMS

A QUANTITATIVE COMPARISON OF TWO WATER QUALITY MONITORING PROGRAMS

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ABSTRACT

Useful water quality information systems exhibit a competent system design process, comprehensive documentation of system design and operation, and a routine performance evaluation process. The Framework for Evaluating Water Quality Information System Performance introduced here is an instrument which can help water quality managers and information systems designers meet these criteria. The Framework has four major phases: (1) evaluation planning, (2) watershed and management system analysis, (3) information system analysis, and (4) information system performance evaluation. Application of the Framework and its utility are demonstrated in the evaluation of water quality monitoring programs associated with a unique municipal water transfer project .

INTRODUCTION

The application of systems concepts to water quality information programs has resulted in program designs increasingly focused upon satisfying management objectives and information needs. A recent example of this trend is the development of data analysis protocols to enhance information system designs (Adkins, 1993). Further work is necessary to develop information system design approaches appropriate to the future water quality management realm where technical questions will become complicated by social, economic, political and legal considerations. That environment will demand actively managed and continuously improved information systems which are closely linked to water quality management objectives and decisions.
Evolution of Water Quality Information Systems Design
Water quality information systems have evolved through several phases:

- No formal water quality data collection or information generation

- Water *Quality Monitoring Networks:* Early twentieth-century water quality data collection networks were designed to collect data to answer narrow technical questions with respect to limited areas and limited time periods. Over time, monitoring network designs embodied improved data collection and storage techniques, and an increased appreciation of information needs and the stochastic nature of water quality. Many researchers contributed to this network development process, including Velz, Clark and McKee in the 1950s and 1960s, and many USGS, USEPA, and academic researchers in the 1970s and 1980s.

- Water *Quality Monitoring Systems:* The 1980s witnessed continued development of data collection and storage methods, refined statistical characterization of water quality variable behavior, and an emerging appreciation of the need to design water quality networks according to systems principles. The distinction between data and information was emphasized. Systematic design procedures developed included a 12-Step Process (Sanders et al., 1979), a 5-Step Process (Sanders et al., 1983), a formal (4-Task) Systematic Design Procedure (Mar et al., 1986), the New Zealand National Network design (Smith et al., 1989), " Water Quality Monitoring System Following the Flow of Information" (Ward, 1988), and "Wheel and Axle" Frameworks (Payne and Ford, 1988; Ward et al., 1990).

- Water *Quality Information Systems:* The late 1980s and early 1990s witnessed continued development of data collection and statistical analyses, and of frameworks to guide the design of more comprehensive water quality information systems. Increasingly, their potential contribution to broad water quality management systems and ecological management systems has been recognized. Examples of these efforts include a "Framework for Designing Water Quality Information Systems (Ward et al., 1990) and the development of "Data Analysis Protocols" (Adkins, 1993).

CURRENT STATUS OF WATER QUALITY INFORMATION SYSTEM DESIGN

Even as research advances, traditional problems and shortcomings continue to plague water quality information programs. Often, the "data-rich, information poor" syndrome is evident; i.e., where data collection has become an end in itself, either because data are not converted to information (i.e., not used) or are not related to an information need (i.e., not useful). Also, the cost of monitoring, especially sampling and laboratory expenses, is escalating rapidly.

Recent water quality management trends have raised a number of concerns:

- Water quality managers will face fewer decisions based only upon technical criteria and more decisions which also involve non-technical, qualitative considerations (e.g., social, economic, legal, and political factors).

- Water quality managers will be held more accountable (by the public, Congress, and watershed stakeholders) for the operation of their management systems and the contribution of those systems to the actual quality of the water.

- Resources available for water quality management purposes will become increasingly constrained and uncertain as regulatory philosophy is reviewed and as federal and state agencies, municipalities, and private industry all try to operate under reduced budgets.

- Large system perspectives (e.g., watersheds, river basins, ecosystems, the USGS's NAWQA program, the USEPA's EMAP program) and complex problems will call for integration and coordination of systems, more information sharing, and more participative, open and diverse decision making processes (e.g., the Intergovernmental Task Force on Monitoring Water Quality and Florida's statewide ambient monitoring program).

To meet public water quality goals and corresponding management objectives, water quality managers will require more effective and efficient information systems. Water quality monitoring and information systems must become Water Quality Management Information Systems. As management information systems, these programs must: (1) derive from watershed goals and associated water quality management system objectives, (2) satisfy expanded water quality management objectives and information needs, (3) link directly to specific water quality management decision needs and decision processes, and (4) be dynamic; i.e., designed to be continuously reviewed and improved.

FRAMEWORK FOR EVALUATING WATER QUALITY INFORMATION SYSTEM PERFORMANCE

The performance evaluation approach described here can be used to guide the design of water quality management information systems as well as to analyze and redesign existing systems. The Framework integrates concepts drawn from sources in several fields of study, including: (1) general systems theory, (2) management systems, (3) information systems, (4) water quality monitoring and information systems, and (5) system performance measurement.

Attributes of a useful performance measurement process include (adapted from Sink, 1985):

- Validity - Are we really measuring what we think we are?
- Accuracy and precision - Does the measurement process faithfully and consistently track the behavior of interest?
- Completeness - Are all relevant performance measures included?
- Uniqueness - Are redundant or overlapping measures avoided?
- Reliability - Are measurement results consistently valid? Are error levels known and minimized?
- Comprehensibility - Are measures as simple as possible to convey the intended message? Does the measurement process match the user's skills and knowledge?
- Quantifiability - Are measures expressed numerically where appropriate, and supplemented by qualitative information when necessary?
- Controllability - Are we measuring variables, factors and relationships over which we have control or influence?
- Cost-effectiveness - Is the potential payback from performance measurement commensurate with the effort and resources expended?

Literature sources and professional opinion (Hotto, 1994) suggest that approaches to evaluating water quality
information system performance should be:

- convenient and easy to use, with minimal consultant assistance required,
- easy to understand and fitted to the manager's analytical process,
- inexpensive to develop and operate,
- easily modified and sufficiently flexible to apply to many information system formats and situations,
- useful in real time with fast turnaround,
- well documented and easily audited,
- able to predict the impact of information system modifications (i.e., perform sensitivity analyses), and
- able to clearly indicate management system and information system improvement opportunities.

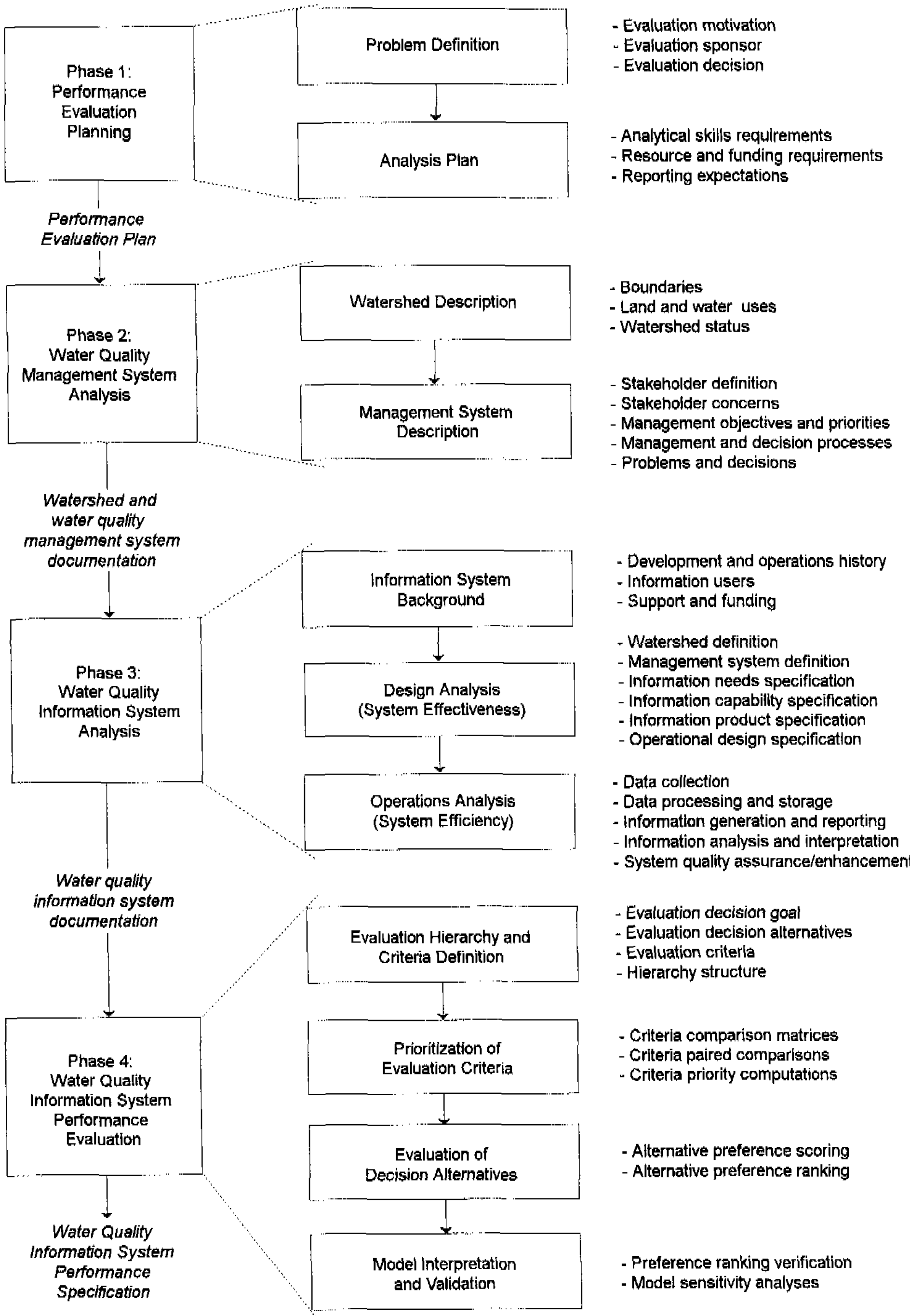

Figure 1: Framework for Evaluating Water Quality Information System Performance

A total system perspective implies that water quality information system performance must be judged with respect to the watershed it purports to describe and the management system it supports. This Framework assumes that perspective, satisfies the listed measurement process criteria, and guides the water quality manager through designated system analysis phases. The sequence of the Framework's activities and specific evaluation considerations in each phase are shown in Figure 1.

PHASE (1): PERFORMANCE EVALUATION PLANNING

The rationale for the water quality information system evaluation (or design) is defined and the feasibility of a successful effort is established. A comprehensive performance evaluation approach is documented, which includes:

- Problem Definition - Motivation for the evaluation is clearly defined and the sponsor or champion of the effort is identified.

- Analysis Plan - Skill requirements are identified, funding requirements are established, reporting expectations are identified, and activity schedules are determined.

PHASE (2): WATER QUALITY MANAGEMENT SYSTEM ANALYSIS

The watershed and the management system(s) which control or influence its behavior are characterized and documented. This analysis provides:

- Watershed Description - This document details watershed boundaries (spatial, temporal, and jurisdictional), land and water uses (historic, current and projected), and any relevant economic, social, and regulatory factors.

- Management System Description - An explicit description of management programs in the watershed that the information system serves or may impact is critical. The description must identify watershed stakeholders and their standing, list stakeholder concerns, describe management objectives and priorities, describe the management and decision-making processes served, and specify the problems and decisions to be addressed.

PHASE (3): WATER QUALITY INFORMATION SYSTEM ANALYSIS

The design and operation of the water quality information system(s) are analyzed and documented. The steps of that analysis include:

- Information System Background - An understanding of the original system's design (i.e., designer, design methodology, development and operations documentation, clients and users, and sources of support and funding) provides a foundation upon which to evaluate performance and redesign the system.

- Design Analysis - A description of the system's effectiveness (i.e., how well it addresses management objectives and information needs) includes statements as to the specification of watershed, management system, information needs, information capability, information product, database requirements, and operating procedures.

- Operations Analysis - A description of the system's efficiency (i.e., its ability to satisfy information needs with minimal resources and effort), focuses upon data collection (field, lab, or other), data processing and storage, information generation and reporting, information analysis and interpretation, and system quality assurance and enhancement.

PHASE (4): WATER QUALITY INFORMATION SYSTEM PERFORMANCE EVALUATION

The findings from the watershed, management system, and information system analyses are translated into specific information system evaluation criteria which are rated as to their relative importance in the evaluation at hand. These criteria may simply represent the recognized steps of effective information system design practice, or may be more explicitly defined with respect to the systems in question. Information systems to be evaluated are scored with respect to the criteria and system grades computed in order to quantify the evaluation.

Information system performance is defined as a relative preference score computed for each alternative system being compared. A system alternative may be another operating information program, a proposed or redesigned system, or some hypothetical "ideal" system postulated by the evaluator. Preference scores are derived using the Analytic Hierarchy Process (AHP), a multi-attribute, hierarchical decision analysis model. The steps of the Analytic Hierarchy Process include: (1) constructing a decision (evaluation) hierarchy which defines decision objectives, evaluation criteria, and decision alternatives, (2) relative weighting of the system performance evaluation criteria, (3) evaluating the decision alternatives against the criteria, and (4) interpreting and validating the model's results.

In fact, a number of multiple-attribute decision models could be used in this evaluation/quantification phase of the Framework. For example, models employing utility functions and/or defining performance indices can be used to accomplish this same purpose . The AHP's virtues include its ability to accommodate subjective and objective criteria in one decision process and to conveniently produce a composite numerical measure of an alternative's value (the preference score). Figure 2 illustrates a generic AHP hierarchy. The reader is referred to Saaty (1990), the originator of the AHP technique, for a complete description of the method.

To summarize, the Framework for Evaluating Water Quality Information System Performance compels the water quality manager to define and document his or her information system performance criteria, and offers a convenient model with which the system's design and operation can be scored against those criteria. The systematic examination of the watershed and its associated management systems helps to assure the manager that "all bases have been covered" when assessing the utility of any supporting information system. The performance evaluation model allows the evaluator to explicitly rank the criteria and score the system's performance with respect to his or her unique measures of utility.

APPLICATION OF THE FRAMEWORK TO A MUNICIPAL MONITORING PROGRAM

The Framework for Evaluating Water Quality Information System Performance has been used to assess two water quality monitoring and information programs related to a unique municipal water transfer project. The comparison is retrospective in nature, intended to demonstrate the use of the Framework. The Framework was not employed as a decision aid at the time of the investigation described.

The City of Thornton, Colorado (Thornton) elected to augment its future water supplies through the purchase of existing remote basin water rights rather than rely solely upon traditional development procedures. As a condition of the purchase contract, Thornton was required to establish a water quality monitoring program. The program's purposes were to assist in: (1) determining the water's suitability for municipal use, (2) specifying design requirements for future water delivery and treatment systems, and (3) certifying that other basin users could continue to use the water for traditional purposes, as dictated by Colorado water law.

The monitoring program operated for three years (1986-1989) without formal review or analysis. During that period: (1) new water quality information goals and information needs appeared, (2)

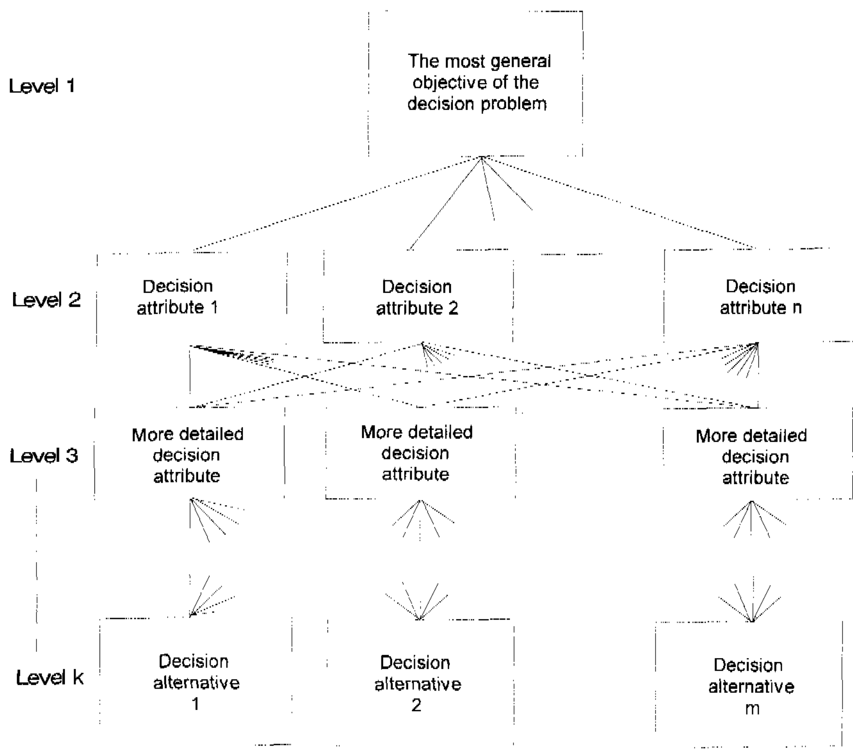

Figure 2: Analytic Hierarchy Process Decision Schema-A Hierarchy with K Levels. Source after Zahede, 1986

key program personnel resigned or were reassigned, (3) program scope was enlarged to include groundwater monitoring, (4) program procedures were modified, and (5) program costs escalated. Reacting to these concerns, Thornton's water quality managers authorized an investigation to enhance the water quality monitoring program. The investigation and redesign efforts were conducted from 1989 through 1992, following systematic design concepts promoted by Sanders, Ward and other researchers. Details of an enhanced monitoring program were recommended to Thornton's water quality managers early in 1992 (Hotto and Sanders, 1991; Hotto, 1992).

EVALUATE PERFORMANCE OF ALTERNATIVE SYSTEMS

The water quality information systems being evaluated using the Framework are Alternative 1 - Thornton's original water quality monitoring program, and Alternative 2 - the enhanced water quality monitoring program. To accomplish the decision goal of identifying the preferable alternative, criteria were defined and linked in a decision hierarchy. Composite preference scores describing the priorities of the alternatives were computed from criterion priority values derived at each level of the hierarchy (see Saaty, 1990). For the decision hierarchy and comparisons outlined in this demonstration, the composite preference scores for the water quality information system alternatives were 0.25 for the original monitoring program and 0.75 for the enhanced monitoring program. Figure 3 illustrates the decision hierarchy constructed to evaluate these water qualit information systems alternatives.

INTERPRET AND VALIDATE EVALUATION MODEL RESULTS

These preference scores suggested that the enhanced water quality monitoring program offered a significant improvement in performance over the original program. However, the ultimate inter

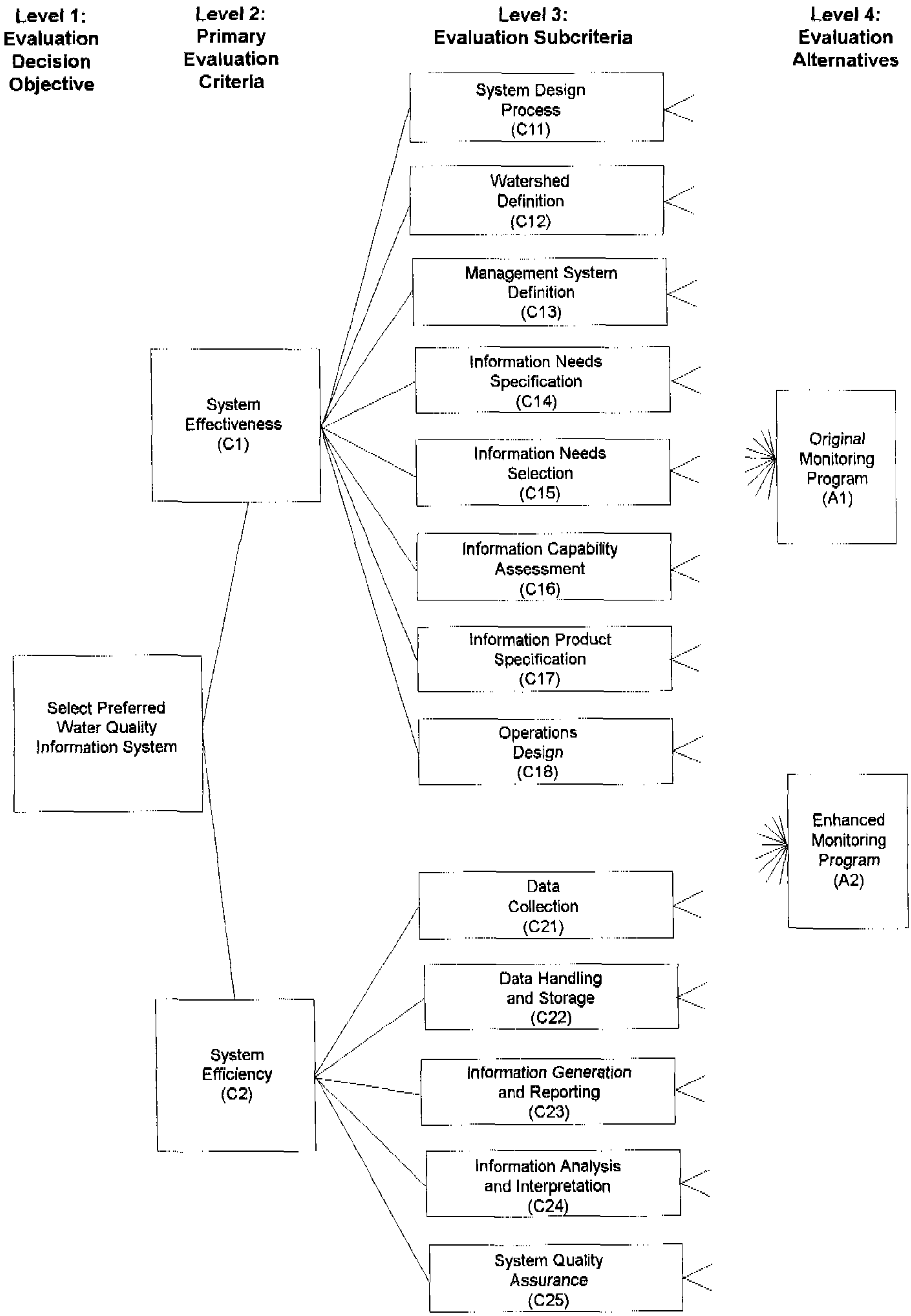

Figure 3: Decision Hierarchy for Comparison of Water Quality Information System Alternatives

pretation of the alternative preference scores is a subjective exercise on the part of the evaluator. Preference scores may suggest strong distinctions among the decision alternatives, but the evaluator must keep in mind that the significance imputed to the scores is based upon confidence in all of the assumptions that he or she has built into the model. The evaluator's trust in the validity of the alternative scores (and in the decision model itself) can be raised through sensitivity analyses, which test the constancy of conclusions drawn with respect to variations in the structure, assumptions, and inputs of the model.

In this comparison the preference order of the alternatives was never in question, but the author was surprised at the magnitude of the preference associated with the enhanced program. To gauge the stability of the model's output and to resolve that uncertainty, sensitivity analyses were undertaken to determine the effect on alternative preferences of varying assumptions regarding primary criteria, effectiveness and efficiency subcriteria, and alternatives scores on subcriteria. These analyses indicated the ranking and preference scores of the information system alternatives of this model were consistent over the range of reasonable attribute assumptions. The model appeared quite stable, with no single criterion, assumption or estimate of an alternative's performance radically altering the behavior of the model or causing unexplainable results.

SUMMARY AND CONCLUSIONS:

Water quality managers, regulators and consultants must adopt a more "bottom-line" viewpoint in executing their duties. They must be more accountable to their public, and will be required to demonstrate the results of past water quality management expenditures, as well as to justify future programs. To meet those demands, water quality managers must have information which can be directly related to regulatory or managerial objectives, and can support the corresponding management decision processes.

To help managers obtain decision-relevant water quality information, a comprehensive and consistent approach to water quality information system analysis has been presented. The Framework for Evaluating Water Quality Information System Performance is an auditing, documentation and evaluation approach which embodies four major activities: (1) preplanning of the evaluation effort, (2) characterization and documentation of the watershed and all relevant water quality management systems, (3) analyses of the effectiveness and efficiency of the water quality information system(s) in supporting water quality management objectives, and (4) evaluation (quantification) of water quality information system performance with respect to effectiveness and efficiency criteria.

The Framework is intended to help a water quality professional decide how well a water quality information system serves management's purposes. It is a management and engineering instrument, not a tool to perform specific technical or statistical analyses. The Framework helps the manager or analyst ask relevant and incisive questions about what information the water quality information system should provide, and how well the information is provided.

The Framework's advantages in supporting water quality management decision-making include:

- *The Framework is comprehensive.* Its process promotes fundamental system understanding and demands explicit identification of the water quality information needs and management decision objectives to be served. The performance of information system design and operations is evaluated directly with respect to how those objectives and needs are (or should be) satisfied.

- *The Framework is user-friendly.* Assessment and evaluation tasks are straightforward and can be accomplished by the water quality manager with a minimum of assistance or consultation. The Analytic Hierarchy Process and available affordable software allow the manager to prioritize evaluation criteria, and to create a personalized performance evaluation process.

- *The Framework is flexible and widely applicable.* Water quality managers can apply the proposed assessment questions to watersheds, management systems and information systems of any size or complexity. The Framework produces a customized analysis of the information system, enhancing the manager's understanding of, confidence in, and commitment to any resulting decisions or actions.

- *The Framework is economical.* Management system and information system assessments can be carried out to whatever level of detail the manager deems appropriate or affordable. The potential financial investment is low and easily controlled. The risks and consequences of process or model obsolescence are minimal.

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AN ASSESSMENT OF WATER QUALITY MONITORING IN THE DUTCH COASTAL ZONE: NEEDS, AIMS AND OPTIMISATION

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ABSTRACT

January 1996 saw the launch of a new programme for the monitoring of pollutants in marine waters around the Netherlands. The first step was making an inventory of information needs. The aims of the programme are to assess whether water quality meets targets and to detect trends over time for substances with high concentrations in the eleven marine waters which are distinguished. An extensive statistical study was conducted, resulting in recommendations for more efficient monitoring. The number of sites in water was reduced from 75 to 32. The new programme includes fewer heavy metal analyses since concentrations of these substances frequently meet the targets. Instead, there is an increase in pesticide analyses. There is also a shift in emphasis between compartments: less on the water phase and more on sediment. In addition, there are now clear criteria for starting and stopping monitoring activities.

INTRODUCTION

In the early seventies, a number of programmes were set up to monitor trends in the water quality of marine waters. These programmes focused primarily on the measurement of a number of traditional variables such as salinity, suspended matter, oxygen and nutrients (Beukema et al., 1986). The North Sea monitoring network was distinguished by its strictly section-based approach: there were over 70 monitoring sites located along 10 transects perpendicular to the coast (Figure 1). A similarly large number of sites were selected in the other marine water areas, e.g. the Wadden Sea. Samples were collected every two weeks.
As time went on, the focus within water policy shifted more towards organic micropollutants (e.g. PCBs and PAHs), and uniform quality objectives were introduced (V&W, 1989). The techniques

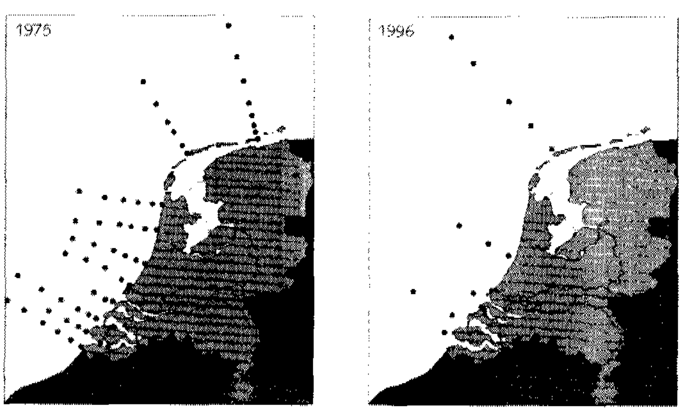

Figure 1: Overview of monitoring sites for the water compartment in the Dutch coastal zone in the early 1970s and in 1996.

of analysis used in the monitoring improved, but the analyses were still expensive. In addition, there was an increasing awareness that many (organic) micropollutants were accumulating in sediment and in organisms, and monitoring was therefore extended to include these compartments too.

As a result, by the early nineties a situation had been reached in which numerous substances were being measured in numerous compartments, but there was, nevertheless, a general feeling that not enough information could be derived from the data acquired at such expense. The familiar 'data-rich but information-poor syndrome' was making itself felt (Ward et al., 1986; 1990).

At this point, an evaluation of the chemical monitoring network in the marine water areas was initiated. The aim was to produce an efficient monitoring network able to satisfy the need for information. The project consisted of three distinct phases: 1. identification of information needs, 2. a statistical evaluation of the data assembled during the 1988-1994 period, and 3. production of a monitoring strategy for 1996-2000.

INFORMATION NEEDS

As a first step, an inventory of information needs was undertaken with regard to the chemical monitoring of national waters (Swertz & Akkerman, 1994; Hofstra 1994). Interviews were conducted with interested parties from the water policy and water management fields and with relevant experts (including the client, the Directorate of Public Works and Water Management).

The questions related to the presence in the water systems of chemical compounds, for convenience termed pollutants. This may seem a trivial point, but it does mean that a number of traditional water quality variables, such as suspended matter, transparency and chlorophyll- , were excluded from the evaluation. These come under the heading of physical or biological monitoring.

The interviews led to the conclusion that the purpose of such monitoring is to evaluate national water policy. Seen from that point of view, there were two questions: Is the water polluted? And if so, are the concentrations decreasing? The criterion for assessing pollution is whether the water meets water quality objectives. In the case of the marine waters, these are the target figures. The aim of monitoring is to see whether the water meets the target. If it does not, or if there is no target for a particular substance, the next question is whether the relevant concentrations are changing over time. The aim of monitoring is then to detect trends. These two monitoring aims were then quantified.

The exclusive concentration on these two aims means that the chemical monitoring network is not designed to meet other information needs, such as the need to quantify fluxes or to verify numerical models. The monitoring programme will, however, take account of all international commitments made by the Netherlands. In other words, the main goal of the programme is to evaluate the undertaken water quality policy in the Nethertlands and there is no special scientific purpose.

The next step was to define the spatial level at which information was required. Here there was coordination with the Aquatic Outlook project (Luiten & Van Buuren, 1994). The marine waters were divided on the basis of chemical and hydrological characteristics into eleven water systems (Figure 2).

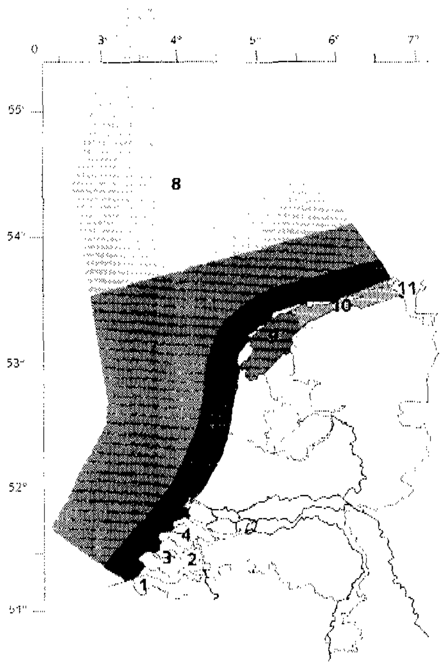

Figure 2: Division of the Dutch marine water areas into eleven water systems. 1: Western Scheldt, 2: Eastern Scheldt, 3: Veerse meer, 4: Grevelingenmeer, 5: Voordelta, 6: Coastal Zone, 7: Southern Part of North Sea, 8: Central Part of North Sea, 9: Western Wadden Sea, 10: Eastern Wadden Sea, 22: Ems- Dollart estuary.

STATISTICAL STUDY

By way of evaluation, but first and foremost in order ultimately to be able to design an effective monitoring network, an extensive statistical study was conducted. This had five main components:

1. Characterizing the datasets by means of tables and graphs;
2. Estimating the components of variation using analysis of variance;
3. Investigating whether monitoring sites were superfluous by way of a correlation study;
4. Comparing the different compartments in which monitoring took place;
5. Optimizing the number of observations.

Some results relating to these five components of the study are summarized below (Swertz et al., 1996).

CHARACTERIZATION OF DATASETS

The data available from each water system were set out in box-and-whisker plots (Figure 3). The probability distribution was examined (including a test for normality) and the data were plotted with the sites, the months and the years on the X-axis. This gave an impression both of the quality of the data and of the influence of the variation components.

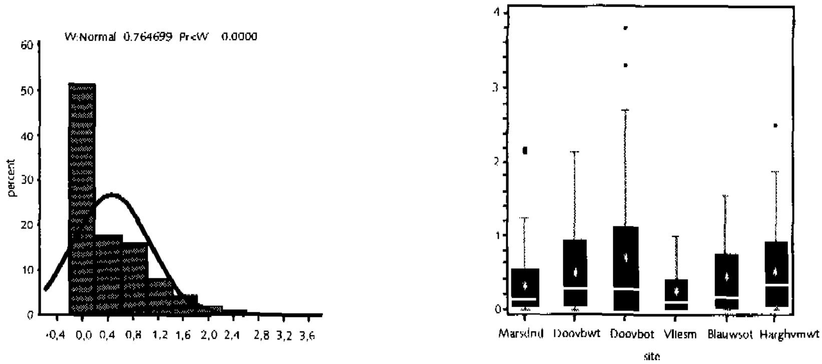

Figure 3: An example of the probability distribution and a box-and-whisker plot of dissolved nitrate and nitrite in mg N/l at six sites in the Western Wadden Sea.

ESTIMATING THE VARIATION COMPONENTS

An analysis of variance was conducted both for the variables and for the water systems using year, month and site as factors. Since these are the same factors as are used in the box plots, it was then possible to quantify the variation components. Table 1 gives an example of the results. The table shows that spatial variation is generally pronounced, and particularly so in the estuaries (Western Scheldt, Ems-Dollard, Western Wadden Sea) and the coastal zone. In the salt-water lakes, the seasonal component predominates. The contribution of sampling and analytical variation to total variation is small.

Water	CV	Te	Sp	Sa	An	Re
Western Scheldt	9	11	75	3	<1/2	10
Eastern Scheldt	9	70	2	6	<1/2	22
Lake Grevelingen	11	87	<1/2	4	<1/2	9
Voordelta	12	18	50	4	<1/2	29
Coastal Zone	11	16	47	4	<1/2	33
Southern North Sea	14	21	26	5	<1/2	48
Central North Sea	15	24	9	5	<1/2	61
Wadden Sea West	9	10	41	5	<1/2	45
Wadden Sea East	7	23	5	6	<1/2	66
Ems-Dollard	11	7	74	2	<1/2	16
median in water systems	9	20	32	5	<1/2	43

Table 1: The variation components in the total concentration of phosphorous in the marine water systems, first the coefficient of variation (CV) as percentage of the mean concentration, then the temporal (Te), spatial (Sp), sampling (Sa), analytical (An) and remaining variation (Re) as percentages of the total variation.

INVESTIGATION OF POSSIBLE SUPERFLUITY

A correlation study was conducted in order to examine the extent to which observations at the monitoring sites can be predicted on the basis of each other (Van der Meulen, 1995). It emerged that observations of dissolved substances at sites closely adjacent to each other (several kilometres apart) could be used to predict each other with an accuracy of over 90 per cent (for example Figure 4, which considers the dissolved orthophosphate concentration). This is not, however, the case with variables relating to suspended matter. In terms of the correlation over time (the autocorrelation), it emerged that monitoring at intervals of less than a month provided superfluous data in the marine water areas.

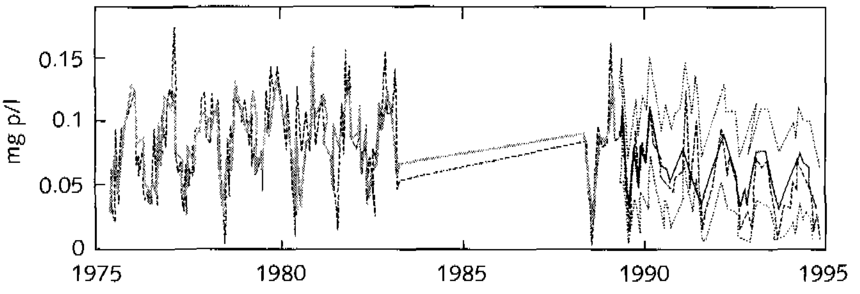

Figure 4: The correlation for dissolved orthophosphate between the sites at Noordwijk 4 and 10 km off the coast (plotted as solid and dotted line respectively); the times series until 1990 is used for model identification, from 1990 until 1995 for model verification and for this period the interval in which the prediction falls with in 95 per cent confidence is plotted.

COMPARISON OF COMPARTMENTS.

Many substances are monitored in several different compartments: water, suspended matter, sediment and organisms. In the early nineties, metals were being monitored in eight different compartments. The results of monitoring over the 1988-1994 period were analyzed in order to establish the minimum detectable trend. At the same time, a rough estimate was made of the costs. Table 2 shows the detectable trend of a persistent substance, PCB 153. This is the minimum detectable trend, given this dataset, which means that only larger trends can be detected with this power and confidence level (95 and 90 %). The lower the detectable trend the better. The conclusion is that the compartments in which monitoring can most efficiently take place in order to detect trends are suspended matter and sediment.

compartment	detectabletrend (% over five years)	costs (NLG)
suspended matter	40-80	1300
sediment	50-80	1400
mussel (active sampling)	40-130	2000
mussel (passive sampling)	80-160	1200
flatfish	90-200	1000

Table 2: The detectable trend and the costs for analysis of PCB 153 in five compartments with one observation a year, using a power of 90 % and a confidence of 95 %.

FREQUENCY OPTIMIZATION

Once a compartment has been selected, the next question is how many observations need to be made in order to obtain sufficient accurate information in an efficient way. To find the answer, the

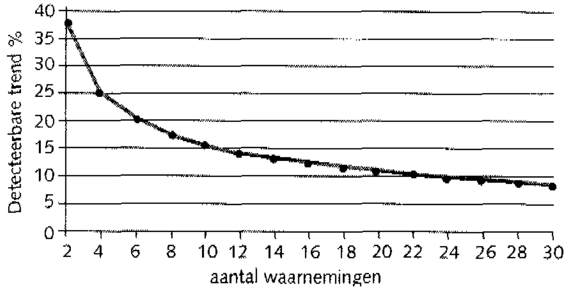

Figure 5: The detectable trend over ten years versus the number of observations per year for dissolved orthophosphate in the Ems-Dollart estuary in winter using a power of 90 % and a confidence of 95 %.

The final recommendations relate to the completion of the monitoring cycle, since data alone are insufficient to meet the entire demand for information (e.g. UN/ECE, 1996). There is still no accepted normalization method for assessing concentrations of metals in sediment, since the present standard method (CUWVO, 1990) is unsuited to marine conditions (use of organic carbon and grain size fraction). Likewise, a method of analysis has still to be perfected for the butyltin compounds, one of the problem substances in Dutch marine waters (Evers et al., 1995). Lastly, work is being undertaken on a quality control system for the determination of chemical indicators, including protocols. Then, the information can be reported.

CONCLUSION

Overall, the situation can be summed up as follows. Following the evaluation there has been no change in the overall amount of money expended on chemical monitoring. However, the programme has been rendered far more effective by coordinating it more closely with actual information needs and by creating a more intelligent spread of monitoring over the various substances, compartments and water systems. In addition, there are now clear criteria for starting and stopping monitoring.

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10.

ON THE PROBLEMS ASSOCIATED WITH VARIOUS SAMPLE MEDIA USED TO MONITOR TRACE ELEMENT FLUXES

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ABSTRACT

Trace element flux monitoring requires the collection of calendar- and event-based dissolved and suspended sediment samples for subsequent chemical analyses. There are unique problems associated with the accurate collection and analyses of both types of sample media.
Over the past 10 years, reported ambient dissolved trace element concentrations have declined. These decreases may not reflect better water quality, but rather improvements in the methodologies used to collect, process, preserve, and analyze water samples without contaminating them during these steps. Contamination reduction substantially increases sampling and analytical costs for all types of water quality studies. Further, recent studies have shown that the currently accepted operational definition of dissolved constituents (material passing a 0.45-µm membrane filter) is inadequate due to sampling and processing artifacts.
The existence of these artifacts raises serious questions about the the generation of accurate dissolved trace element data. Suspended sediment and associated trace elements can display marked short-term spatial and temporal variability.
This implies that spatially representative samples only can be obtained using depth- and width-integrated sampling techniques. Additionally, temporal variations have led to the view that the determination of annual fluxes requires nearly constant, high frequency sampling, and subsequent chemical analyses.
However, when time-scales shift from periods of a few hours or days, to an entire year, quantitating short-term variability becomes markedly less important. As such, it may be possible to use mean/median chemical data, combined with measurements of discharge and suspended sediment concentration, to estimate solid-phase contributions to annual fluxes, within acceptable error limits.

INTRODUCTION

The USGS (U.S.Geological Survey) NASQAN (National Stream Quality Accounting Network) was initiated in 1973 and had four major objectives: 1) to account for the quantity and quality of water moving within and from the U.S.; 2) to depict the areal variability of water quantity and quality; 3) to detect changes in water quantity and quality; and 4) to set the stage for future assessments of water quality (Ficke and Hawkinson, 1975). At its peak, over 500 stations were sampled, up to twelve times per year, for a variety of chemical constituents (OWQ, 1996).

In 1994, in response to diminishing resources, changes in data requirements, and to better integrate it with other USGS ambient water-quality monitoring programs, NASQAN was redesigned. The major objective of the new program, which began in late 1995, is to characterize large, selected U.S. rivers by measuring the mass flux of selected constituents at critical nodes in various river basins; selected river basins include the Mississippi, Columbia, Colorado, and Rio Grande.

SAMPLING MEDIA

Concentrations of dissolved and suspended sediment-associated trace elements, during baseflow and major events (e.g., spring runoff, floods) are necessary for the estimation of annual chemical fluxes. However, there are unique problems associated with the collection and subsequent chemical analyses of both types of sample media (dissolved and suspended). The success of any flux-based monitoring program, regardless of scale, (local, regional, national, or global) requires that these various problems be addressed.

WHOLE WATER SAMPLES

Historically, most water-quality investigations have tried to assess trace elements in aquatic systems by analyzing water samples. This has entailed determining the concentrations of total recoverable (whole water) and dissolved trace elements through the collection and analysis, respectively, of unfiltered and filtered water. However, at typical suspended sediment (≤70 mg l^{-1}) and associated trace element concentrations, whole water total recoverable analyses generally do *not* provide an accurate measure of sediment-associated trace element concentrations due to dilution effects and limitations in analytical techniques. Thus, the use of whole water total recoverable trace element data in flux-based monitoring programs should be discouraged.
As an example, examine the data for a sample from the Susquehanna River in which the sediment concentration was 4 mg l^{-1} (Table 1). The chemical concentrations for the suspended sediment were determined after it had been physically separated from a whole-water sample by flow-through centrifugation.
The chemical concentrations for several trace elements are elevated (Ag, Zn, Ni, Co, Cd, Cr, and As; Concentration, Table 1). On the other hand, when the chemical data for the suspended sediment are converted back to whole-water sample values, the concentrations appear quite low (Recalculated Whole-Water Concentration, Table 1). Comparison of the recalculated whole-water concentrations with those for currently accepted dissolved concentrations from unimpacted areas indicates that the suspended sediment accounts for a significant proportion of the total concentration of many of the trace elements (Calculated % of Solid Phase Contrib., Table 1). Despite this, if the whole-water concentrations are compared to typical reporting limits for many water quality laboratories, not one sediment-associated trace element concentration would have been detected because the whole-water values were less than their respective reporting limits (NWQL Reporting Limit, Table 1). These data show a major problem with determining suspended sediment-associated trace element concentrations using whole water samples and the 'method of difference' [subtracting the concentrations from a filtered ('dissolved') sample from the concentrations from a whole-water (suspended sediment plus water) sample].
To place this problem in an appropriate context, it helps to see what concentrations would occur if a whole water sample were 'created' using a sediment containing average trace element concentrations (Average Sediment-Associated Trace Element Conc., Table 1). The values represent typical chemical levels associated with fine-grained sediment samples collected in unimpacted areas (Horowitz et al., 1989). Using currently available reporting limits, it is possible to calculate the minimum suspended sediment concentration required before each trace element could be detected in a whole-water sample (Mass Required to Reach NWQL Reporting Limits., Table 1). Considering the typical NASQAN median suspended sediment concentration (e.g., ~70 mg l^{-1}), relative to the requisite masses listed, many of the trace elements would be at or below current reporting limits. The problem is actually far worse because the calculated mass requirements assume that all the trace elements are quantified (a total analysis). Typically, this is not the case for whole-water analyses because the presence of water in the samples, as well as the digestion procedures used, preclude complete solubilization/quantification of all the entrained trace elements.

Table 1: Sediment-Associated Trace Element Data for a Suspended Sediment Sample From the Susquehanna River containing 4 mg l^{-1} suspended sediment.

	Ag	Cu	Pb	Zn	Ni	Co	Cd	Cr	As	Sb	Se	Fe	Mn	Al	Ti
Concentration ($\mu g\ g^{-1}$) [1]	1.8	50	58	450	120	77	1.2	123	17.2	2.0	1.3	57,000	6400	93,000	4800
Recalculated Whole Water Concentration ($\mu g\ l^{-1}$) [2]	0.01	0.2	0.2	1.8	0.5	0.3	0.01	0.5	0.07	0.01	0.01	228	25.6	372	19.2
Average Dissolved Trace Element Concentrations ($\mu g\ l^{-1}$) [3]		0.2	0.05	0.2	0.3	0.05	0.01	0.1	0.5	0.05	0.08				
Calculated % Solid Phase Contrib. to Whole Water Conc. [4]		50	82	90	62	51	50	83	12	17	11				
NWQL Reporting. Limit ($\mu g\ l^{-1}$) [5]	1	1	1	3	1	1	1	1	1	1	1	3	1	3	
Average Sediment-Associated Trace Element Conc.($\mu g\ g^{-1}$) [6]	0.5	25	50	100	25	18	0.6	20	7.0	0.6	0.4				
Mass Required to Reach NWQL Reporting Limits (mg) [7]	2000	40	20	33	40	55	1650	250	140	1650	2500				

[1] - trace element concentrations in a separated (centrifuged), freeze-dried, and totally digested suspended sediment sample from the Susquehanna River.
[2] - calculated whole-water concentrations of the suspended sediment sample (from [1] above) based on a suspended sediment concentration of 4 mg l^{-1}.
[3] - average dissolved concentrations from unimpacted areas.
[4] - calculated percent contributions of the suspended sediment-asociated trace element concentrations using dissolved concentrations reported for relatively clean areas.
[5] - current reporting limits for the NWQL for dissolved and/or digested whole-water samples.
[6] - average total trace element concentrations for fine-grained, unimpacted bed sediments.
[7] - calculated suspended sediment concentration required to produce a whole-water concentration equal to current NWQL reporting limits using the average total trace element concentrations in [5], in conjunction with the reporting limits given in [4].

Table 1: Sediment-Associated Trace Element Data for a Suspended Sediment Sample From the Susquehanna River containing 4 mg 1-1 suspended sediment

DISSOLVED WATER SAMPLES

Contamination: Over the past 10 years, reported ambient dissolved trace elements levels have declined from tens of parts-per-billion (µg l^{-1}), to single digit parts-per-billion, into the parts-per-trillion (ng l^{-1}) range (e.g., Shiller and Boyle, 1987; Windom, et al., 1991; Benoit, 1994; Nriagu, et al., 1996). These decreases generally do not reflect improved water-quality, but rather, improvements in sampling, processing and analytical methodologies which limit contamination The procedures required to reduce contamination have increased sampling and analytical costs for all types of water-quality studies, and have made it difficult for many first world countries, let alone second and third world countries, to generate accurate dissolved trace element data. However, a flux-based monitoring program requires data on several dissolved constituents; thus, a relatively simple, user-friendly protocol for the collection and processing of uncontaminated samples must be available to permit trace element quantitation at ambient levels (Horowitz, et al., 1994).

Processing Artifacts: Reductions in contamination, and concomitant decreases in reported ambient dissolved trace element levels have been accompanied by the view that the currently accepted operational definition of dissolved constituents (filtration of unspecified volumes of natural water through unspecified 0.45-µm membrane filters) is inadequate due to sampling and processing (filtration) artifacts (Horowitz, et al., 1992; 1996). In fact, once significant sample contamination has been eliminated, 'dissolved' trace element concentrations appear to result predominantly from colloidal material passing through 0.45-µm membrane filters, rather than to truly dissolved constituents. As such, the inclusion/exclusion of colloids has a major impact on reported 'dissolved' trace element concentrations. Studies have indicated that the inclusion/exclusion of colloidal material can be affected by such diverse factors as: 1) filter type; 2) filter diameter; 3) filtration method; 4) suspended sediment concentration; 5) suspended sediment grain-size distribution; 6) concentration of colloids; 7) concentration of organic matter; 8) volume of sample processed because of 4, 5, and 6; and 9) method of sample collection, again because of 4, 5, and 6. The issue of filtration artifacts raises questions about the utility and useability of dissolved trace element data for trend identification, for the generation of accurate chemical averages for comparison/regulatory purposes, and for the determination of other than local annual fluxes, since these all tend to rely on data from multiple sources. Thus, any successful flux-based monitoring program that calls for data on 'dissolved' concentrations requires sampling and processing protocols that address the problems of filtration (processing) artifacts.

As an example, examine the data from a field study designed to evaluate the effect(s) of using different (e.g., manufacturer, diameter) 0.45/0.40-µm membrane filters to field-process whole-water samples for subsequent 'dissolved' trace element quantitation (Table 2). Three filters were used: 1) a 47mm, 0.40-µm polycarbonate Nuclepore plate filter with a surface area of 17.3 cm^2, 2) a 142 mm, 0.45-µm cellulose nitrate MicroFiltration Systems (MFS) plate filter with a surface area of 158 cm2, and 3) a 47 mm, 0.45-µm polyethersulfone Gelman capsule filter with a surface area of 600 cm2. The Nuclepore was selected because it is the filter of choice for numerous aquatic chemists (e.g., Shiller and Boyle, 1987; Benoit, 1994); the MFS was selected because it was used extensively by the USGS and other U.S. monitoring agencies until the development of a new sampling and processing protocol (Horowitz, et al., 1994); and the Gelman capsule was selected because it was used by Windom, et al. (1991), and is currently used by the USGS (Horowitz, et al., 1994) and recommended by the U.S. EPA.

Whole-water samples were processed through each filter type, and sequential aliquots (250 mL if possible) collected. The Nuclepores were the slowest, and processed the smallest whole-water volume (~100ml). MFS filters were faster, but filtration rates declined after 750 to 800ml. Gelman capsules did not appear to slow even after 3500ml were processed. Although the Tangipahoa sample did not have as much suspended sediment as the Mississippi sample, it contained more organic matter. In fact, the processing rates for all three filters were slower for the Tangipahoa than for the Mississippi sample. Field blank concentrations were either below detection limits, or were sufficiently low relative to the measured concentrations, as to be insignificant. The detectable Mississippi dissolved trace element data fell into one of three categories: 1) affected by filtration artifacts, 2) possibly affected by filtration artifacts and/or dilution, and 3) not

Mississippi River[2,3]	Analytical Method[1] and Constituent																	
Filter[4] and	MS	OES	MS	MS	MS	MS	MS	MS	MS	MS	MS	MS	OES	OES	OES	OES	OES	OES
Aliquot Volume	Al	Fe	Cr	Mn	Co	Ni	Cu	Zn	Mo	Ba	Pb	U	B	Sr	Ca	Mg	Na	Si
Nuclepore																		
0-100mL	8.2	7	0.5	14	0.4	<0.8	1.8	2.1	1.3	57	<0.2	0.9	33	160	39	11.1	19.5	7.0
MicroFiltration Systems																		
0-250mL	26	27	1.3	14	0.4	<0.8	2.2	4.6	1.3	54	<0.2	0.9	31	153	37	10.7	18.2	6.7
250-500mL	8.6	14	0.8	14	0.5	<0.8	2.1	3.8	1.5	58	<0.2	1.0	34	164	39	11.3	19.8	7.1
500-750mL	4.9	4	0.9	14	0.6	<0.8	2.4	4.1	1.5	58	<0.2	0.9	31	161	39	11.3	19.4	7.1
Gelman Capsule																		
0-250mL	43	58	<0.3	13	0.2	2.2	50	6.1	1.2	52	<0.2	0.9	31	152	37	10.6	18.6	6.9
250-500mL	41	56	0.3	14	0.4	2.0	3.9	2.7	1.2	57	<0.2	1.0	32	160	39	11.2	19.6	7.2
500-750mL	32	50	<0.3	14	0.3	1.7	2.2	2.5	1.1	59	<0.2	1.1	34	163	39	11.3	19.7	7.2
750-1000mL	27	43	0.3	14	0.3	2.0	1.9	1.5	1.3	55	0.6	1.1	37	155	39	11.3	19.7	7.2
1125-1375mL	22	34	0.4	14	0.4	2.1	2.2	2.4	1.5	56	0.6	1.1	31	161	39	11.2	19.3	7.1
1375-1625mL	18	36	0.4	13	0.4	2.5	2.2	1.9	1.7	56	0.8	1.1	34	163	40	11.3	19.2	7.2
1625-1875mL	15	19	0.5	14	0.4	3.0	2.5	2.2	1.8	57	0.7	1.1	32	161	39	11.2	19.5	7.0
1875-2125mL	11	24	0.6	13	0.5	2.3	2.7	2.8	2.0	58	0.7	1.1	32	161	39	11.2	19.5	7.1
2125-2375mL	8.6	20	0.6	14	0.5	2.6	2.5	2.1	1.8	57	0.9	1.0	32	162	39	11.2	19.4	7.1
2375-2625mL	8.2	20	0.7	13	0.5	2.8	3.0	2.7	1.9	58	1.0	1.0	34	164	40	11.4	19.6	7.1
2625-2875mL	7.4	17	0.6	14	0.5	2.6	2.7	2.5	1.9	59	0.9	1.1	32	161	39	11.2	19.9	7.0
2875-3125mL	6.8	15	0.6	14	0.5	2.9	2.7	2.6	2.0	59	1.0	1.1	31	161	39	11.1	19	7.0
3125-3375mL	3.4	13	<0.3	14	0.3	1.9	2.1	2.4	1.2	58	<0.2	1.1	33	162	39	11.2	19.6	7.0

[1]Analytical methods included inductively coupled plasma-mass spectrometry (MS) and iductively coupled plasma optical emission spectrometry (OES).

[2]The concentrations for Be (0.6μg/L), Ag (0.2μg/L), Cd (0.2μg/L), Sb (0.2μg/L), Li (1μg/L) and V (3μg/L) were excluded because they were all below the method detection limit (given in parentheses).

[3]The concentration of suspended sediment was 157 mg/L.

[4]Filters were a 47mm, 0.40-μm Nuclepore, a 142mm, 0.45-μm MicroFiltration Systems, and a 47mm, 0.45-μm Gelman capsule.

Tangipahoa River[6,7]	Analytical Method[5] and Constituent														
Filter[*] and	MS	OES	MS	MS	MS	MS	MS	MS	MS	OES	OES	OES	OES	OES	OES
Aliquot Volume	Al	Fe	Mn	Co	Ni	Cu	Zn	Ba	Pb	B	Sr	Ca	Mg	Na	Si
Nuclepore															
0-100mL	36	55	75	<0.2	<0.8	1.0	2.2	24	<0.2	10	13	1.3	0.7	2.3	4.2
MicroFiltration Systems															
0-250mL	185	252	81	0.3	<0.8	1.4	4.1	28	0.5	13	13	1.4	0.7	2.3	4.7
250-500mL	66	109	81	<0.2	<0.8	0.7	2.5	27	<0.2	11	14	1.4	0.7	2.4	4.4
625-875mL	42	74	77	<0.2	<0.8	0.8	1.8	28	<0.2	11	14	1.4	0.7	2.4	4.4
Gelman Capsule															
0-250mL	320	302	72	0.3	1.4	4.8	5.6	27	0.8	11	12	1.5	0.6	2.2	4.4
250-500mL	308	289	89	0.3	<0.8	1.6	3.1	33	0.5	10	14	1.5	0.7	2.4	4.7
500-750mL	153	248	81	0.3	<0.8	2.0	3.0	30	0.4	11	14	1.5	0.7	2.4	4.7
750-1000mL	145	228	85	0.4	<0.8	1.5	2.6	31	0.3	12	14	1.4	0.7	2.4	4.6
1125-1375mL	64	169	82	0.2	<0.8	1.6	2.2	30	<0.2	13	14	1.5	0.7	2.4	4.6
1375-1625mL	75	143	83	0.3	<0.8	1.4	2.2	30	<0.2	12	14	1.5	0.7	2.4	4.5
1625-1875mL	50	142	81	<0.2	<0.8	0.9	2.3	28	<0.2	11	13	1.5	0.7	2.4	4.3
1875-2125mL	54	105	80	<0.2	<0.8	0.8	0.9	27	<0.2	12	14	1.2	0.7	2.4	4.4
2250-2500mL	54	98	79	<0.2	<0.8	0.8	0.8	28	<0.2	12	14	1.5	0.7	2.4	4.4
2500-2750mL	45	95	79	<0.2	<0.8	0.7	0.8	28	<0.2	12	14	1.4	0.7	2.4	4.4
2750-3000mL	44	93	79	<0.2	<0.8	0.3	0.6	27	<0.2	10	14	1.4	0.7	2.3	4.4
3000-3250mL	44	87	80	0.2	<0.8	0.6	1.0	28	<0.2	11	14	1.4	0.7	2.4	4.3

[5]Analytical methods included inductively coupled plasma-mass spectrometry (MS) and iductively coupled plasma optical emission spectrometry (OES).

[6] The concentrations for Be (0.6μg/L), Cr (0.3μg/), Mo (0.6μg/), Ag (0.2μg/), Cd (0.2μg/), Sb (0.2μg/), U (0.1μg/L), Li (1μg/L), and V (3μg/L) were excluded because they were all below the method detection limit (given in parentheses).

[7]The concentration of suspended sediment was 39 mg/L.

[8]Filters were a 47mm, 0.40-μm Nuclepore, a 142mm, 0.45-μm MicroFiltration Systems, and a 47mm, 0.45-μm Gelman capsule.

[*]All filters were washed/preconditioned with deionized water (see Horowitz, et al., 1994). Blanks were run through each filter prio processing any samples and were either below the method detection limit and/or insignificant relative to the measured concentratio

Table 2: Comparison of Selected Dissolved Trace Element Concentrations in sample Filtrates From the Mississippi River at St. Francisville and the Tangipahoa river at Robert, Louisiana,
[all concentrations in μg/L exept Ca, Mg, Na,and Si (mg/L)]

Table 3. *Minimum, Maximum, Mean, Standard Deviation, and Median Concentrations for Suspended Sediment from the Arkansas River in 1989 and 1990.*

	1989					1990					1989/1990				
Parameter	Min.	Max.	Mean	Std. Dev.	Median	Min.	Max.	Mean	Std. Dev.	Median	Min.	Max.	Mean	Std. Dev.	Median
Time Btwn. Samples (Days)	0.5	13	3.4	3.2	2.5	0.5	39	12.4	11.4	10.5	0.5	39	7.0	8.7	3.0
Discharge ($m^3 s^{-1}$)	9.91	48.4	32.4	8.16	31.0	3.71	114.4	19.2	19.3	9.91	3.68	114.4	23.0	17.9	21.4
Suspended Sediment (mg l^{-1})	19	600	258	185	216	9	1700	464	570	111	9	1700	341	396	209
Cu (mg kg^{-1})	36	62	49	7	50	33	111	58	23	50	33	111	53	16	50
Pb (mg kg^{-1})	64	133	100	19	95	28	327	103	67	92	28	327	101	44	94
Zn (mg kg^{-1})	390	1270	780	255	780	215	1050	625	223	630	215	1270	720	252	740
Cd (mg kg^{-1})	1.7	7.6	3.9	1.6	3.7	1.0	5.0	3.3	1.3	3.4	1.0	7.6	3.7	1.5	3.7
Co (mg kg^{-1})	7	17	12	2	12	10	39	14	6.2	12	7	39	12	4	12
As (mg kg^{-1})	3.8	8.5	6.4	1.2	6.4	4.1	13.6	7.1	2.2	6.8	3.8	13.6	6.7	1.7	6.5
Sb (mg kg^{-1})	0.6	1.2	0.8	0.1	0.7	0.5	2.6	1.1	0.5	0.9	0.5	2.6	0.9	0.4	0.8
Se (mg kg^{-1})	0.3	1.7	0.7	0.3	0.6	0.4	3.0	1.3	0.8	0.8	0.3	3.0	0.9	0.6	0.7
Fe (wt. %)	2.2	4.4	3.4	0.4	3.5	2.8	4.5	3.4	0.4	3.4	2.2	4.5	3.4	0.4	3.4
Mn (wt. %)	0.08	0.17	0.13	0.02	0.13	0.08	0.27	0.14	0.05	0.13	0.08	0.27	0.13	0.04	0.13
Al (wt. %)	6.5	9.0	7.2	0.5	7.1	5.1	8.2	6.9	0.7	7.1	5.1	9.0	7.1	0.6	7.1
Ti (wt. %)	0.22	0.44	0.35	0.05	0.34	0.25	0.4	0.33	0.04	0.33	0.22	0.44	0.34	0.04	0.34

Table 3: Minimum, Maximum, Mean Standard Deviation, and Median Concentrations for Suspended Sediment from the Arkansas River in 1989 and 1990.

Table 4. Calculated Fluxes for Suspended Sediment and Associated Trace Elements (Tonnes) for the Arkansas River in 1989 and 1990.

Sample Date/Type	Days	SS	Cu	Pb	Zn	Cd	Co	As	Sb	Se	Fe	Mn	Al	Ti
1989 Comp														
Total (30 Samples) w/Sample Q (1)	103	59,998	2.85	5.91	40.2	0.20	0.71	0.39	0.045	0.035	2,071	72.3	4,394	220
Total (30 Samples) w/Daily Mn Q (2)		58,998	2.80	5.81	39.6	0.20	0.70	0.38	0.044	0.034	2,035	71.1	4,318	217
Total/2 (15 Samples) w/Sample Q (3)		80,934	3.69	7.04	50.6	0.24	1.02	0.47	0.057	0.044	2,807	90.6	5,801	308
Total w/Mn.Q/period (4)*		56,767	2.69	5.55	38.0	0.19	0.67	0.37	0.042	0.033	1,962	68.0	4,149	209
Total w/Mn.Q/period/2 (15 Samples) (5)		71,196	3.24	6.20	44.8	0.21	0.89	0.42	0.051	0.040	2,469	79.7	5,102	269
Total w/Sample Q, SS w/Mn Chem/all (6)		56,767	3.01	5.73	40.9	0.21	0.68	0.38	0.051	0.051	1,930	73.8	4,030	193
Total w/Sample Q, SS w/Md Chem/all (7)		56,767	2.84	5.34	42.0	0.21	0.68	0.37	0.045	0.039	1,930	73.8	4,030	193
Total w/Sam SS, w/Mn Q & Chem/all (8)		52,709	2.79	5.32	38.0	0.20	0.63	0.35	0.047	0.047	1,792	69.0	3,742	179
Total w/Sam SS, w/Md Q & Chem/all (9)		50,539	2.53	4.75	37.4	0.19	0.61	0.33	0.040	0.035	1,718	66.0	3,588	172
Total w/Mn Q, Mn SS, Mn Chem/all (10)		74,482	3.65	7.45	58.1	0.29	0.89	0.48	0.060	0.050	2,532	96.8	5,363	261
Total w/Md Q, Md SS, Md Chem/all (11)		59,686	2.98	5.67	46.6	0.22	0.72	0.38	0.040	0.040	2,089	77.6	4,238	203
1990 Comp														
Total (20 Samples) w/Sample Q (1)	250	252,230	11.0	22.7	137	0.74	3.4	1.88	0.20	0.20	9,033	316	18,137	897
Total (20 Samples) w/Daily Mn Q (2)		214,391	9.4	19.3	113	0.59	2.9	1.56	0.17	0.17	7,686	268	15,467	769
Total/2 (10 Samples) w/Sample Q (3)		273,000	12.4	24.9	147	0.78	3.7	2.06	0.22	0.22	9,826	348	19,680	961
Total w/Mn.Q/period (4)*		157,237	6.7	15.0	94.7	0.51	2.0	1.16	0.12	0.13	5,422	186	11,270	547
Total w/Mn.Q/period/2 (10 Samples) (5)		170,549	7.6	16.9	104	0.56	2.1	1.29	0.14	0.14	5,952	209	12,257	586
Total w/Sample Q, SS w/Mn Chem/all (6)		157,237	8.3	15.9	113	0.58	1.9	1.05	0.14	0.14	5,346	204	11,164	535
Total w/Sample Q, SS w/Md Chem/all (7)		157,237	7.9	14.8	116	0.58	1.9	1.02	0.13	0.11	5,346	204	11,164	535
Total w/Sam SS, w/Mn Q & Chem/all (8)		100,031	5.3	10.1	72	0.37	1.2	0.67	0.09	0.09	3,401	130	7,102	340
Total w/Sam SS, w/Md Q & Chem/all (9)		51,715	2.6	4.9	38.3	0.19	0.6	0.34	0.04	0.04	1,758	67.0	3,672	176
Total w/Mn Q, Mn SS, Mn Chem/all (10)		192,111	11.7	20.7	126	0.66	2.8	1.43	0.22	0.26	6,838	282	13,877	664
Total w/Md Q, Md SS, Md Chem/all (11)		23,796	1.22	2.25	15.4	0.08	0.3	0.17	0.02	0.02	831	31.8	1,735	80.6

***Considered the best estimate of transport for the period.**

(1): Calculated assuming that the discharge (Q) and the suspended sediment and chemical concentrations determined at the time of sampling were constant for the entire sampling interval.

(2): Calculated assuming that the mean daily discharge (Q) for the day the sample was obtained as well as the suspended sediment and chemical concentrations were constant for the entire sampling interval.

(3): Calculated as in (1) but only using every other sample.

(4): Calculated using the mean discharge (Q), and assuming that the suspended sediment and chemical concentrations for the sample were constant, for the sampling interval.

(5): Calculated as in (4), but only using every other sample.

(6): Calculated using the mean discharge for the sampling interval, with the sample suspended sediment concentration, and the mean chemistry for all the Arkansas River samples (50).

(7): Calculated as in (6) but using the median chemical concentrations.

(8): Calculated using the sample suspended sediment for each interval along with the mean discharge for the year and the mean chemistry for all the Arkansas River samples (50).

(9): Calculated as in (8) but using the median discharge for the year and the median chemistry for all the samples.

(10): Calculated using mean discharge and suspended sediment concentrations for the year, and the mean chemistry for all the Arkansas River samples.

(11): Calculated as in 10 but using the median discharge and suspended sediment concentration for the year and the median chemistry for all the Arkansas River samples.

Table 4: Calculated Fluxes for Suspended Sediment and Associated Trace Elemntents (Tonnes) for the Arkansas River in 1989 and 1990

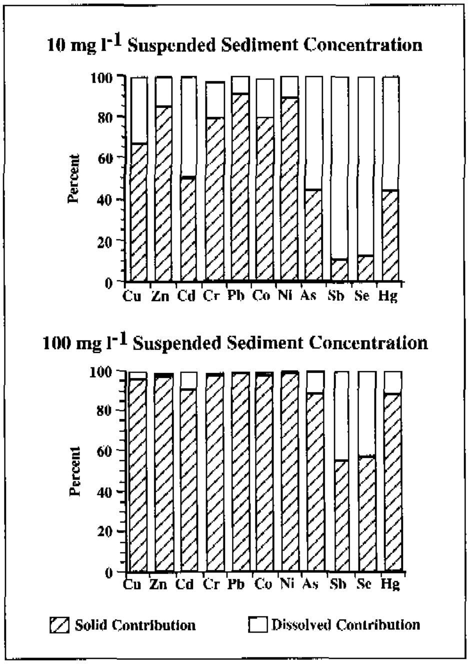

Figure 1: Solid phase contributions to the concentration of trace elements for a typical whole-water (suspended plus dissolved phases) sample for two suspended sediment concentrations . Source after, Horowitz, 1991.

affected by either filtration artifacts or dilution (Table 2). The first group includes Al, Fe, Ni, Cu, and Zn. The general pattern always was the same; the lowest concentrations occurred in the Nuclepore filtrates, the next highest concentrations occurred in the MFS filtrates, and the highest concentrations occurred in the Gelman filtrates. This pattern is proportional to the respective surface areas of the three filters, and may imply a correlation with the filtration rate and/or clogging.

The effects on Al and Fe are obvious (Table 2). Note that the 8.2 µg l^{-1} Al level in the 100 ml Nuclepore filtrate was not reached in the MFS filtrate until 500 ml had been processed, and was not reached in the Gelman filtrate until over 2000 ml had been processed. In fact, the Gelman filtrates eventually contained lower Al levels than the Nuclepore filtrates, but only after 2600 ml had been processed. The patterns for Ni, Cu, and Zn are not as pronounced as for the Al and Fe. The Cu concentration (50 µg l^{-1}) in the first 250 ml Gelman aliquot was 'verified' (it was quantitated in both halves of a split field sample) but almost certainly represents some form of contamination introduced during sample handling. Even so, the second Gelman aliquot (250-500 mL) still contains more Cu than either the first Nuclepore or MFS aliquots. There appears to be a correlation between filter surface area and the Fe and Al concentrations in the filtrates. However, although filtration artifacts appear to have affected Ni, Cu, and Zn, and despite the sorptive capacity of Fe and Al oxides and hydroxides for these elements, there does not appear to be a direct correlation between Ni, Cu, and Zn concentrations and filter surface area. These conconclusions are similar to others reported elsewhere (Horowitz, et al., 1992; Karlsson, et al., 1994). Co, Mo, Pb, and possibly Cr concentrations appear to have been affected by a different type of artifact associated only with Gelman filters (Table 2), and may be a function of their high surface area and/or composition (polyethersulfone). Note that the concentrations of these elements increased after processing 1500 ml. This may indicate that some proportion of these elements had been sorbed to the filter during initial processing, then, once all the potential sorption sites had been filled, higher concentrations began to appear in the filtrate.

The concentrations of several elements (e.g., Sr, Ba, and Ca) appear to have been affected by dilution due to some entrained deioninzed water that was used to condition the filters. Note the concentration increases in the second and third filtrate aliquots, relative to the first aliquot, for these elements for Gelman and MFS filters (Table 2). This effect has been noted before (Horowitz, et al., 1994). If dilution is affecting trace element concentrations, it should affect them all equally; however, as a result of low concentrations and/or limited analytical sensitivity, it is not significant below 50 μg l^{-1}. On the other hand, this may indicate that some proportion of these elements initially had been sorbed to the filter, then, once all the potential sorption sites had been filled, higher concentrations began to appear in the filtrate. Although data were not included, the concentrations of a variety of dissolved nutrients (nitrate, nitrite, ammonium, orthophosphate, and total phosphorus) did not appear to be affected by filtration artifacts or dilution. In fact, constituents occurring at ≥mg l^{-1} concentrations apparently are not affected by filtration artifacts.

The data from the Tangipahoa are similar to those from the Mississippi (Table 2). In most cases, the elements affected by filtration artifacts in the Mississippi sample also were affected in the Tangipahoa sample, with the exception of Pb. The Pb in the Tangipahoa sample appears to be affected by filtration artifacts in the same way as Al, Fe, Cu, and Zn, whereas in the Mississippi sample, Pb did not appear to be affected by the exclusion of colloids due to filter clogging, but may have been affected by initial sorption onto the Gelman filter. The sorption artifacts noted in the Mississippi sample for Cr, Co, and Mo, could not be evaluated in the Tangipahoa sample due to lower concentrations. All the unaffected elements, or those possibly affected by dilution, in the Mississippi sample also displayed similar patterns in the Tangipahoa sample (Table 2).

Three viable options are available for dealing with filtration artifacts; two entail substantive changes in the current definition of dissolved constituents. 1) The artifacts can be reduced/eliminated by using filters with very high surface areas (e.g., capsule filters) and collecting initial aliquots for the quantitation of artifact-affected constituents. 2) Artifact-induced differences in trace element concentrations can be limited by pretreating the sample (e.g., filtration or centrifugation); prior to filtration with a 0.45-μm membrane. However, pretreatment enhances the chances for random contamination due to increased sample handling, and markedly lowers a number of trace element concentrations in the filtrates (Horowitz, et al., 1996). 3) Colloids should be treated as contaminants which should be excluded from 'dissolved' samples. Although there is controversy over what constitutes a colloid, current data indicate that material coarser than 0.015 to 0.005-μm would have to be removed (Karlsson, et al., 1994; Taylor and Shiller, 1995) This approach represents a substantive departure from the current 'dissolved' definition and would require the use of multiple filters, more expensive equipment (e.g., tangential-flow filtration systems), or the use of such subjective procedures as 'exhaustive filtration' (Shiller and Boyle, 1987; Karlsson, et al., 1994; Taylor and Shiller, 1995; Horowitz, et al., 1996). The USGS has elected to follow the first option, which permits continued use of the currently accepted operational definition of a 'dissolved' constituent (Horowitz, et al., 1994).

SUSPENDED SEDIMENT SAMPLES

Recent evidence indicates that the majority of fluvial trace element transport occurs in association with suspended sediment. Even in waters with suspended sediment concentrations less than 10 mg l^{-1}, the sediment can represent the major carrier for many trace elements; at 100 mg l-1, the solid phase dominates the transport of most trace elements (Fig.1). Typically, fluvial suspended sediment and associated trace elements display marked short-term spatial and temporal variability. This has led to the view that the determination of annual fluxes requires the commitment of substantial resources to permit, high frequency, depth- and width-integrated isokinetic sampling, and subsequent chemical analyses. In turn, this is so resource limiting as to preclude accurate annual flux estimates. However, in the context of annual fluxes, short-term spatial and temporal variations (on the order of a few hours or days) are relatively insignificant, and probably do not have to be measured (de Vries and Klavers, 1994; Horowitz, 1995).

The determination of annual suspended sediment-associated trace element fluxes requires data on discharge, suspended sediment concentration, and suspended sediment chemistry. Of these, accurate chemical data have been viewed as the most difficult and expensive to procure. Detailed studies on the Arkansas River at Portland, Colorado, U.S.A., indicate that the differences in median chemical concentrations, between the first and second years of the study, were less than the imprecision of the chemical analyses themselves (Table 3; ±10%). This occurred despite the fact that for the study period, discharge ranged from 3.7 $m^3 s^{-1}$ to 114.4 $m^3 s^{-1}$ and median discharge differed by a factor of three (from 9.9 $m^3 s^{-1}$ to 31 $m^3 s^{-1}$) whereas suspended sediment concentrations ranged from 9 mg l^{-1} to 1700 mg l^{-1} and median suspended sediment levels differed by a factor of 2 (from 111 mg l-1 to 216 mg l-1). Subsequent calculations of annual trace element fluxes for the site indicate that the substitution of mean/median chemical values for actual individual sample chemical data, used in conjunction with individual values for discharge and suspended sediment concentration, introduced errors only on the order of ±20% (Table 4).

Although there is controversy over how to calculate annual fluxes, all current methods require data on discharge, suspended sediment, and dissolved and sediment-associated constituent concentrations (e.g., de Vries and Klavers, 1994). These data must be available on a site-by-site basis for an effective flux-based monitoring network. Typically, the USGS determines discharge based on an established stage-discharge relation, and the use of stage recorders that monitor levels at least hourly. 'Continuous' data on suspended sediment concentration requires the use of either automatic samplers or some type of nephelometer/transmissometer. However, these produce point sample data which, typically, are not representative of fluvial suspended sediment cross-sectional distributions. Thus, appropriate site-specific correction factors (sometimes called 'box' equations) are required to relate point sample suspended sediment concentrations to depth- and width-integrated concentrations.

The need for 'continuous' data on discharge and suspended sediment concentrations, as well as mean/median dissolved and suspended sediment associated trace element concentrations for each long-term flux-monitoring site, can be met by conducting an initial site characterization. Resource limitations, as well as a need to cover a variety of site-specific hydrologic conditions, implies that these characterizations will probably take from two to three years and entail both calendar and event-based sampling. Once an initial site characterization is completed, actual sample collection and subsequent analyses will be substantially reduced, and only are needed to: 1) ensure the continued validity of the stage discharge relation, 2) the suspended sediment 'box' equations relating point measurements/point samples to depth-integrated concentrations, and 3) the mean/median dissolved and suspended sediment-associated chemical concentrations. Substantive site-specific changes occurring during the post characterization phase may necessitate a subsequent intensive sampling and measurement phase to recharacterize the site. An important ancillary benefit of the site characterization approach is that variances associated with the resulting long-term data would be known. Thus, end users could understand the limits that should be placed on forthcoming data and data interpretations.

CONCLUSIONS

The USGS NASQAN program has recently been redesigned as a flux-based monitoring network to characterize large U.S. river basins for a variety of chemical constituents. The development of this network is predicated on a two-stage approach: 1) intensive calendar- and event-based measurements and sampling to establish methods for continuously determining discharge and suspended sediment concentrations as well as mean/median dissolved and suspended sediment-associated constituent concentrations, followed by 2) substantially reduced sampling intended to insure the continued validity of the interrelations established during the initial stage. Based on this approach, a cost effective flux-monitoring network can be established and maintained during a period of continuously diminishing resources.

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GROUNDWATER MONITORING NETWORKS IN SPAIN

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ABSTRACT

A Groundwater Monitoring Programme is currently being implemented by the Spanish government, with the purpose of establishing two basic reference networks: one for water level long-term trend assessment and the other for ambient groundwater quality evaluation. The number of observation points in the different hydrogeological units is determined as a function of a set of selected variables. Spatial distribution of wells and boreholes installed in the networks is based on hydrogeological criteria and geostatistical analysis. Construction of user-oriented and problem-oriented networks is envisaged in selected areas.

INTRODUCTION

Spain is one of the most mountainous countries in Europe. Rainfall presents an uneven spatial distribution: average annual precipitation is 1400 mm in the northern river basins, 380 mm in the southeastern basins and 360 mm in the Canary Islands. This uneven distribution in precipitation règime causes significant differences in surface runoff and groundwater avalibility. Areas where semiarid climatic conditions and high water demand prevail, like the Mediterranean coastal region, are particularly affected by episodes of water resources scarcity.

The outcrop of groundwater aquifers covers approximately 35% of the Spanish territory: 16% correspond to porous media aquifers, 11% to karstic areas and 8% to other water bearing formations. Groundwater resources account for 20-25% of total water consumption in Spain, supplying 25% of irrigation demands and drinking water for 31 % of the population. (MOPTMA, 1995).

In accordance with the Spanish Water Act of 1985, the Ministry of Public Works, Transport & Environment has undertaken action to maintain statistics relevant to "the monitoring of inland water evolution with regard to both the quantitative and qualitative aspects". These duties have been transferred to the recently created Ministry of Environment. River Basin Authorities, responsible under the Water Act for the control of water resources and water quality, will be in charge of monitoring network operation and maintenance.

The purpose of the Groundwater Monitoring Programme, one of sixteen groundwater programmes to be developed by the administration in coming years, is an official control network for the *hydrogeological units* (administrative units containing the most relevant aquifers) in the Spanish territory. A six-year period is envisaged for carrying out the programme, the first year being devoted to drafting the projects, and the remaining five for putting the networks into operation (Varela, 1995). Activities in this field aim to correct deficiencies that existed before the Spanish Water Act came into force. The programme will be coordinated by the organisations that have jurisdiction over matters concerning groundwater resources.

EXISTING MONITORING NETWORKS

Groundwater levels, groundwater quality and coastal saline intrusion monitoring networks have been in operation for more than 20 years. The Instituto Tecnológico GeoMinero de España (ITGE) has been responsible for measuring and collecting data, stored in a nation-wide database. On the basis of the number of observation points -1169 for groundwater levels, 713 for water quality- their spatial distribution, their location in areas with intensive groundwater use or contamination problems and the period and frequency of temporal series, the ITGE networks are currently regarded as the "official" networks at state level (MOPTMA-Comisión Europea, 1993).

Mention also should be made of the piezometric network controlled by the Junta de Aguas de Cataluña. The network comprises 361 points, the majority of which were constructed for water level monitoring purposes (Junta d'Aigües, 1995). The Diputaciones Forales of Navarra & Alava have in operation since 1980 piezometric and groundwater quality networks located in several hydrogeological units of their respective territories. (Sánchez, 1996)

Drawbacks associated with the networks controlled by the ITGE, where representativity is concerned, are as follows (Estrela et al.,1995):

- most of the elements within the networks are exploitation wells that were not specifically constructed for control and monitoring purposes;
- over the last ten years the frequency of measurements has gradually decreased;
- numerous hydrogeological units are not subject to monitoring;
- monitoring data have not been published on a continuous basis.

THE GROUNDWATER MONITORING PROGRAMME

As mentioned in the preceding paragraphs, a programme is currently being developed to implement the national control networks in the different hydrogeological units.Work related to this programme got under way in 1994, the first phase consisting of the preparation of the installation, maintenance and operation projects for the piezometric, hydrometry and quality networks in the different river basins (MOPTMA, 1994).

Two types of basic reference monitoring networks will be established:

(a) Piezometric networks, for monitoring groundwater levels in the aquifers. This type of networks enables managers to determine the status of groundwater storage, to forecast the river base flow in relation to groundwater inflows, and to obtain insight into the seasonal and long term trends in quantitative terms.

(b) General surveillance and ambient quality networks, to assess the evolution -in time and space- of the basic parameters which define groundwater quality. The final objective of this type of networks is to gather information on long-term trends in background groundwater quality.

These reference networks will provide information on various factors for the hydrogeological units concerned, namely sustainability of groundwater exploitation and deterioration of groundwater quality in specific zones of the aquifer. Consequently, design requirements will be less strict for this type of network than for problem-oriented ones (i.e. overexploitation or point-source pollution control).
Several elements were taken into consideration in order to standardise network design: spatial

density (number of monitoring sites per area) and frequency of observations, characteristics of piezometers and water sampling wells, instrumentation and methods for gathering, storing and handling data.

At an initial phase, the descriptive characteristics of the hydrogeological units were systemised, and a first estimation of the number of control points required on both networks was made for each unit, on the basis of factors such as outcrop area, aquifer type, magnitude of groundwater resources, degree of exploitation, hydrodynamic parameters, population density, pollution level, etc. (DGOH, 1992).

The criteria adopted for the establisment of the new networks are explained below.

SPATIAL DENSITY AND LOCATION

Spatial density and distribution of observation points are fundamental aspects of monitoring network design. A compromise between construction and maintenance costs and precision in representativity should be envisaged.

Network density will depend on the value or use of groundwater resources, the intrinsic characteristics of the hydrogeological unit and the impacts caused by aquifer exploitation.

The differing characteristics of the hydrogeological units justify a different treatment in terms of monitoring level and needs. In order to increase the degree of objectivity used to quantify these peculiarities, a set of descriptive parameters was defined as a numerical modifier of the initially-established spatial density factor.

The number of *piezometers* to be installed in each hydrogeological unit has been determined according to the following equation:

$$N_p = (\text{unit extension } \Pi_{\text{weighting factors}}) / \text{basin density index}$$

which includes the units area expressed in km^2, an initial common density index (expressed as *n* km^2 per piezometer) and a set of weighting factors related to the following variables: outcrop extension, geology, topography, population density, aquifer exploitation, groundwater recharge régime, aquifer porosity/transmisivity, drainage and water use (see Table1).

PARAMETER	RANGE/VALUE		
Aquifer extension (km^2)	>500=0.7	500<>100=1.0	<100=1.4
Population density	low = 0.8	medium=1.0	high=1.2
Exploitation/recharge (%)	<15=0.8	15<>70=1.0	>70=1.3
Recharge (hm^3/yr)	<10=0.9	10<>100=1.0	>100=1.1
Transmissivity (m^2/day)	>2.000=0.9	200<>2000=1.0	<200=1.1
Drainage pattern	point=0.9	linear=1.0	multiple=1.1
Groundwater use	irrigation=0.9	industry=1.0	drinking=1.1
Topography	mountainous=0.9	medium=1.0	plains=1.1
Geology	simple=0.9	medium=1.0	complex=1.1

Table 1: Weighting factors for piezometric networks

The total product of the weighting factors values has a range of $0{,}25 \leq \Pi \leq 4{,}00$.

Common density standards of 75 and 50 km^2 per piezometer were tested, and, as a result a general density index of 75 km^2 per piezometer adopted. For the Guadalquivir, southern basins and Balearic Islands, one piezometer for each 50 km^2 was considered appropiate. In the case of the Duero Basin, this figure was reduced to one control point every 150 km^2 as the network otherwise would have had an excessive number of piezometers.

Following these criteria, units with an outcrop surface of less than 25 km^2 were not considered (as only 0,5 piezometers would be installed in the unit); in units with outcrop surfaces between 25 and 50 km^2, a maximum of one piezometer will be installed, depending on the value of the unit's weighting factors.

Resulting spatial densities are shown in Table 2.

BASIN	Hydrogeological unit area (km^2)	Existing points	Proposed points	Density ($1/n km^2$)
NORTE	6.263	0	67	1/94
DUERO	54.009	243	226	1/239
TAJO	17.571	33	153	1/115
GUADIANA	12.763	12	144	1/89
GUADALQUIVIR	14.882	122	249	1/57
SUR	5.127	193	113	1/46
SEGURA	8.033	120	98	1/82
JUCAR	23.935	168	277	1/86
EBRO	17.078	80	158	1/108
MALLORCA	2.798	127	72	1/39
MENORCA	415	0	9	1/46
IBIZA	451	11	13	1/35
FORMENTERA	82	0	2	1/41
TOTAL	150.674	1.109	1.581	
Density		1/136 km^2	1/95 km^2	

Table 2: Current and proposed piezometric networks

The *groundwater quality* monitoring networks present a greater degree of complexity. The parameters to be controlled will be field parameters (temperature, pH , EC,DO) and major ions: Cl , SO_4^-, CO_3 , H^-CO_3. NO_3^-, Na^+, K^+, Ca^{++}, Mg^{++}, NO_2^-, NH_4^+. In problem-oriented networks selected parameters will be measured depending on the specific purpose for which they are established. Specifications will be set in accordance with the guidelines included in the appropiate EC directives, already incorporated in the Spanish Water Act.

Following criteria analogous to those applied in estimating piezometric network density, the number of control wells to be installed in each hydrogeological unit is determined according to the same equation. In this case, the weighting factors are related to the following variables: unit area, aquifer exploitation, ratio of transmisivity/porosity, water use, population density, pollution level and groundwater total dissolved solids concentration. The respective weighting factors are shown in Table 3. The range of weighting factor product is $0{,}2 \leq \Pi \leq 5{,}0$. Comparison between current and proposed initial densities is depicted in Table 4.

The spatial densities proposed for the different hydrogeological units constitute a basic reference for network design. The final figures will be set on a case-by-case basis, upon assessment of aquifer heterogeneities and overlying land-use patterns.

PARAMETER	RANGE/VALUE		
H. unit extension (km^2)	>500=0.8	500<>100=1.0	<100=1.3
Population density	low=0.8	medium=1.0	high=1.2
Exploitation/recharge (%)	<15=0.8	15<>70=1.0	>70=1.3
Transmisivity (m^2/day)	>2.000=0.9	200<>2.000=1.0	<200=1.1
Groundwater utilization	irrigation=0.9	industry=1.0	drinking=1.1
Groundwater pollution	low=0.9	medium=1.0	high=1.1
T.D.S.	<500=0.6	500<>2.000=1.0	>2.000=1.6

Table 3: Weighting factors for groundwater quality networks

BASIN	Hydrogeological unit area (km^2)	Existing points	Proposed points	Density (1/$n$km^2)
NORTE	6.083	15	34	1/178
DUERO	53.919	41	136	1/396
TAJO	17.525	100	89	1/197
GUADIANA	12.763	81	95	1/134
GUADALQUIVIR	14.979	23	189	1/78
SUR	5.022	21	70	1/72
SEGURA	7.902	52	55	1/144
JUCAR	23.843	118	163	1/146
EBRO	16.938	105	97	1/175
MALLORCA	2.798	2	49	1/57
MENORCA	415	2	7	1/59
IBIZA	451	6	9	1/50
FORMENTERA	82	0	1	1/82
TOTAL	149.730	566	994	
Density		1/265km^2	1/151km^2	

Table 4: Current and proposed groundwater quality networks

For implementation, an estimated 1.800 groundwater level observation points will be put into operation. 40% of these being piezometers constructed during the the programme. The ambient quality network will include 1.200 sampling sites, most of which will be exploitation wells or piezometers. (Sánchez, 1996).

The *location* of the network elements will be decisive in achieving the objective of representativity. The following hydrogeological criteria will be applied:

(a) piezometric networks

- geographic distribution will be conditioned by groundwater flowpaths;

- piezometers will be located in areas of aquifer recharge and in zones where the effects of heavy pumping can be assesed:

- spatial density will be diminished in constant-head areas;

(b) groundwater quality networks

- *densities in aquifer edge areas can be lower than mean value for aquifer*
- in exploitation zones and in areas where potential sources of pollution are located, densities will be increased;
- maximum concentration of observation wells should correspond to discharge areas in the vicinity of springs, rivers, lakes, wetlands and coastal water bodies.

INCORPORATION OF PIEZOMETERS FROM EXISTING NETWORKS

The selection of currently-operating piezometers to be incorporated into the new networks will be based on hydrogeological and statistical analysis.

Hydrogeological assessment will be performed at an initial stage, and will serve to determine piezometer groups in the same aquifer that should be the object of joint statistical analysis. This will avoid utilizing groups of heterogeneous data which could lead to statistically invalid results.

Three types of analysis should be performed:

- "cluster" type group analysis, aimed at reducing information redundance;
- piezometric kriging referred to the period of maximum available data; missing data, as well as measures of variance, will be analyzed to determine the structure of the piezometric data's variograms, piezometric maps and errors.
- kriging in the most recent period available, following a process similar to that describe above.

FREQUENCY OF MEASUREMENTS

Sampling frequency will be conditioned by two factors: expected temporal variation of the parameters to be monitored and availability of funds.

In the case of groundwater quality networks, conditioning factors to be taken into consideration include rainfall pattern and irrigation and fertilising regimes. Two to four analyses a year are generally sufficient for basic quality monitoring schemes; in some coastal zones or areas where a specific problem has been identified, measurements may have to be made more frequently, or a specific parameter may even have to be monitored on a continual basis.

With respect to piezometric networks, monthly, and in some cases, quarterly measurements are sufficient for regional control; some automated continuously-recording piezometric stations will be installed in order to provide more precise information on specific time periods.

Manual level measuring and water sampling will be the established procedure in most of the cases. Automatic control operations, using transducer-converter devices, will be envisaged in the following situations:

- *karstic aquifers, subject to rapid responses to rainfall episodes;*
- areas of difficult access;
- water supply or irrigation wells integrated in operation and control systems.

OWNERSHIP OF MONITORING WELLS

Piezometers will be in the public domain, preferably state-owned. Privately-owned observation wells will be incorporated into the network only as an exception.

Groundwater quality can be monitored in exploitation wells already in operation, preferably owned by cities, municipalities, user associations, etc. In all cases, the selected existing wells have to meet the representativity requirement set for the aquifer.

Ideally, all control points should be used exclusively as elements of the network and should be owned by the administration. In practice, however, private wells have been used traditionally in Spain (as well as in other European countries), for groundwater monitoring purposes. This situation will definitely continue in the short and mid-term. Experience shows that when private owners act as collaborators, network operation efficiency usually improves.

Water-supply wells in urban areas would be particularly suitable for quality control purposes: pumping is controlled regularly; the wells are usually in the public domain; records are kept accurately and the results of efficient control are perceived promptly.

Finally, it should be noted that monitoring sites could be used in some cases for both piezometric and quality control purposes, especially for hydrogeological units with reduced extension and small volume groundwater resources, where only one control point will be installed.

OTHER MONITORING NETWORKS

In addition to the networks described above, user-oriented and problem-oriented monitoring networks will be established.

Protection of drinking water sources will be the objective of surveillance networks to be controlled by local authorities. Standards compliance networks will be constructed in selected areas where point source emissions and discharges are located. Nitrate pollution monitoring, as required under Directive 91/676/EEC, will be the subject of a national scale programme.

In those hydrogeological units legally declared as overexploited, compliance control networks will also be established.

The design and characteristics of these networks will be decided by the competent authorities at the initial phase of the respective groundwater protection programmes.

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MONITORING AND ASSESSMENT OF LAKES AND WATERCOURSES IN SWEDEN

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ABSTRACT

The Swedish environmental monitoring system is currently being revised. Ten programme areas have been identified: air, seas, lakes and watercourses, groundwaters, wetlands, mountains, forest areas, agricultural land, landscapes, health and urban areas. Within each programme area, a set of subprogrammes have been established that will be implemented at national and/or regional levels.

The monitoring programme for lakes and watercourses places two aspects in focus: (i) biodiversity and (ii) the use of water as a resource for fisheries, drinking water, etc. In addition, the programme should indicate losses of material from terrestrial ecosystems and the input of material to marine areas.

Chemical and biological indicators relevant to these various aspects have been selected, based on considerations of ecological relevance, variability and cost. At the national level, the programme has a three-tiered nested design. At the first tier, synoptic lake and watercourse surveys consisting of a minimum of 1000 sites, respectively, will be performed at five year intervals. Measurements of indicator variability from these data sets will allow for the determination of large-scale spatial patterns and classification of the Nation's freshwater resources. A second tier, consisting of ca. 100 temporal reference sites will allow for statements on temporal change. A third tier, consisting of integrated trophic level monitoring, will allow for the assessment of the importance of trophic-level interactions on the natural variability of ecological indicators. The third tier will normally not be included at the regional level, whereas the first two normally will. Separate subprogrammes have been proposed to address the monitoring of surface waters for recreation, fishery and drinking water purposes, as well as the monitoring of waters for rare and endangered species and areas of national interest.

Results from the monitoring programme will be made available in an annual State of the Environment Report, as official environmental statistics and by the dissemination of results to regional authorities and the general public.

INTRODUCTION

Sweden's environmental policy is being developed along the following main lines (Anonymous, 1991, see also Wiederholm, 1993):

- Increased international cooperation to limit transboundary and global environmental problems,
- Improved access to and structuring of information and statistics, and better follow-up of the state of the environment,
- Increased sector responsibility and decentralisation, including personal responsibility and corporate and municipal participation.

	Number of lakes	watercourses	Parameter(s)(1)	Co-ordinator(2)
Water quality of transport to marine areas		49	PC	SEPA
Water quality of relatively unperturbed watercourse reaches		39	PC	SEPA
Material transport to Sweden's four great lakes		35	PC	SEPA
Water quality of Sweden's four great lakes	4		PC, B	SEPA
Chemical monitoring of ecoregion, reference lakes	190		PC	SEPA
Biological monitoring of ecoregion, reference lakes	26		PC, B	SEPA
Monitoring of limed lakes and watercourses	14	7	PC, B	SEPA
National lake survey(3)	4000		PC	SEPA
Integrated monitoring of small catchment areas(4)		5	PC, B	SEPA
Monitoring of discharge		418	PC	SMHI
Monitoring of lake and watercourse temperature and ice conditions	320	87	P	SMHI
Sediment transport		19	P	SMHI
Small research basins(5)				SMHI
Monitoring of pelagic fish populations is Sweden's five largest lakes	5			FISH
Regional, catchment-wise monitoring	670	1500	PC, B	REGION
Regional, effect follow-up of liming	7500	700	PC, (B)	REGION

(1) PC = physico-chemical and B = biological monitoring

(2) Co-ordinator responsible for funding and/or implementation (SEPA = Swedish Environmental Protection Agency; SMHI = the Swedish Meteorological and Hydrological Institute; FISH = Fisheries Board of Sweden; REGION = regional monitoring).

(3) the1990 NLS included some 4000 randomly selected lakes

(4) integrated terrestrial and aquatic monitoring programme of 18 small catchment areas

(5) hydrological, physical, and climatical data have been collected from 20 relatively small (1 - 200 km^2) research that are relatively homogeneous in regime patterns, topography, geology, and vegetation

Table 1: Overview of national and regional environmental monitoring programmes for lakes and watercourses at the onset of SEPA's revision (Wiederholm 1992, Löfgren 1993, Anonymous 1996)

Monitoring is vital to the second of these lines of development. It serves to follow fluctuations in the state of the environment and to identify changes resulting from human activities and, specifically, to provide the data needed to identify environmental problems, set targets for an environmentally sound development of society, draw up priorities and decide on measures, and to follow up the effects of such measures.

With the recognition that monitoring is fundamental to environmental protection and nature conservation work, it follows that monitoring must be aimed at relevant targets and produce representative, accurate, easily available and understandable results. The Government therefore initiated a revision and expansion of the current monitoring system, and the task to achieve this was given to the Swedish Environmental Protection Agency (SEPA).

As a result of the SEPA's work, ten programme areas have been identified: air, seas, lakes and watercourses, groundwaters, wetlands, mountains, forest areas, agricultural land, landscapes, health and urban areas. Within each programme area, a set of subprogrammes have been established that will be implemented at regional and/or national levels. The present paper describes the monitoring of lakes and watercourses.

PREVIOUS MONITORING

PROGRAMMES

National monitoring of lakes and watercourses in Sweden has been performed on various spatial and temporal scales, with the timing and frequency of sampling being dependent on the goals of the respective subprogrammes (Table 1). Monitoring has basically addressed three problem areas: (1) discharges of primarily nutrients and metals to marine areas and Sweden's four largest lakes; (2) eutrophication; and (3) acidification of inland lakes and watercourses.

Regional monitoring programmes have been ongoing in some areas since the early 1950's, and much of Sweden is now covered by catchment-wise coordinated regional monitoring programmes. Regional programmes also monitor the chemical and biological effects of liming activities. In addition, monitoring occurs at a number of individual sites, e.g. lakes and watercourses that serve as public water supplies, swimming areas, or waterbodies subject to pollution by local point sources.

EXPERIENCE GAINED FROM SEPA'S PROGRAMMES

Use and dessimination of monitoring data

Data from the monitoring programmes have been used for environmental assessments at regional and national levels since the 1960's, including SEPA's action programmes on freshwaters and the marine environment (SEPA, 1990a, 1990b, 1994). The data have also been used as a basis for national water quality criteria and the establishment of environmental goals for lakes and watercourses. Moreover, the experiences gained within the monitoring programmes on sampling methodology have resulted in recommendations and instructions concerning standardized methodology for local and regional monitoring.

The Department of Environmental Assessment (formerly SEPA's Freshwater Section) has been responsible for the development and management of the national freshwater data base. SEPA and county authorities have received annual data compilations. Some of the material has also been included in the national environmental data-base system. Data has also been made available on request in hardcopy or digitised form at no or a nominal cost to other users (municipalities, consultants, researchers, various agencies, etc.).

International and user evaluations

An international peer review of Sweden's environmental monitoring program was performed in 1985 (SEPA 1986a). The review-group found the program to be unique in many respects. It was considered as being a well established and an all around good environmental monitoring program with "overall good returns for the investment". It was also considered as a supplier of environmental information on an international/global scale.

The lake and watercourse programmes were deemed as having very high standards, and it was considered imperative that these programmes continue. It was suggested that a decrease in the number of sites and/or frequency of sampling might be possible, and that stations not considered necessary for the national programme should then be financed regionally. It was stressed that greater emphasis should be placed on use of biological monitoring variables, measurements of trace metals and organic pollutants, and sediment analysis of reference lakes.

A user evaluation summarised the experience of sector authorities, branch organisations, the various sections of SEPA, county authorities, municipalities and consultants (SEPA, 1986b), and pointed out areas where improvements were needed. For example, the time lag between sampling and data availability, and the low frequency of biological and metal analyses were noted, and there was a general agreement on the need for more synthesis and interpretation of data, as well as increased analysis of regional data.

EXPERIENCE GAINED FROM OTHER PROGRAMMES

Data from SMHI's (Swedish Meteorological and Hydrological Institute) sampling networks have been extensively used as a basis for hydrological mapping, design and forecasting, and delivered regularly to the SEPA's programmes for use in further analyses of material transport in rivers. Runoff stations have also been used for calibration of hydrological models that, in turn, have been used to calculate runoff at ungauged sites as a support to water quality programs.

Many of the regional programmes have been found to be useful for regional and local assessments of surface water quality (Löfgren, 1993). However, improvements were often found necessary on a number of points. For example, operational goals should be more clearly defined and reports should be more reflective of anthropogenic impacts and be more easily comprehensible for administrators and the general public.

SEPA REPORT - ENVIRONMENTAL MONITORING IN THE 21ST CENTURY

SEPA (1990c) listed a number of changes that should be considered in a revised freshwater programme: sampling network, choice of variables, frequency of sampling, co-ordination of national and regional monitoring, increased use of biological variables. These suggestions largely reiterated and underlined the comments and suggestions of the international expert group.

REVISED NATIONAL MONITORING PROGRAMMES FOR LAKES AND WATERCOURSES

PROGRAMME STRUCTURE

The future monitoring program (Wiederholm, 1992) was developed addressing the overall objective for the Nation's freshwater resources: "Native species should occur in stable, well-balanced populations, and pollution should not limit the value of water as a fisheries, recreation, and raw water resource" (SEPA, 1990d). To meet the overall objective, and the more specific monitoring goals mandated in SEPA (1990c), requires an unbiased characterisation of the status of lakes and watercourses, and a determination of whether populations are improving, degrading or have remained unchanged, as well as the rate at which these processes may be occurring. To achieve this, a nested sampling design with three tiers was developed with: (i) national lake and watercourse surveys every 5 yr using physico-chemical and biological indicators to determine spatial patterns, (ii) annual monitoring of temporal reference lakes and watercourses to determine among-year variability and trends of physico-chemical and biological indicator metrics, and (iii) annual monitoring of physico-chemical and biological indicators of lakes and watercourses to

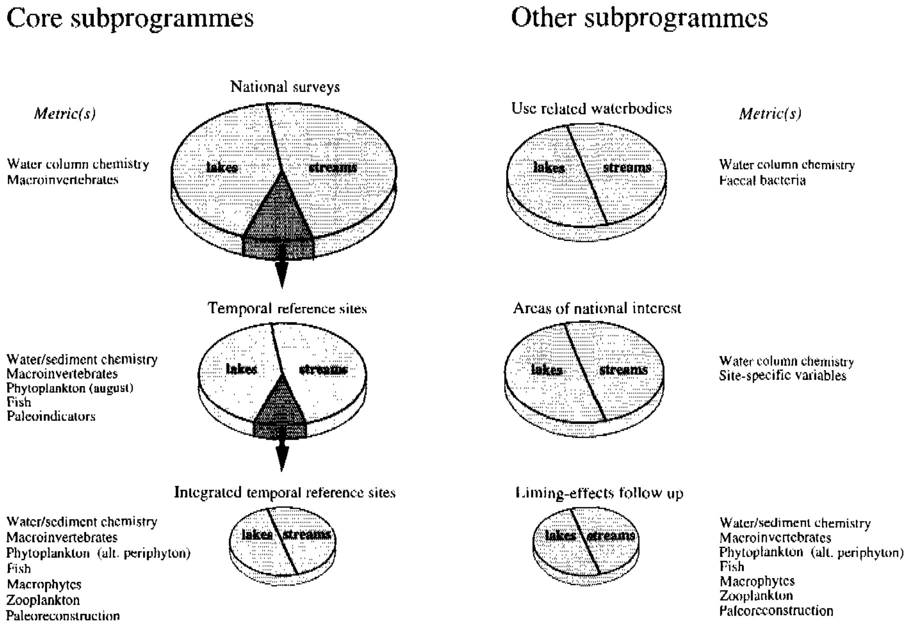

Figure 1: Schematic diagram showing the nested design of the core and other subprogrammes of the national lake and watercourse monitoring programme.

better understand the importance of interactions between physico-chemical and biological processes in lakes and watercourses and between sites and their watersheds (e.g. determine the importance of complex, trophic-level interactions on selected indicator metrics) (Wiederholm, 1992) (Fig. 1). Moreover, a number of lakes used as reservoirs for water abstraction (e.g. drinking-water) and/or for recreation, and limed lakes and watercourses, as well as areas of national interest should be monitored.

INDICATORS OF ENVIRONMENTAL QUALITY AND SAMPLING FREQUENCY

The overall objective for the Nation's freshwaters places two aspects in focus, namely, (i) the biodiversity of freshwater systems and (ii) water as a resource. Consequently, the freshwater environmental monitoring program should use variables that are representative indicators of these two resource goals, and the sampling design (station network, etc.) should be selected to meet these goals. Furthermore, other than freshwater as an environmental resource, monitoring of lakes and watercourses should reflect changes occurring in the terrestrial systems, as well as provide estimates of the potential impact of pollutant discharges to marine systems. The discussion here is primarily limited to indicators and the number of sampling sites needed for monitoring the biodiversity of lakes and watercourses.

BIODIVERSITY

To meet the monitoring goals of sustained biodiversity, a number of ecological indicator variables are needed to assess a system's integrity. On the basis of knowledge gained through previous Swedish monitoring programs and programs of other countries, a number of indicators were pro-

	Programme(1)	Number of sites lakes	watercourses	Sampling frequency
Water chemistry	survey	1000	1000	once every 5 year
	temporal	100	100	4 - 12x per year
	trophic-level	15	15	8 - 24x per year
Sediment chemistry	temporal	100	-	once every 10 year
Phytoplankton	temporal	100	-	1x per year (August)
	trophic-level	15	-	6x per year
Periphyton	trophic-level	-	15	6x per year
Macrophytes	trophic-level	15	15	1x per year
Zooplankton	trophic-level	15	-	4x per year
Macroinvertebrates(2)	survey	1000	1000	once every 5 year
	temporal	100	50	1x per year
	trophic-level	15	15	1x per year
Fish	temporal	100	50	once every 3 year
	trophic-level	15	15	1x per year
Paleoindicators	temporal	100	-	every 3-10 year
Paleoreconstruction	trophic-level	15	-	once

(1) programme types: survey = national lake and watercourse survey (probability sample), temporal = temporal reference sites, and trophic-level = integrated temporal reference sites

(2) macroinvetebrates are collected from littoral and profundal habitats of temporal reference lakes, and littoral, sublittoral and profundal of integrated (trophic-level) temporal reference lakes; riffle habitats are sampled in watercours

Table 2: Overview of selected indicator metrics for the core subprogramme of the national lake and watercourse monitoring programme.

posed to assess *biodiversity at the ecosystem level* (Table 2). A number of additional indicators were considered, e.g. bioassays indicating water/sediment toxicity, biomarkers, and toxic substances in fish, but were given lower priority due to economical and/or practical constraints. In addition, statistics on land use, pollution loads and other supplementary data and information are needed for the analysis and synthesis of freshwater monitoring data.

Assessment of *biodiversity at the species and genetic levels* was also considered in the revised freshwater programme. Some 50 vertebrate and 100 plant (not including algae) species connected to the freshwater environment are considered endangered or rare in Sweden. The varied ecology and environmental requirements of these species complicates implementation of a standardised monitoring programme. It was suggested that the current registration of rare and endangered species be continued, but that a revision of methods should be considered. Similarly, it was considered premature to include genetic methods as direct indicators of anthropogenic stress, given the relatively poor understanding of the effects of natural variability on allele frequency and the practical difficulties of establishing a genetic finger-print. However, this does not preclude the use of genetic methods in biomonitoring of genetically interesting populations such as populations of rare or endangered species.

USE RELATED MONITORING

The variables that have been suggested above as indicators for biodiversity also may be used in a general assessment of the usability of water as a *drinking-water* resource, but may need to be complemented by the assessment of toxic and nuisance algae in certain waterbodies. In addition, monitoring data from reservoirs used for water abstraction should be included in the national assessment of surface water quality. Many of these systems have been monitored over relatively long time periods by water industries.

As a complement to the monitoring of biodiversity, statistics on the commercial fishery should be included in the national environmental monitoring programme as this will reflect major fluctuations in population densities and may be indicative of environmental change. Sport fisheries is probably more important than commercial fisheries in terms of number of people and waterbodies involved. The need for environmental monitoring must in this case be met by the monitoring of general ecology and biodiversity in particular. As regards the function of lakes and watercourses as sites for reproduction of migratory fish, statistics on fish routinely collected for hatchery use at the major electric power stations should be used.

The criteria that decide the usability of surface waters for *recreation* varies considerably depending on the use of the system. The indicators listed above should suffice for most purposes. In addition, faecal bacteria should be monitored in waterbodies used for swimming, and included in national reports of surface water quality.

TERRESTRIAL AND MARINE ECOSYSTEMS

The ongoing river monitoring subprogramme was considered sufficient for the purpose of calculating nutrient and other substances loadings to *marine* areas surrounding Sweden. Hence, no further suggestions were given. However, it was noted that future international agreements may require additional variables and other changes to be included in the subprogramme.

TOXIC SUBSTANCES

Currently, monitoring is developed to follow trends of trace metals in lakes and watercourses and certain organic pollutants (PCB, DDT, HCH and HCB) in fish from a number of lakes. In addition, biological material is preserved for future analyses.

SITE PLACEMENT AND NUMBER

METHODS

The ability to detect change is dependent primarily on variability and the magnitude of effect to be detected. In deciding site placement and sample sizes of the revised national monitoring programme it was therefore considered necessary to evaluate the variability of indicator metrics across varying spatial and temporal scales. Estimates of the *spatial* variability were obtained using data from the 1990 national lake survey, where some 4000 randomly selected lakes were sampled in the spring. Lakes were post-stratified on spatial scales that were considered important: ecosystem size (surface area) and, secondly, geographic position (latitude, areas under pollution-stress). Estimates of *temporal* variability of selected indicator metrics were obtained using time-series data from reference lakes sampled over a 10 year period.

Using estimates of post-stratified indicator metric variability and assuming a "resampling of sites" design, the minimum number of lake and watercourse sites needed to obtain statistical robust estimates of change for the various strata was calculated according to Sokal and Rohlf (1980) as:

$$n \geq \left(\frac{\sigma}{\delta}\right)^2 \{t_{\alpha[\nu]} + t_{2(1-P)[\nu]}\}^2$$

where n = sample size, $t_{\alpha[\nu]}$ and $t_{2(1-P)[\nu]}$ are the t statistics that represent α (t_α) and β (t_β) error probabilities, σ^2 = among-year error variance of indicator metrics, ν = the degree of freedom of the sample standard deviation, and δ = the desired delta (effect size) to be detected. Here α and β errors were set at 0.05 and deltas of 5% and 10% were assumed.

SELECTION OF SITES FOR THE NATIONAL LAKE AND WATERCOURSE SURVEYS

Ecosystem size stratification
The variability of a number of metrics was higher in small compared with large lakes. For example, analysis of indicator metrics showed that small lakes (0.01 - 0.1 km²) had lower pH (5.89) and higher variability (CV = 14%) than large lakes (lakes of the size class 10-100 km² had an average pH of 6.74 and CVs of 6.4 %). This is not too surprising, as indeed both lake and stream characteristics are often dependent on size (e.g. ratio of site to catchment size) and soil properties in the catchment area. An overall analysis of indicator metric variability across varying ecosystem-size scales showed that a number of indicator metrics had CVs ≤ 50%. Assuming that deltas of 5 - 10% will need to be detected, and that the probability of type I (α) and type II (β) errors should not exceed 5%, some 200 to 800 sites were estimated to be needed for each size class used in the national lake survey. Stratifying the Nation's lake population by ecosystem-size alone thus would result in from 800 (δ = 10%) to 3200 (δ = 5%) sites needed in synoptic surveys. Similar data were not available for indicator metric variability of watercourses. However, assuming that indicator metric variability is similar to that in lakes results in the same number of sites needed for national watercourse surveys.

Geographic stratification
Latitude- In Sweden the "limes norrlandicus" (~60° N latitude line) is an important biological ecotone, marking the transition zone between the upper range of deciduous (e.g. oak) forests and the northern predominance of coniferous forests. Coefficients of variation for the selected indicator metrics were generally lower in lakes situated north (CVs usually ≤ 25%) compared with south (CVs ~50%) of limes norrlandicus. The higher variability of indicator metrics in the southern part of the country is, however, confounded by pollution-stress, and may not be due to differences of land types and climate between regions. The higher variability of indicator metrics in the southern part of the country results in from 200 (δ = 10%) to about 800 (δ = 5%) sites (lakes or watercourses) needed, compared with less than 100 sites needed in the north.

Areas with known anthropogenic-stress – Acidification of freshwater resources from airborne pollutants is a serious environmental problem in Sweden. In particular, ecosystems situated in the southwest sections of the country are exposed to higher airborne pollutant concentrations (acid deposition) and comparatively higher precipitation. In southern Sweden it is estimated that wet sulphur deposition is about 10x greater today than during preindustrial times (Bernes, 1991). Analysis of indicator metric variability showed that metrics for acidification-stress often had lower CVs in areas strongly impacted by acidic precipitation. For example, in catchment areas where S and N deposition average > 12 kg ha^{-1} yr^{-1}, acidification indicators had CVs of 13% (pH), 47% (alkalinity), and 25% (SO_4), compared with CV values of 12%, 74%, and 48%, respectively, for lakes situated across the country. These indicator CVs result in between 200 (δ = 10%) and 800 (δ = 5%) sites that are needed to monitor change in areas highly impacted by acidification.

Similar to acidification-stress, southern parts of the country experience generally more stress from nutrient enrichment than areas in the north. However, large regional differences exist in the use of nitrogen fertilisers in agricultural areas. Indicator metrics for eutrophication-stress also had lower CVs in areas with intense agriculture. In areas where agricultural land usage is most extensive eutrophication indicators: Σ NO_2-N + NO_3-N, TN, and TP had CVs of 15%, 9%, 15%, respectively, compared with 28%, 12%, and 27% for lakes across the country. Indicators with CVs of ~30% would result in a minimum of 70 (δ = 10%) to ~300 (δ = 5%) sites that are needed to detect change in areas with problems of eutrophication.

Areas with unknown/unidentified pollutant stress – To account for as yet unknown or unidentified environmental stressors, a number of sites should be situated close to urban areas, i.e. stratification by per capita. The greater majority of inhabitants in Sweden are situated in the south. Hence this form of stratification would de-emphasise the sparsely populated areas of the north. As much of the previous discussion of problem areas occurs south of limes norrlandicus, the discussions regarding indicator variability and the minimum number of sites needed in national surveys applies here.

Final selection of sites

The above theoretical sample size estimates for national surveys indicate that stratification of sites by ecosystem-size alone resulted in some 800 to 3200 sites that are needed to detect deltas of 10 and 5%, respectively, with α and β errors of 5%. Other strata may need additional sites to complement ecosystem-size stratification. Stratification by latitude indicated that detection of change between N and S regions would not require additional sites, whereas stratification by problem areas might need larger sample sizes. For example, to detect changes in regions strongly affected by acidification and nutrient enrichment a minimum of 200 to 800 (acidification) and 70 to ~300 (eutrophication) sites are needed for deltas of 10% and 5%, respectively.

Practical constraints resulted in a number of modifications compared to the above theoretical considerations when the 1995 national lake survey was performed in cooperation with the other Nordic countries (Henriksen et al. 1996). Lakes were selected at random from the national lake register with the assumptions that: (i) a minimum of 1% of the lakes within any country should be included, and (ii) the proportion of lakes in size classes 0.04-0.1, 0.1-1, 1-10,10-100 km^2 should be 1-1-4-8; all lakes > 100 km^2 should be included. The greater inclusion probability was given to large size classes to prevent over-sampling of the numerous small lakes. The number of lakes from different counties was selected so as to achieve a larger proportion in areas with a high degree of acidification or critical load exceedence, few lakes and/or more variable lake chemistry (as estimated from the previous survey, see above). This resulted in a total of 3075 lakes or 3.2 % of the total lake population > 4 ha. All lakes were sampled for physico-chemical constituents and subsets were analyzed for trace metals (~1000) and littoral macroinvertebrates (~700).

The 1995 national survey also included for the first time a number of watercourses. Some 700 watercourses were sampled for physico-chemical constituents and riffle macroinvertebrates. Sites were randomly selected from the Swedish watercourse and catchment area registers, and stratified within two catchment area sizes (i.e. 350 sites were selected for catchments of 15 to 50 km2 and 350 sites for catchments of 50 to 250 km2). These size classes were given priority, as focus was placed on assessment of ecosystem biodiversity.

SELECTION OF TEMPORAL REFERENCE SITES

Representatives of regions and ecological gradients

Since the early 1980's, some 190 reference lakes fairly uniformly distributed across the country have been monitored annually to follow the effects of acidification. These reference lakes were subjectively selected, with five major criteria: (i) emphasis was placed on selecting sites that were considered "representative" so that data extrapolations could be done for the regional lake population, (ii) sites should be relatively uniformly distributed across the country, (iii) sites should not be directly affected by point-source discharges, liming activities, or other forms of manipulation such as ditching, water regulation or implantation of fish, (iv) sites should be relatively easily accessible, and lastly (v) priority should be given to lakes for which monitoring data already exist.

Before reference lakes of the national monitoring program could be proposed as study objects for the revised programme, it was necessary to determine whether the existing lakes could be regarded as regionally representative reference sites, and whether ecologically important environmental gradients were well represented. Predictive modelling by discriminant function analysis showed that the greater majority of reference lakes may be considered as regionally representative. In the alpine/subalpine and northern boreal and southern boreal zones > 82% were classified correctly. In the transitional region between these two zones (i.e. in the middle boreal zone), a greater number of sites were misclassified (38%) as occurring in either of the adjacent ecoregions. However, the highest percentage of misclassifications occurred in the southern parts of the country, south of limes norrlandicus. This is not too surprising, as these regions are relatively diverse in topography and land use. Moreover, as reference lakes originally had been selected so as not to be directly influenced by man, many forest lakes had been included and sites in agricultural areas were under-represented.

Ordination of temporal reference lakes with an unbiased lake population by principle components analysis was used to examine if ecologically important environmental gradients were adequately represented by reference sites. Although reference lakes were fairly well distributed with the randomly sampled lakes in the alpine/subalpine and northern boreal and the middle boreal zones, ordination showed that small, brownwater lakes were under-represented. This bias was most apparent for lakes situated in the southern boreal zone. Furthermore, similar to the relatively high number of misclassifications that occurred with the predictive modelling of lake types by discriminant function analysis, ordination of reference lakes showed an under-representation of clearwater, eutrophic lakes of the boreo-nemoral and nemoral zones.

Number of sites

The number of temporal reference sites needed for estimates of change between years (assuming $\delta = 10\%$) varied from 10 for pH as an indicator metric to ~500 for alkalinity, considering the variability of indicator metrics within size classes (as above). Indicator metrics such as TP, TN and water colour required between 30 and about 200 sites. Moreover, assuming that linear regression will be used for trend analysis, estimates of the number of years that are needed to have a statistical power of 0.95 and a type I error of 5% was solved for using the simplified algorithm of Gerrodette (1987). For individual sites, power estimates indicate that generally a minimum of 20 years are needed to detect changes of 2%, and about 10 years are needed for deltas of 5%. Similarly, it is estimated that at least 20 years are needed to detect significant trends for a number of macroinvertebrate metrics commonly used in biomonitoring.

Final selection of lakes

For economic reasons, only 100 temporal reference lakes were finally included in the national monitoring programme. These lakes were selected among those found to be regionally representative. Where ordination had showed that ecological gradients were not adequately covered, new sites were selected with the assistance of regional monitoring boards. As regional programmes generally also include a number of temporal reference sites, it is expected that altogether some 200 sites will be sampled annually, and these data will be used in national estimates of trends in surface water quality.

Watercourses

In the revised watercourse subprogramme, it was assumed that the number of sites needed to monitor ecosystem change was similar to that calculated for lakes. However, for economic reasons only 100 sites could be included in the national monitoring programme. Since the early 1960's, some 120 watercourses have been monitored annually using physico-chemical metrics. Fifty of these were included in the revised subprogramme, with the objective being to monitor river transport to coastal regions. The selection of the fifty remaining sites is under consideration. Here subprogramme objectives emphasise monitoring of biodiversity and ecosystem integrity. Hence sites that are considered to be regionally representative will be primarily situated in small to medium size catchment areas.

SELECTION OF INTEGRATED TEMPORAL REFERENCE SITES

Included in the temporal reference subprogramme are sites where interactions between physico-chemical and biological processes within and between sites and their watersheds are studied. Fifteen lakes were selected as being representative of major ecological gradients and eco-regions, based on an analysis of existing integrated temporal reference lakes (Persson 1996). The selection of integrated watercourse sites is under consideration.

AREAS OF NATIONAL INTEREST

Some 380 aquatic sites have been designated as being of national interest to nature conservation and outdoor life. Given the wide diversity of habitat types, it is difficult to design simple and uniform monitoring programmes. However, a basic programme similar to that conducted in the national surveys was suggested, and may be complemented with studies of site-specific characteristics associated with their status as areas of national interest.

SELECTION OF LIMED LAKES AND WATERCOURSES

The national monitoring subprogramme of limed lakes and watercourses is similar to the integrated temporal reference sites subprogramme regarding site selection (ecoregion representatives), choice of indicator metrics and sampling frequency. The subprogramme consists of 14 lakes and 18 watercourses.

SELECTION OF USE RELATED SITES

No new sites were included in the revised monitoring subprogramme. However, statistics from existing sites (drinking-water reservoirs and swimming areas) will be included in the national assessment of water quality (as mentioned above).

SAMPLING AND ANALYSES

Regional-scale sampling is used for most of the national freshwater monitoring programme. Most chemical and biological analyses are performed at the Department of Environmental Assessment (DEA) at the Swedish University of Agricultural Sciences. Standardized methodologies are used for analyses, such as SIS, ISO or CEN standards. Standard taxonomic keys and nomenclature are used for biological analyses. Minor programme constituents are run by other contractors, most notably those involving fisheries, which are performed by the Freshwater Laboratory of the National Board of Fisheries.

ASSESSMENT AND REPORTING

A media-responsibility system has been adopted for the handling of national and some of the regional monitoring data, with DEA being responsible for most of the data from lakes and watercourses. Data are distributed to SEPA and regional authorities on a yearly basis and submitted free of charge to anyone requesting the data. The data base is currently being made available over Internet (http://www.ma.slu.se).

Annual assessments of status and trends will be published by SEPA. They will primarily be based on the national programme, but also include results from regional monitoring. The major findings will be included in SEPA's annual "State of the environment" report . Results from the

national monitoring programme will also form the basis for official statistics published regularly by Statistics Sweden (SCB). Furthermore, SEPA and SCB are developing an environmental quality index or profile for freshwaters, that will largely be based on results from the national monitoring programme.
Results from monitoring activities have been used in SEPA's action programmes on freshwaters and the marine environment, and are expected to be increasingly important in the setting of environmental objectives and priorities on the national level. Regionally, monitoring results have formed a basis for county-wise environmental assessment reports requested by the Government from the counties.
Another area where monitoring results are being used is that of international agreements and conventions. Results are reported to and/or form the basis of environmental assessments within the framework of e.g. HELCOM and OSPARCOM (river transports of pollutants to marine areas) as well as the UN ECE Convention on long-range transport of air pollutants (critical load and S and N protocols; cooperation on studies of the effects of acidification on lakes and integrated monitoring).

Finally, a number of contributions to the scientific literature have appeared throughout the years as a result of the monitoring activities and are expected to continue to do so, dealing with various methodological aspects as well as the ecology of, and man's contribution to, environmental change.

QUALITY ASSURANCE

Procedures for quality assurance include: standardised field and laboratory protocols, training of field and laboratory personnel, and regular participation in interlaboratory comparisons. SEPA is currently distributing and updating instructions for the various aspects of the environmental monitoring ("Handbook for environmental monitoring"). The handbook will contain detailed plans for quality assurance on the national level, and instructions for plans on the regional level.

IMPLEMENTATION

Future monitoring along the lines that have been described above means that much of previous and present activities continue, but also that stations and routines have to be abandoned and new ones established instead. Whereas the programme has been accepted in principle by SEPA's monitoring board, the final degree of implementation is under consideration by SEPA.

At the national level, the programme is largely running as described above, with the exception of periphyton, macrophyte, and paleoindicator studies that have not started, and all temporal reference and integrated temporal reference watercourses have not been selected.

At the regional level, catchment-wise and liming-effect follow up monitoring continue as before. In addition, regional monitoring programmes are being implemented with a design similar to that of the national subprogrammes (i.e. nested sampling effort, with large-scale survey's done every 5-yr and a selected number of temporal reference lake and watercourse sites). Nearly all stations that were removed from the previous national programme have been included at regional levels, including Sweden's four largest lakes. Also, special projects are devoted towards monitoring habitat types or species that are regionally characteristic (e.g. small and ephemeral waterbodies, glacial relicts, etc).

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RIVER FLUX MONITORING OPTIMISATION: TWO CASE STUDIES ON THE SEINE AND RHINE RIVERS.

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ABSTRACT

Flux survey strategies are tested on sets of daily concentrations measured and/or reconstituted in the Seine and the Rhine rivers. Tested variables include TSS, Cl-, SO4-2, PO4-3, NO3- and particulate Pb. Biweekly to monthly frequencies are tested and the related flux compared to the true value according to the method of Walling and Webb (1982, 1985). The time base of the flux estimates range from 3 months - a high water stage on the Seine- to 20 years of TSS record on the Rhine. Optimal frequency to reach 5% precision ranged from biweekly to monthly depending on the target variable and period of estimate. An unusual anthropogenic weekly Cl- cycle in the Rhine is responsible for major bias at this frequency.

INTRODUCTION

The assessment of riverine fluxes is one of the key objectives of water quality surveys (Chapman, 1992) although it still receives limited consideration in many networks. Flux stations are generally established upstream of lakes and reservoirs, at interprovincial or international boundaries, and at river mouths upstream of the dynamic tidal influence to assess pollutant inputs to the ocean (GESAMP, 1987; Walling, 1987). Such fluxes may involve many substances ranging from total suspended solids (TSS) to nutrients and micropollutants, in both dissolved and particulate forms. A few substances may also be measured on unfiltered waters (e.g. total P, total organic carbon -TOC, or Kjeldhal N -NK), although these concentrations are generally closely linked to TSS. The required accuracy of flux determination greatly depends on the objectives of water quality surveys. For dissolved substances it is not realistic to aim at accuracies much better than 10%, i. e. the water discharge accuracy and analytical precision combined. For particulate pollutants the accuracy can not be better than the one performed on a single TSS measurement at the sampling station which is limited by the water column heterogeneity, i. e. about 10 to 20% depending on sample procedure. Other problems in the establishment of fluxes include sample contamination and preservation, sampling frequency and the flux computation method (Meybeck et al., 1993). It must not be forgotten that flux estimates mainly depend on water quality and water discharge precisions at high flow period. For particulate matter and associated pollutants it is very often found that more than 50% of fluxes are generated during floods, i. e. less than 10% of the elapsed time. Only sampling frequency and flux computation method are addressed here. They have also received particular attention from Walling and Webb (1981, 1982, 1985) and many other scientists (Hubert *et al.*, 1988).

BASIC DATA

Two rivers have been investigated: the Seine at its mouth (Poses station, 60000 km2, 430 m3/s) and the Rhine at Lauterbourg (49947 km2, 1270 m3/s) and at Maxau (50196 km2, 1280 m3/s), on either side of the German-French border downstream of Strasbourg.
Several sets of daily measurements have been used:

- Rhine (Maxau): 20 years of TSS available at the Koblenz Bundesandsalt für Gewässerkunde.
- Rhine (Lauterbourg): 20 years of Cl- available at the Agence de l'Eau Rhin-Meuse, Metz. Only one year is used here (from December 28th, 1992 to December 30th, 1993).
- Seine (Poses): 2 hydrological years of TSS available at the Service de la Navigation de la Seine (Rouen), from September 1983 to August 1985.
- Seine (Poses): 90 days of daily TSS, SO4-2, PO4-3, NO3-, and particulate lead (pPb) for a major flood from December 27th, 1993 to April 1994.

The 1993/94 Seine flood magnitude (maximum Q at 1900 m3/s) has only been observed 5 times between 1959 and 1994. The daily concentrations were not all measured: only 35 samples were taken during 90 days especially on the rising stage when TSS variations are rapid. Missing concentrations have been interpolated on the basis of the C vs Q relationships established during a 1990/92 pilot study at the same station (Cossa et al. 1994, Meybeck and Idlafkih 1995). Particulate lead (pPb) was analysed on Nuclepore filters by Graphite Furnace Atomic Absorption at the Ifremer-Nantes Laboratory. Analytical precision ranges from 3 to 6%.

METHODS

Various sampling frequencies ranging from biweekly to monthly sampling scenarios have been simulated using these daily data sets for periods extending from 3 months (the major Seine flood) to 20 years (TSS on the Rhine). In this work we have only considered the commonly used regular sampling strategies. Stratified sampling, e.g. weekly during the high water stage and monthly during summer has not been considered.
Two computation methods have been used to estimate dissolved fluxes as previously documented by Walling and Webb (1985).

$F_2 = \Sigma\ (Ci\ Qi)\ /\ n$ (Average of instantaneous fluxes)
$F_4 = Qm\ (\Sigma(Ci(Qi)/\Sigma Qi)$ (Discharge weighted concentration x annual discharge)

Where F_2, and F_4 are the flux estimates, n: number of samples taken, Ci: concentration at sampling date, Qi: water discharge at sampling dates, and Qm: average water discharge during the whole period of the flux estimate.
Simulated fluxes have been compared with the true fluxes calculated as the average of measured of reconstituted daily fluxes.

FLUX SURVEY STRATEGY DURING A MAJOR SEINE FLOOD

In the regular survey, started in 1971 (Reseau National de Bassin), the water discharge on the sampling dates never exceeded 1000 m^3/s. In 1993/94 the flood exceeded this level for two months (Fig 1) and the peak discharge reached 1900 m^3/s. The reconstituted daily fluxes for TSS, NO_3^-, PO_4^{-3}, SO_4^{-2} and pPb, have been established using both measured and extrapolated concentrations (Fig 1 and 2).
While TSS (Fig. 2a) and pPb (Fig. 2c) fluxes are much more variable than water discharge (Fig 1a),

the NO_3^- (Fig 2d) and SO_4^{-2} (Fig. 1b) flux patterns are very similar to that of discharge. The PO_4^{-3}, flux (Fig. 2b) is variable but does not show a marked increase during the flood. This is confirmed by the ranges and maximum / minimum ratio of specific fluxes (Table 1).

	Q (m3/s)	TSS (kg/s)	SO4-2 (kg/s)	P-PO4-3 (kg/s)	N-NO3- (kg/s)	pPb (kg/s)
Maximum	1956	562	57	0.35	11	0.067
Minimum	333	6.66	18	0.04	1.48	0.0016
Max./Min.	5.87	84.4	3	9	7	40.3

Table 1: Flux variability during the 3 month flood event in the Seine at Poses (1993/94), for TSS, SO_4^{-2}, P-PO_4^{-3}, N-NO_3^- and pPb.

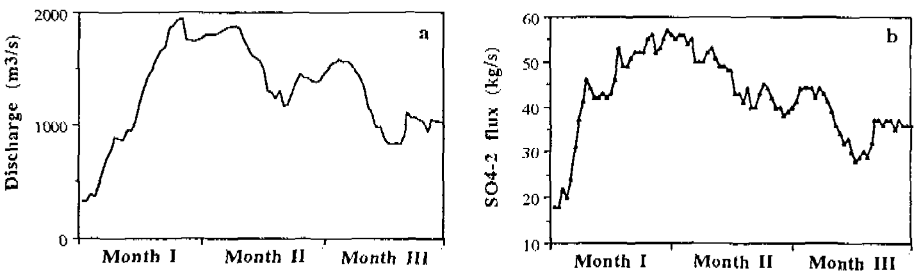

Figure 1: Major flood in the Seine river at Poses from December 6th, 1993 to March 5th, 1994. (a) water discharge, (b) reconstructed daily SO_4^{-2} flux

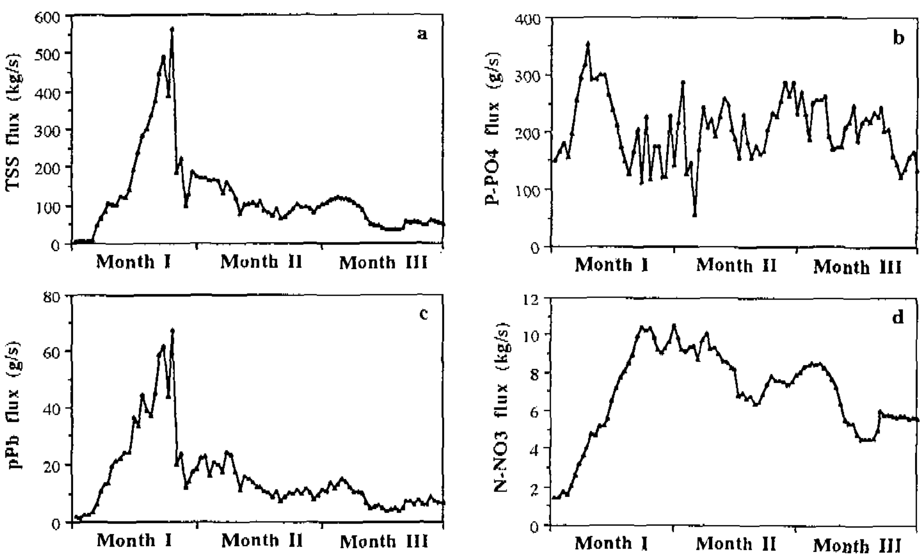

Figure 2: Major flood in the Seine river at Poses from December 6th, 1993 to March 5th, 1994. Reconstructed daily flux: (a) TSS, (b) P-PO_4^{-3}, (c) pPb, and (d) N-NO_3^-

These different patterns can directly be explained by the various C vs Q relationships that can be represented as C = a Qb:
-1< b < 0: dilution pattern, as for PO_4^{-3} (b = -0.5) originating from Paris sewage treatment plant. The flux variability is much less than that of water discharge.
b ≈ 0: as for SO_4^{-2}, both elemental flux and water discharge variability are similar. However here the SO_4^{-2} flux range is slightly less than that of Q since the b coefficient is slightly negative (-0.14).
0 < b < +1: as for NO_3- (b= 0.17) originating mainly from diffuse agricultural sources. The elemental flux variations slightly exceed those of discharge.
1 < b as for TSS during the rising stage of floods (b = 1.04). The fluxes increase much faster than water discharge.

Frequency		**12/year**	**24/year**	**52/year**	**104/year**
TSS	median	-35	3.4	-1.3	0.5
	maximum	74	28	21	3.2
SO_4^{-2}	median	-0.7	-0/09	0.08	-0.25
	maximum	9	3.6	0.8	1
N-NO_3-	median	-1.5	-0.09	-0.03	-0.08
	maximum	9	4	1.3	1.2
P-PO_4^{-3}	median	-3.2	-16	0.8	-0.7
	maximum	43	20	11	2

(f4 computation method).

Table 2: Sampling strategies and simulated material fluxes during the three month flood event. Median and maximum errors in % observed on replicate data sets for various sampling frequencies.

For the particulate matter a marked hysteresis in the C vs Q relation is noted with a much higher TSS at the rising stage than on the receding stage as commonly observed in lowland basins (Walling and Webb, 1981)-As a result the period of significant TSS transport is limited to two weeks as compared to three months for the water discharge (Fig. 2a and 1a).
The very similar patterns of pPb and TSS fluxes indicate that the Pb content of sediment is much less variable - here from 100 to 160 µg/g - than the TSS levels, from 20 to 300 mg/L. If pPb was mainly coming from point sources, a dilution pattern of Pb content with TSS would have been observed. Actually this is not the case and it is believed that most of the river particulates transported at the Poses station were stored in the river bed prior to the flood, the first of the hydrological year.
The variability of PO_4^{-3} flux does not fully reflect reality: during the second month of the flood the few measured PO_4^{-3} concentrations were lower than the estimated concentrations for these water discharges based on previous surveys and the reconstruction of daily P may therefore be *unreliable, although still used here in simulations of sampling strategies.*
Four types of sampling strategies have been simulated for the Seine flood event ranging from biweekly to monthly frequencies (e. g. every monday, tuesday, etc. for weekly samples, every 15th, 16th, etc. for monthly samples) (Fig. 3).
For instance 15 replicate sets of 6 samples were assembled for the bimonthly simulation. Corresponding fluxes were computed and compared to the reconstructed ("actual") daily fluxes for the whole period presented on Figures 1 and 2.
If a monthly sampling requency is selected (Fig. 3a) the median of the replicate TSS flux estimates is slightly biased, around 5% less than the actual flux. For 1/3 of the simulated samples sets the resulting fluxes are underestimated by 20% and for 1/4 of simulations the fluxes are overestimated by 20% to 100%. A weekly TSS sampling programme (Fig. 3c) is satisfactory: there is a small bias (-3.2%) in the median of the replicate fluxes and the error is less than 10% for 5 simulations out of 7, with maximum errors reaching 20% for the 2 others.

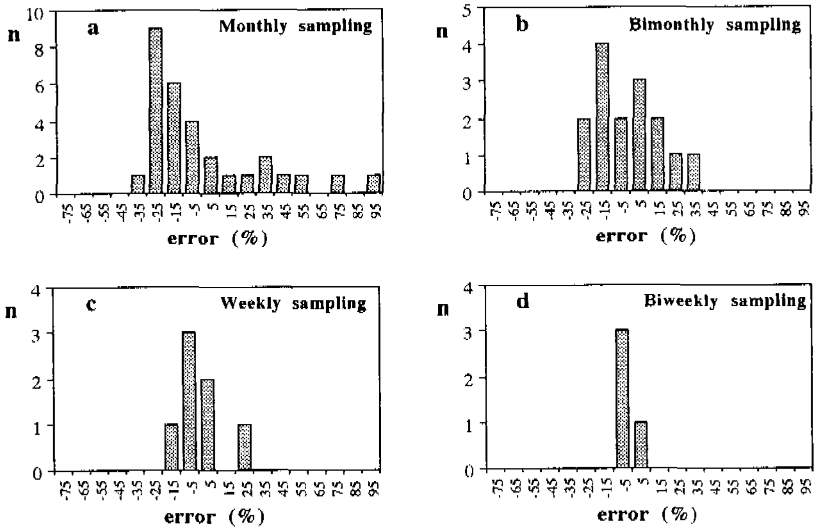

Figure 3: Sampling strategies and TSS fluxes on the Seine river (1993/94 flood). Distribution of errors in % of actual fluxes on drawn data sets simulating various sampling strategies, with 30 sets of 12 samples (3a), 15 sets of 24 samples (3b) 7 sets of 52 samples (3c) and 4 sets of 91 samples (3d).

All other variables tested need less frequent sampling to reach the 5% level of precision, bimonthly sampling for SO_4^{-2} and NO_3^-, biweekly for PO_4^{-3}. As noted above the results for PO_4^{-3} may be affected by the poor reconstruction of daily concentrations for the second month, and weekly sampling is more likely to be needed. Since TSS concentration in rivers are themselves generally not known with a precision better than 10% - specific vertical TSS profiles coupled with flow velocity measurements are generally lacking - this level of precision is the maximum possible objectve for most TSS flux surveys.

LONG TERM TSS SURVEYS IN THE SEINE AND RHINE

Three years of daily records (1983 to 1985) are available on the Seine and 20 years on the Rhine at Maxau. Both rivers are not very turbid and maximum TSS values barely reach 200 mg/L. For the Seine the average basin relief is very low, thus limiting erosion and sediment transport, while for the Rhine the low TSS levels result from the storage of alpine-derived material in subalpine lakes such as Konstanz, Zurich and Lucerne.

Sampling strategies of 12 samples/years to 52 samples/years were simulated and tested for flux computation (Table 3). For these computation the hydrological year starting September 1st has been used for both rivers. Due to their water regimes it is common that major floods are split between December and January, so that the bulk of the particulate load may be transported in one civil year while the bulk of the water runoff may be in the next one. (if medium term trends for periods exceeding 3 years are considered the distinction between civil and hydrological year is unimportant).

Frequency		Weekly	Bimonthly	Monthly
Median of error (%)	(1)			
Rhine (2)	F_2	-1.5	-5.4	-6.2
	F_4	-1.1	-3.5	-5.6
Seine (3)	F_2	1.7	0.04	-0.13
	F_4	1.7	0.58	-01.5
Maximum error (%)	(1)			
Rhine (2)	F_2	22	90	200
	F_4	22	80	170
Seine (3)	F_2	4	8	18
	F_4	4	6	16

Table 3: Sampling strategies and simulated river TSS fluxes on the Seine (2 years) and the Rhine (20 years). Median and maximum errors in % observed on replicate data sets for various sampling frequencies.

On the Seine river two consecutive hydrological years (one dry, one wet) have been tested. With a monthly sampling strategy the maximum errors observed in the two annual fluxes range from -40% to +50% and, for the two year combination, from -18% to +18%. With a bimonthly sampling strategy the errors associated with the simulated fluxes are between -20% to +30% for the wet year but drop between -15% and +10% for the dry year. When the two years are combined the maximum error is less than 10% for the bimonthly sampling. The bias, as expressed by the median error associated with the replicate flux estimates (i.e. for 30 flux estimates for the monthly frequency, and 7 flux estimates for the weekly frequency) is near zero for the two-year period (Table 3).
On the Rhine the error distribution can be defined for each of the 20 years tested (from 1973 to 1993). For a monthly sampling frequency the maximum errors recorded for one year range from 40% (dry years) to 200% (wet years). These figures drop to 4% and 22% for weekly sampling, i. e. for the 20 (7 weekly sampling replicates the observed fluxes were always between 0.78 and 1.22 times the actual values (Table 3). The 20 years median error is here determined from the population of the 20 median errors observed for each year (Table 3). Again the bias is limited: the underestimation reaches 6% for monthly sampling.

FLUX COMPUTATION METHODS

The two methods F_2 and F_4 documented by Walling and Webb (1982, 1985) are nearly equivalent (Table 3). Only in very few cases, when the average discharge for the sampling date differs significantly (more than 10%) from the average discharge over the survey period, is the correction introduced by method F_4 necessary.

WEEKLY BIAS ON RHINE CHLORIDE FLUXES

Daily chloride measurements over one year were considered at Lauterbourg. Here chloride levels are very high, circa 200 mg/L compared to less than 15 mg/L at the upstream station near Basel, this increase reflects the Alsace potash mine (MDPA) brines that are directly released in the Rhine. Sampling frequency scenarios ranging from biweekly to monthly sampling have been tested (Table 4).

Sampling frequency	Maximum error of annual flux (%)	median error (%)
Monthly	20	+2.7
Bimonthly	12	-1.6
Weekly	31	+0.23
Biweekly	0.8	-0.26

(F2 computation method).

Table 4: Sampling strategies and simulated river C1-fluxes on the Rhine at Lauterbourg (over one year). Median and maximum errors observed on replicate data sets for various sampling frequencies.

The most striking feature is that the maximum error observed for weekly sampling is higher than that for monthly sampling! This is related to a marked weekly cycle of the Cl- concentrations, with a Cl- range from 80 to 200 mg/L being caused by the release pattern of NaCl brine. As a result the maximum flux is generally observed on Saturdays (120 kgCl-/s) and the minimum on Wednesdays (60 kgCl-/s). For this rare pattern of variability any regular survey strategy undertaken on fixed week days would lead to a major systematic error in fluxes even for a high sampling frequency (i. e. weekly). A monthly sampling at various week days would be quite sufficient for a long term estimate since the median error is only 2.7%, provided that different week days are chosen.

CONCLUSIONS

Sampling strategies for flux estimates should be carefully designed to take into account the variation of the target pollutant with river discharge. TSS and associated pollutants such as particulate nutrients (N, P and C) and metals (Cd, Cu, Hg, Pb, Zn) and poorly soluble organic micropollutants (PCBS) should be more frequently surveyed than most dissolved elements. However the pollutant content per gramme of suspended matter is generally much less variable (from 50% to 500%) than the amount of TSS which commonly ranges over one or two orders of magnitude. Therefore a combination of frequent TSS measurements, particularly during the rising stage of floods and for the first flood events, should be combined with much less frequent TSS analyses. For less turbid medium-sized rivers such as the Seine and the middle Rhine, the optimum frequency for TSS flux surveys providing a 5% precision is bimonthly to monthly for long-term averages and weekly for estimates over one flood or one year. Major ions, phosphate and nitrate require less frequent surveys for the same precision.
In more variable fluvial conditions - smaller basins, mediterranean rivers - sampling should be more frequent. Differences due to the use of different flux computation methods are not significant in the Rhine or the Seine tests.
Stratified sampling, not considered here, could reduce the total number of samples required per year. It should be based on pilot studies during which the concentration vs Q relationships and other variations are established.

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WATER QUALITY MONITORING IN RUSSIA (STATUS, ISSUES, PERSPECTIVES)

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ABSTRACT

Russia has a long history of the development of geophysical monitoring systems, including Water Quality Monitoring System (WQMS). Unfortunately, a positive potential accumulated in Russia and Former Soviet Union (FSU) over many decades has been seriously endangered recently due to dramatic decrease in federal funding and poorly coordinated decentralization. The paper provides a brief description of the history of water quality monitoring in Russia, along with basic principles of the system operation and role of key federal agencies involved in WQM. Current status of existing monitoring of pollution sources and monitoring networks for surface and ground waters are outlined, as well as the most important deficiencies and issues in the Russian WQMS. Analysis of emerging water quality information needs has been made and possible directions of WQMS reconstruction on the federal and regional/basin levels are discussed. Examples of internationally funded projects aimed at WQM improvement are provided.

WQMS HISTORY AND CURRENT ORGANIZATION

First studies of river water quality were conducted in Tsarist Russia at the end of the last century. At the same period hydrological observations of surface and ground water bodies were initiated. Before the Second World War concentrations of major ions, pH, temperature, color, transparency and some other basic parameters were regularly measured of a few hundred river sites. After the War, the monitoring network was expanded to 1500-2000 sites located on major rivers, lakes and reservoirs. At that time measurements of nutrients, Fe, Si, O2, COD became a part of a routine monitoring program. Regular monitoring of pollutants such as oil and grease, surfactants, phenols, heavy metals has been under way since the mid 1960s. Organochlorine pesticides have been monitored in surface water bodies for about 20 years. Regular biological monitoring of water ecosystems was introduced in the late 1970s.
The observation of ground waters initially concentrated on aquifers closest to the surface and in the 1950s - 1960s the network was expanded to include deeper confined aquifers as well. Since that time the network had been expanding continuously and in the 1970s there were about 22,000 sites in the FSU. The initial objective of the network was to study the regularities of variations in the natural ground water regime. However, due to a growing anthropogenic impact more and more observations were carried out to collect information on water intakes, irrigated lands, industrial and mining enterprises.

The current water quality monitoring system in Russia is inherited from the Former Soviet Union. It was designed in the mid 1970s when the National System of Observation and Control of Environmental Pollution had been developed (Izrael et.al., 1978). At the end of the 1970s it was one of the largest and best organized national WQMS on the globe. Compared to a similar system in the USA in the 1970s, it was certainly less developed in terms of laboratory capacities, sampling equipment, technical support, but it had some other advantages, for example: almost all monitoring sites were provided with measured or calculated hydrological parameters, routine aquatic biology observations were initiated and national reviews on water quality were annually published. However, during the last 20 years the water quality monitoring system has not changed in inland surface water quality. The samples are sent to laboratories by regular mail and, thus, for 30-35% of the sites shipment of the samples or their portions takes more than 15 days. There are 82 analytical laboratories of 22 Roshydromet Regional Departments, major laboratories can analyze up to 50-60 parameters. Most of the laboratory instruments are outdated both physically and morally. There is a shortage of all types of supplies and materials. Every year, the laboratories analyze standard samples of few individual substances centrally distributed among the laboratories. However, the scope of the work is insufficient and quality control is not conducted at the other stages of data acquisition and processing.

Data obtained in analytical laboratories are recorded and processed at the regional computer centers. Roshydromet Regional Departments prepare, publish, and disseminate water quality information on the territories they monitor. Such information for the whole territory of Russia is prepared by the Hydrochemical Institute (physical and chemical data) and the Institute of Global Climate and Ecology (aquatic biology data). In addition to routine "regime" information, the existing surface water quality monitoring system should provide short-term (operational) information primarily in the form of cables and reports on the "extremely heavy" and "heavy" pollution events in water bodies.
The Annual Report on Surface Water Quality in the Russian Federation is the basic information document used in preparation of all major water management projects. This document includes water quality information for hydrographic areas of Russia and basins of major rivers. Water quality is estimated by analyzing the frequency and value of maximum allowable concentrations (MAC) exceedances for individual substances, as well as the information on the events of heavy and extremely heavy pollution. Attempts to use various modifications of integrated indices are made. Conclusions on improvement or deterioration of water quality are made based on the comparison with previous years. The Annual Report presents available information on the cases and sources of unfavorable water quality and its variations as well as the lists of the most heavily polluted water bodies.
Water quality information collected and presented by different Agencies is generalized by Minprirody in the section "Water Resources, Their Status, Protection, and Use" of the Federal Report on the State of Natural Environment in the Russian Federation.

DEFICIENCIES AND ISSUES OF RUSSIAN WQMS

In addition to disadvantages mentioned above, there are some other deficiencies and problems the most important of which are as follows:

- Lack of clearly formulated goals and objectives of the monitoring system reflecting its close relationship with the environmental management system

- Reliance upon a wrong fundamental concept based on the assumption that all information needs may be addressed with the help of one unified system managed from the center

- Unclear distribution of responsibilities, poor coordination between Federal Agencies and frequent reorganizations

- Outdated monitoring principles based primarily on a fixed-station approach and, thus, limited possibilities for the estimation of the spatial parameters of pollution and detection of emerging issues

- Lack of specific monitoring programs for major river basins and aquifers, as well as monitoring programs for assessment of major pollution problems, especially related to human health risks

- Inadequate attention to the observation of suspended sediments, physical parameters of habitats, concentrations of toxicants in biota, ecotoxicological parameters, etc.

- Outdated instruments and equipment and, as a result, quite limited possibilities to detect toxicants, pesticides, and heavy metals

- Inefficient QA/QC system of monitoring activities and, hence, a low reliability of the results

- Water quality assessment based on an outdated and inflexible system of MACs; lack of modern information technologies and limited possibilities for the dissemination, processing and presentation of information

- Poor accessibility, low reliability and, sometimes, absence of relational data on land and water use, pollution sources, data on population and economic activities, etc.

- Lack of mechanisms for WQMS performance evaluation and introduction of improvements in case of a poor performance.

Similar generic WQMS deficiencies were typical for the FSU and Russia in the late 1980s and early 1990s (Tsirkunov, 1994). During the recent years, some of these deficiencies have become somewhat less important (e.g. points e, i, j) while most of the others have not changed or, may have been aggravated (c, d, g, h) primarily due to a dramatic decrease in monitoring funding. In 1995, at least 10-15% of the sites shown above were not operational. The number of samples taken and parameters determined have also substantially decreased. Last year Federal Agencies have only been able to irregularly finance staff salaries (which are at the lower end among the federal institutions) and some urgent operating costs (power, fuel, etc.). Federal Agencies managed to buy, in recent years, just a few pieces of new equipment to upgrade already outdated laboratories. All laboratories are facing a severe shortage of supplies, chemicals, glassware and other consumables. Regional departments and scientific institutes are usually insolvent and are not able to pay for their power supply, the delivery of samples by mail, use of vehicles and boats, etc. Unfortunately, it is impossible to demonstrate the level of funding decline because WQM cost estimates at the national level are nonexistent. It is quite obvious though that under available funding national WQMS will continue its deterioration.

The new economic and political situation in Russia has created new needs and problems in water quality management and monitoring. One of the problems is related to the tendency for decentralization and devolution of the bulk of responsibilities to the Subjects of Federation (Republics, Oblasts, Krais, etc.). The rights and responsibilities of these regional administrative units in environmental management have been increased as well as public awareness of the need to combine federal, regional and municipal efforts in order to improve environmental quality. Some of the administrative units have already demonstrated their willingness to finance their monitoring systems. Unfortunately, regional environmental authorities usually do not have capacities and resources to develop their own monitoring systems. In most cases, they are trying to run the same federal monitoring programs in their territories and fill what is perceived as the most critical gaps in national funding. For these activities separate agreements with the departments of Federal Agencies (Roshydromet, Roscomnedra, Roscombod, etc.) are negotiated annually.
Due to the poor condition of most industrial and transport facilities, the deterioration of the working discipline, outdated treatment facilities and other factors, the risk of major accidents with dangerous environmental consequences has considerably increased. This requires the development of early warning monitoring systems in the most valuable and vulnerable areas with potentially high accident risks.
The issue which appeared recently after the disintegration of the FSU and formation of the Russian Federation as an independent state is monitoring of transboundary water bodies, especially on the borders with the Ukraine, Baltic States and Kazakhstan. Additional needs in water quality

data are arising in the process of environmental assessment for large international projects in Russia (oil and gas development, other natural resources extraction, etc.). Such data should comply with international QA/QC procedures and therefore require additional monitoring capacity.

PERSPECTIVES FOR THE DEVELOPMENT

Summarizing the foregoing one may say that water quality information needs have never been so large in Russia and the possibilities for obtaining information so limited. How to resolve this contradiction and transform the present monitoring system into a modern integrated system satisfying the basic needs of water resources management is an extremely complicated problem. The issue by itself is quite unique. There are no clear precedents on redesigning and restructuring of a massive outdated and centrally managed system into a more flexible, objective-driven system where federal and regional/basin systems may complement each other. How to implement such transformation under severe budget cuts and in the process of revolutionary economic and political changes is an especially challenging, if at all possible, undertaking,. Under any circumstances such efforts in order to be successful should be well planned and coordinated among major stakeholders. At the same time, it is very unlikely that conflicting Federal Agencies confronted with the shrinking federal funding, duplicating and changing mandates, migration of the best professionals to the private sector and abroad will be able or willing to develop any feasible approach in the near future. It is more likely that the agencies will continue to fight over federal budget reallocation which will lead to a further decline of WQMS. Such attempts may be illustrated by the plans of the Ministry of Environment to develop an ambitious Integrated Federal System of Ecological Monitoring which has to include "systems of monitoring of natural environment, natural resources, natural engineering systems, natural complexes, ecosystems and sources of anthropogenic impact". Such a system can not be sustainable in the present situation and attempts to implement it can only contribute to an overall decline of WQMS. The monitoring system should be reconstructed with great care to prevent a complete disintegration and destruction of the existing network which is the only source of environmental quality information. Well designed and tested modifications can be gradually introduced in the system. The reconstruction could possibly be initiated simultaneously at the federal and regional (basin) levels.

At the **federal level**, activities could be implemented that do not need considerable expenditures, but may have a significant effect. Such measures could comprise the development of necessary legal and organizational prerequisites to improve monitoring systems, development and coordination of the goals and objectives of the monitoring systems based on the management needs, signing of long-term agreements between federal and regional authorities, interagency coordination and information exchange, development of guidelines for the design of regional/basin WQMS, and improvement of a water quality evaluation system. In the mid-term, a joint interagency approach may be developed on how to use very limited resources in the most efficient way. It can include recommendations on the combination of federal, regional and other types of WQMS, optimal use of monitoring types (fixed stations vs. synoptic surveys, etc.), selections of matrices, sites, parameters, frequency of observations and other measures which are absolutely critical for national WQ evaluation.

At the **regional and basin levels**, it is necessary to start developing one or several integrated water quality monitoring systems in the regions or river basins with the most serious water resources and environmental problems where a well-developed institutional and scientific infrastructure, as well as a strong support from regional and local authorities are available. In the process of designing and developing such monitoring systems it is possible to gain certain experience, elaborate interagency cooperation and train personnel.
The pilot project "Prototype Integrated Water Resources Monitoring System" prepared in the framework of the Russian Environmental Management Project, funded by a World Bank Loan, may be used as a basis for the development of such a system. The scope of the project was develo-

developed jointly by the experts of the US Geological Survey and US EPA, World Bank and Russian experts. The project is being implemented in the Lower Don River Basin. Steps in the project design include:

- development of databases for water resources management;
- improvement of the present system of water resources data collection;
- identification of the most serious data gaps and collection of selected data;
- assessment of current water quality conditions;
- identification of management needs; and
- designing of a new monitoring program.

Experience gained in the process of the pilot project implementation will be disseminated among the other regions and river basins of Russia. There are other bilaterally (e.g., US AID) and internationally (e.g. WHO, TACIS) funded projects fully or partially aimed at WQMS improvement. International organizations and bilateral donors may also wish to consider the opportunity to support some key monitoring activities in Russia which are of critical international importance. For example, to support the operation of GEMS/Water stations in the mouths of major Siberian and Far East rivers and baseline stations. The situation in these remote areas is especially alarming. Discontinued observations at these sites which usually have water quality and hydrology records over 40-50 years long will result in essential losses for world science. Another area which may be attractive for donors is the facilitation of database development and use of modern information technologies. The vast amount of already accumulated data can be used for global assessments (geochemistry, global modeling, etc.) for environmental assessments for new economic projects (hydrological records, background water quality data, etc.).

Today, Russia is facing very complicated and intertwined social, economic, and ecological changes. It is difficult to predict perspectives for WQMS development under such circumstances. What is clear, though, is that the Federal government will never be able (as it used to be in the FSU) to afford financing of massive monitoring systems comprising all types of monitoring activities. Such systems in future should meet the needs of various users (Subjects of the Federation, basin and municipal authorities, private sector, etc.) and rely upon various financial sources with a strong technical guidance from the Federal Agencies. In the short and medium term, specially designed elements of a new water quality monitoring system, tested on the pilot basis may be gradually introduced into the existing deteriorating system, following the course of economic improvement.

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REORIENTATION OF WATER QUALITY MONITORING AND ASSESSMENT PROGRAMMES - THE INDIAN STRATEGY

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ABSTRACT

India, a developing country, is one of the most densely populated countries of the world. The few monsoon months (June - Sept.) bring copious rainfall. The water resources are adequate too. However, the resources management, including that of water quality, are still in the development phase. In the last two decades, information requirement in the water quality sector was more confined to pathogenic indicators (arising out of faecal contamination of both point and non point sources by origin), biodegradable organics and related oxygen balance (arising out of point sources only). Therefore, the focus of all water quality programmes were only on these subjects. The Central Pollution Control Board, one of the key players in the water quality sector, developed its programmes to meet these objectives. A programme reorientation in the freshwater quality sector is now on the anvil.

WATER RESOURCES AND RIVER BASINS

India, with a land area of 3.29 million km^2, has an annual average precipitation of 117 cm. Out of this 4000 billion m3 of rainfall, (due to limitations in storing flood flows which occur during the 4 monsoon months) only 690 billion m3 of surface water and 450 billion m3 of groundwater are fit for use (Reddy, 1993) with the present level of understanding in water management practices, without involving major inter-basin transfers. Against these estimates of water availability, we have the demand projections of around 1000 billion m3 within the next three decades. The present level of water use is of the order of 45% of the utilisable surface water (EPTRI, 1996) and 30% of the utilisable groundwater (Prasad, 1994). But still due to uneven distribution of water availability over time and space, we have the typical flood-drought-flood syndrome pervading in several parts of the country (Reddy, 1993).

River basins of India can be broadly divided into three groups based on catchment areas. Besides these, there are a few desert rivers which flow for some distance and there are some completely arid areas where evaporation and rainfall are equal and there is no surface flow. Fourteen major river basins of 20,000 km2 catchment areas and above support about 80% of the Indian population and cover about 83% of the land area (2.58 million km^2) and 85% of the annual average runoff. Forty-four medium sized river basins have catchment areas between 20,000 km^2 and 2,000 km^2. Rivers with catchment area below 2,000 km^2 are called the minor rivers. These are in the coastal areas (Rao, 1979).

WATER QUALITY INFORMATION NEEDS

India has a population of 844 million, which is second largest in the world. There are a total of 644 cities with population ranging from 50,000 to 12.5 million each (Census,1991). Pathogens and BOD/DO problems are generally considered the most critical water quality issues. These are also the focus of all water quality protection programmes. Understanding water quality issues takes time (faecal pollution, organic pollution, salinization, metal pollution, eutrophication, nitrates, organic micro-pollutants). Thus, a country like India will be left with a shorter period of time to tackle the multiple and complex issues of water quality when compared to developed western countries (Meybeck, 1995). Water quality information is often fragmented. Quantity - quality relationships and holistic assessments warrant a better data sharing mechanism between agencies involved in hydrology, hydrogeology and water chemistry & pollution control.

Presently, there are about 9000 large & medium (capital investment > Rs. 5 million) industries (Karforma, 1996). Industrialisation is on a rapid pace with the Government's policy on economic liberalisation. The environmental impact of such economic growth on water resources is yet unknown. The Central Pollution Control Board, now has 20 strong years of experience in the areas of water resources protection, including monitoring and assessment. The inadequacies in the ongoing programmes together with the changing scenarios warrant a reorientation of the programmes in the fresh water quality sector.

MONITORING, ASSESSMENT AND REPORTING

In pursuance of the Water (Pollution, Prevention & Control) Act, 1974, one of the mandates assigned to CPCB is to promote the cleanliness of streams and wells. Accordingly, programmes have been designed and developed in order to maintain or restore the natural water bodies to the level of designated best use requirements. Major elements in this approach were the use-based classification and zoning of the major rivers followed by the development of the water quality monitoring system - MINARS (Monitoring of Indian National Aquatic Resources). CPCB, the national focal point for the implementation of the UNEP programme called, Global Environmental Monitoring Systems/Water, adopted the complete methodology developed under this programme, for establishing (1977) and developing its own MINARS programme in collaboration with the State Pollution Control Boards. Presently, there are 480 water quality monitoring stations. The resulting information from these stations are published in the water quality year books (mean, standard deviation and range). Locations, where violations in water quality with respect to the designated best-use water quality requirements are observed, are also reported in these books.

The ADSORBS (Assessment & Development Studies of River Basins) programme on watershed management studies was also initiated and developed, side by side. ADSORBS contemplates on the preparation of comprehensive river basin documents along with basin, sub-basin inventory of water polluting sources, vis-à-vis stream quality. Such documents have been published for almost all the major basins. A GIS approach to ADSORBS activity is now under study.

The significance of data quality assurance requirements was better realised in the early 1990s and from then programmes were formulated for Analytical Quality Control activities (Rajan, 1991). About 60 laboratories (CPCB & SPCB) participate in these activities covering 15 parameters and these exercises are conducted twice a year. Since laboratories are also the most fundamental units in monitoring, laboratory developments are also progressing rapidly. It is anticipated that about 100 laboratories (SPCB) will be involved in such activities of monitoring by the turn of the century.

CPCB, under a Biomonitoring Yardstick Development Programme, conducted in-depth studies on R.Yamuna and some southern rivers with the Dutch assistance and led to the AMOEBA (A Method of Ecological and Biological Assessment) system of integrated water quality assessment. Experimental studies on the suitability of Automatic Water Quality Monitoring Stations (5 on R. Ganga and 2 on R. Yamuna) are also under way, since 1991.

Additionally, the Central Water Commission (CWC, 1996) \has been gathering an enormous amount of information on discharge characteristics, sediment loads and several basic water quality variables related to irrigation requirements, through its network of thousands of gauging stations. Likewise, the Central Groundwater Board also gathers data on water level and water quality (major ions). Commencing in 1969 with only 410 hydrograph stations the network expanded to 13,690 stations in 1991 and they were to increase it to 20,000 stations by 1995 and further to 30,000 stations. Water levels are monitored 4 times in a year (Jan, May, Aug. and Nov.) and water quality (major ions) is monitored once a year (May) for these stations (Sinha, 1991).

EVALUATION OF WATER QUALITY PROGRAMMES

The key agencies operating national level water quality programmes are CPCB, CWC, and CGWB. CPCB is under the Ministry of Environment & Forests and has a direct mandate to maintain/restore the water quality of the natural water bodies to the level required for their designated best-uses. CPCB is a relatively new organisation which was established in 1974. The water resources utilisation and management are mostly the subject of CWC and CGWB, both under the Ministry of Water Resources. In India water quality (health of natural water bodies) is a relatively recent concern. Water quantity and hydrological measurements have always been of prime concern due to the agriculture based economy. Hence, the scope of the water quality programmes being operated by these agencies (CWC, CGWB) were limited, though there is a real wealth of information with these agencies on water discharge, sediment load and water potential/balance characteristics. Progress on sharing data between agencies is yet to be made.

MINARS

CPCB water quality programmes are all execute in collaboration with the State Pollution Control Boards (SPCB). 51 Indian water quality stations are also included in the GEMS water network. The thrust area of SPCPís is the consent (licensing of industries) management and hence laboratories are focused on the analysis of industrial effluents to check for the compliance with standards. The MINARS programme developed with the help of the GEMS water operational guide freshwater analytical requirements. An important achievement of the MINARS programme is the establishment of a working system for freshwater data collection across the country. However, there are still inadequacies in the MINARS programme, such as improper sampling, data quality, data checking/validation, follow-up with concerned agencies, data processing, water quality evaluation and reporting.

ADSORBS

This programme was aimed at providing a more holistic approach to address water quality issues. A variety of data covering all relevant activities at the river basin level and their impacts on receiving water bodies were collected, compiled and were shown on a number of maps. In this activity data collection involved interacting with a number of agencies resulting in inordinate delay in completion of projects. These documents had more academic value and their use for river basin management was quite limited.

AQC

This programme started in 191 and was the first of its kind in the country. The external AQC was successful and very practical in offering a basis for the participants to understand their data quality. Participation of the concerned labs in this activity is always 100%. Though considerable progress in formulating and conducting external AQC exercises has been made, inclusion of routine internal AQC procedures has lagged behind. Even in the external AQC, the statistical basis for setting acceptance limits is not very sound. AQC for the parameter ìColiform densityî still remains a challenge. The data quality for this parameter is very poor. The evaluation of the GEMS/Water data (UNEP, 1995) also seem to support this.

RIVER ACTION PLANS

Based upon the results of monitoring and assessment activities, CPCB first formulated the Ganga Action Plan (GAP) in the early 1980s for restoration of water quality. GAP was launched in 1985 by the newly constituted Central Ganga Authority (CGA) with Prime Minister as the Chairman of CGA. The Ministry of Environment & Forests established a separate Ganga Project Directorate, now renamed as National River Conservation Directorate (NRCD), to look into the GAP affairs. Water quality monitoring and assessment inputs are the key for the efficient functioning of NRCD. This is particularly required for studying the impact of the pollution control measures (schemes) implemented. According to a recent nomenclature of the NRCD activities, their programmes can be grouped under 4 categories. These are (i) the programmes on the main stem of Ganga, called GAP - Phase I, (ii) the programmes in the major tributaries, Yamuna, Gomti and Damodar, called GAP-Phase II. With respect to the other major rivers, the strategy for restoration was modified by taking up schemes only on those stretches which were classified as polluted river stretches (PRS), unlike in GAP-Phase I. Here again, it was the result of long term monitoring by CPCB, which led to the identification of 41 PRS on the major rivers. (iii) The programmes that are taken up for the PRS are called NRAP (National River Action Plan). (iv) 11 urban lakes have been identified for restoration in Phase I under the programme called NLCP (National Lake Conservation Plan).

The pollution control measures, such as sewerage systems, treatment plants, river front development, sanitation etc., covered under the programmes of NRCD, involve very high financial investments. Water quality monitoring is the only basis for defending such programmes and expenditure. The resources allocated for water quality monitoring are, however, negligible when compared to the expenditure on implemented pollution control measures. Accountability of public money demands strengthening the component on water quality monitoring and assessment programmes. Incidentally, the same view has been expressed in the evaluation of water quality programmes in the United States (ITFM, 1992).

GANGA (MONITORING, ASSESSMENT AND RESEARCH) EXPERIENCE

MONITORING

Water quality monitoring gained significance in GAP while making an inventory of pollution sources and state, and analysing trends in water quality with the progressive implementation of pollution control measures (Mohan, 1995). The GAP water quality monitoring programme started with 30 monitoring stations on the river Ganga with midstream sampling at 30 cm. from the surface, once a month on any day. The data generated between 1986-1990 was analysed and at the same time 147 schemes were implemented under GAP in three states (Uttar Pradesh, Bihar and West Bengal), and 403 million liters of the total sewage flow of 1440 million per day were diverted from reaching the Ganga. However the impact of this and the concurrent reduction in industrial pollution in the Ganga on the river water quality by enforcing effluent treatment plants has not been proportionate. Opposing trends have even been observed in DO and BOD at stations Hardwar, Garhmukteshwar, Kampur, Allahabad and Varanasi. The simulations using TOMCAT (Temporal Overall Model for CATchments) of the Thames Water Authority, U.K., also indicated discrepancies in the data. As a result the GAP water quality monitoring programme was revamped. It was recommended to sample upstream end downstream of major impact stations. The number of sampling points were to increase to 3 across the river width and samples were to be taken at 0.6 times the depth of flow. The sampling frequency was to be extended to three times a month on fixed days. Heavy metals and pesticides analysis (Industrial Toxicological Research Centre) were reduced to once a month for all stations at one sampling point. Under this revamped programme, 15 stations on the main stem of Ganga mostly in Uttar Pradesh and Bihar were handed over to CWC and the remaining 12 stations in the tributaries and the West Bengal stretch were assigned to state pollution control boards. After 30 months of operation of this revised programme, it was observed that the data sets from the different agencies varied widely. There were no quality assurance routines. CWC could not generate the bacteriological data. Thus, the objectives aimed at by the revamping were not fully accomplished.

Keeping in mind all these aspects, the GAP Monitoring and Research Committees held series of expert consultations to develop guidelines not only for GAP, but also for Yamuna, Gomti, Damodar, and National River Action Plans. The revised guidelines were: (i) The monitoring should be done once a month preferably on the 11th day at 30 cm depth of the surface, (ii) For base-line stations and trend stations sampling should be done midstream while for impact stations, 2 samples should be drawn from one fourth and half the width of the river. (iii) Samples should be analysed for core parameters identified as relevant from the last 10 years of data (temperature, pH, velocity, discharge, DO, BOD, COD, Kjeldahl nitrogen, total coliforms and faecal coliforms). However, location specific parameters relevant to the industrial discharges like chromium, mercury, phenol, cyanide, oil etc., may be analysed too. (iv) Sediment sampling from midstream is to be done once in a season (4/yr). Sediments are to be analysed for TOC, heavy metals and pesticides, if the water sample showed such parameters in excessive levels. (v) Seasonal monitoring of outfall drains for waste water flow rate and analysis in each of the action plan towns to be taken up concurrently with the river water sampling in order to determine cause and effect relationships. (vi) In-built mechanism or AQC is to be ensured. (vii) Training programmes on sampling and analysis on water and sediments for heavy metals and pesticides and also for drain monitoring are to be conducted. (viii) Performance monitoring of the sewage treatment plants is to be carried out once a month. (ix) Once a year all samples from all locations are to be monitored for all the 42 parameters to determine temporal trends. (x) A mid-course correction on monitoring strategy should be carried out if the results are not meeting the objectives.

RESEARCH AND DATA INTEGRATION (GANGA)

Under the auspices of the Planning Commission supported by the Ministry of Environment and Forests an integrated Action-Oriented Research Programme on the Ganga was carried out during the period 1983-1989, wherein about 1000 research scholars and teachers from 14 universities participated. Massive research covered the areas of water quality, ecological, hydrological and biological aspects, which resulted in 54 reports. However, this dat abeing incoherent and more academic in nature could not provide such useful inputs to GAP as was anticipated.

The Indian Scientific Documentation Centre (INSDOC) was assigned the task of creating a data base with the reports on the Ganga. The objectives of this data base were (Bharadwaj, 1995): (i) to organise and collate the factual and bibliographic data scattered in different sources into a meaningful collection. (ii) to provide easy access and analysis of the entire data. (iii) to enable graphical presentation and trend analysis. (iv) to act as reference source for further studies. This database termed Ganga ecosystem has now been created.

SUGGESTED STRATEGIES

- Efforts will be made for addressing the ADSORBS activity through a GIS approach with a facility for continuous updating of basin, sub-basin inventory of water polluting sources, vis-à-vis stream quality (Rajan, 1995). The development of a simplified GIS-ADSORBS for the Yamuna sub- basin is already in an advanced stage.
- The use of information highways such as Internet or other national NETs has so far not been well recognised in the water resources sector including that of water quality. A changed perception will bring about better data sharing between agencies involved in these sectors.
- Assessment of the existing water quality monitoring network (480 stations) should be carried out in a step by step manner, starting from complete treatment of the data collected from individual stations(right from their inception) followed by the evaluation of the performance of the station with respect to the objective for which it was created.
- Assessment of water quality issues - Several types of data sets (besides that of CPCB) will be required for this activity. All possible data are to be collected for this purpose. In addition, pollution source inventories should be able to provide indicative results. This activity should result in the publishing of a series of issue specific reports.
- The exercise for network optimisation for the 480 water quality stations should be carried out. Amongst others, factors such as hierarchy in the various levels of monitoring together with sampling frequency and parameters, stream order, the type of impact station (Meybeck, 1995), baseline and flux stations.
- The Indian approach to water quality management has relied heavily on the concept of 'designated best use', in the last two decades. The defining of primary water quality criteria for the designated best use and the classification and zoning of the rivers were the principle elements in this approach. However, conflicts do exist as to whether such an approach has provided satisfactory results. Therefore, it is to be studied and ascertained whether we should adopt the direct water quality classification (comparable to that of EEC classification) or the present system of use-based classification with a thorough review/modifications of the water quality criteria requirements (comparable to that of USEPA classification).
- Monitoring design for the future : Plans for the short term and long term development of the network should be prepared with greater emphasis on impact monitoring. A manyfold increase in the number of stations will be required to adequately monitor the impact of urban population keeping in view the small % of domestic sewage generated from cities and town in India having adequate treatment before disposal into natural water bodies. Urban population, which is 25.72% of the total population (Census, 1991) is going to increase in the coming decades which poses a real threat to the quality of the receiving water bodies.
- A proper hierarching of the networks, with an active role for the Zonal offices of CPCB (North, Central, West, East, North-East and South) should be considered.
- Monitoring of particulate matter in the aquatic environment : Appropriate incorporation of this element in the monitoring system to track the sediment chemistry and pollutant transport. Also potential sites for study of sediment cores should be identified in order to archive the pollution history of the watersheds (Meybeck, 1995), where the industrial development in the last 50 years has been rampant.
- Data quality assurance routine : The present system of Analytical Quality Control (AQC) concentrates only on External AQC by Youden Two Sample Technique. Even here, many difficulties in sample preparation, preservation, setting acceptance limits and statistical treatment of results have been encountered. One of the important activity will be to streamline this work and also to take care of the internal quality control requirements. High Coliform density in our surface resources is generally encountered. This is the single largest factor causing the most severe impairments of water quality in our aquatic resources. But, we have yet not developed any mechanism for AQC for this parameter. This matter has to be addressed.
- Water quality data handling and assessments : All raw MINARS data have to be first manually checked for erroneous values and screening purpose. Systematic procedures for visual screening of data should be developed. Procedures should be formulated for

interacting and clarifying with the concerned State Board labs which would be essential for screening and validation. A system should be developed for AQC flagging with data. Methods for data treatment, assessments and reporting will be formulated and finalised. RAISON/GEMS software of Environment Canada will have a key role in this activity.

CONCLUSIONS

Twenty years of commitment to water quality monitoring, assessment and protection has resulted in the development of sound institutional arrangements at the State, Regional and National level for a country of the size of India. Many programmes in the freshwater quality sector are still in the initial phase of development and there is an emergent need to reorient and strengthen the programmes.

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MODERNIZATION OF LABORATORY AND MONITORING WATER QUALITY NETWORKS

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ABSTRACT

In 1974, the monitoring network was designed to cover the water quality necessities, according to the Mexican situation in those days. At present time, the monitoring network consist on 803 stations distributed around the country. The network has grown without following any systematic basis. Because of increase of the number of industries, wastewater discharges and population, the water quality problems has changed. In other hand, budget restrictions and lack of technical basis to continuosly redesign the monitoring network, the network has became obsolet.
In adition, there's no quality control/assurance system, therfore data are not very reliable; however, they are adecuate to characterize surface waters according to conventional physico-chemical parameters.
For these reasons, the necessity of redesigning and modernizing the monitoring network has became a priority. This will be possible through to the PROMMA program, which will be partially financed by the World Bank.

BACKGROUND AND PRESENT SITUATION

The present situation reflects a chronic budget problem since the mid 1980's when much of the budget was reallocated to other parts of SARH in the expectation that the water quality program would be lost to another department. This loss was never recovered when the water quality program was reconfirmed in CNA in 1990.
Water quality activities have been carried out in Mexico since 1974. The current network of 803 stations is an inherited network of surface and groundwater stations which has grown through more than 20 years without a systematic basis. Although data are adequate to characterize surface waters according to conventional physico-chemical criteria, the network does not provide the type of data required for current water quality management purposes. Regional staffing levels are more or less adequate for field programs, however headquarters is very understaffed for the level of responsibility.

The present situation of the laboratory network consist on: a) one national laboratory which is largely a shell, since all major equipment was relocated to IMTA in 1992, b) six regional laboratories with variable capabilities, c) 27 state laboratories which are mainly focused with very basic types of analysis, although show higher level capabilities.
In theory, the lab network should have high, medium and low level capabilities (national through state levels); however, capabilities tend to depend on available instrumentation rather than any rational allocation of analytical responsibilities. At this time, there is little systematic quality control over data. CNA has no capability for developing organic chemistry analysis; there is a chronic shortage of trained personnel; many modern equipment are unused because of lack of spare or lack of training.

The information system consists of three databases which are held separately: water quality, sewage treatment plants and municipal and industrial discharges. The water quality database is held in the RAISON platform which has been developed by IMTA for use by GSCA. This platform, developed by Environment Canada, has given GSCA modern capability for water data management and assessment. Maps however, are held by region rather than by basin. The three databases do not communicate with each other. Computer equipment is in very short supply and non-existent for water quality purposes in most of the regional and state offices.

PROMMA

The objective of the redesigned water quality program is to provide only data that meets specific management needs, are cost-effective, and represent the types of modern environmental issues that now face Mexico (eg. toxicity). In addition to modernization of the three components of the program (monitoring network, lab network and information system), GSCA is working in modifying national effluent regulations to include modern analytical and toxicity-based standards.

The **Primary Network** will consist of approximately 200 surface and 100 groundwater stations. These will be selected on the basis of a comprehensive review of current water quality data, and socio-economic and demographic information at the basin level. The groundwater stations will rotate so that major aquifers can be represented over the timebase that groundwater quality undergoes change. The surface stations will be supplemented by surveys and special studies to provide ecological data that can not be collected by fixed site stations. GSCA rejects the conventional "list" approach to chemical monitoring as inefficient and very expensive. Instead, it will carry out initial diagnostic studies using environmental chemistry and toxicology that will identify compounds of major concern; these will be added to the monitoring schedule at key stations. The groundwater network will also be supplemented by special studies.

The **Secondary Network** is supporting effluent monitoring and control. The network will be flexible in time and space, and will be developed in coordination with CNA wastewater discharges control programs. The potential for automatic monitoring will be evaluated when this program has been developed.
The monitoring program is regarded as a service function to CNA and other clients. Therefore, the operating principle is that true costs should be paid by the principal clients in order to reduce monitoring costs to the minimum required to achieve the client's purpose.
Number of the fixed stations will be increased or lowered to carry out **Special Water Quality Studies** during a short time periods.
Redesigning of the **laboratory network** includes the following components:

National Reference Laboratory which will be responsible for direction and control of all analytical services provided by or for CNA. This includes quality control and quality assurance of CNA labs and of private labs that provide services to CNA.

Regional Laboratories: Regional labs will have advanced analytical capability including common pesticides.

State Labs: In general, small state labs will be designed to perform simple analysis, however some of them will include special parameter. It depends on specific water quality problems for each zone.

Mobile Labs: Mobile units with basic analytical capability will provide cost-effective services for the remaining state offices (without fixed labs)

Emergency Mobile Laboratories: In order to handle emergency situations, it is intended to supply specific regional offices with a BTEX-equipped mobile facility. These will be self contained units with gas chromatography etc. that can handle emergencies such as oil spills.

Improvement of the **Information System** will consist of: 1) Integration of the three databases into a single database, accessed through RAISON, 2)Implementation of RAISON FOR WINDOWS in conjunction with IMTA and Environment Canada together with appropriate PC equipment, 3) Integration of this system with SIGA of CNA to establish a wide area network; and 4) Development of specialized interpretive software.

Under PROMMA, GSCA will be establishing a variety of data products for public use, as well as data integration, visualization and prediction for issues of concern to CNA management. GSCA will also be examining the potential for revenue generation through the possible sale of information products.

CONCLUSIONS

For the monitoring network, it has been proposed:

1. The Primary Network: It will have aproximatly 200 surface water stations and 100 ground water stations.

2. The Secondary Network: It will support of municipal and industrial effluents monitoring and control.

3. Special Studies. They will support the continuos process of monitoring network redesigning. At the begining, special studies will be carried out in the main polluted basins.

In the laboratory, it has been proposed:

1. National Reference Laboratory which will be responsible for direction and control of all analytical services provided by public or privated labs.

2. Regional and State Laboratories which will be analytically classified, according with the regional problems in:1) basic, 2) basic plus some special parameter(s) and 3) advanced capabilities.

3. Mobile Labs. Mobile units with basic analytical capability will be located in states without fixed labs.

4. Emergency Mobile Laboratories. In order to handle emergency situations it is intended to supply some regional offices with a BTEX-equipped mobile facility.

The National Monitoring Network will have, for data processing, a data base which will include: the treatment plants and wastewater discharges inventories, and all the water quality information. This data base will be compatible with the Raison for windows system.

EXPERIMENTAL MONITORING OF THE RIVER POLLUTION - THE COMPLEX APPROACH

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ABSTRACT

To support the decision-making process of the selection of determinands for the national surface water quality monitoring programme in the Slovak republic a two year survey programme was carried out. This programme was focussed on the identification of organic micropollutants occurring in the Nitra river. The analytical methods applied in this experimental monitoring included various hyphenated techniques as well as supporting toxicological and hydrobiological measurements. The hyphenated techniques were based on coupling of gas chromatography to mass spectrometry and atomic emission detector and of high performance liquid chromatography to fluorescence and diode array UV detectors as well as to mass spectrometer to obtain a maximum information output. Simultaneously, target analyses of selected groups of priority pollutants were carried out regularly. This approach enabled to obtain a very comprehensive view of the occurrence of organic contaminants in the entire river basin. The thorough knowledge of the seasonal and spatial distribution of the various organic compounds helped to select priority pollutants specific and relevant for the target area.

INTRODUCTION

The selection of optimal parameters for any water quality assessment programme depends upon the objectives of the monitoring programme. Appropriate selection of variables will help these objectives to be met efficiently and in the most cost effective way (Chapman and Kimstach, 1992). The acceptance of this principle initiated the experimental monitoring of the river pollution in the Nitra river. Even though the water quality in the Nitra river basin has been monitored regularly under the mainframe of the governmental environmental monitoring for more than three decades the range of organic micropollutants which have been measured is rather limited and includes several representatives of volatile and nonpolar chlorinated hydrocarbons, pesticides and aromatic hydrocarbons. At present, there are efforts of Slovak environmental authorities to broaden the range of monitored organic compounds to become harmonised with the legislation in countries of the European Union. However, a simple one-step transformation of the list of determinands monitored in the Slovak republic so that it could be corresponding with EU list of priority pollutants can hardly be considered as realistic since the present economic situation in the countries with a transient economy requires a strong cost-effectiveness of any of suggested changes to an environmental monitoring programme. Therefore, it has been decided to perform a long-term preliminary survey as a pilot study which would provide an inventory of relevant pollutants as well as their seasonal and spatial distribution prior to any suggestion on modification of the current monitoring programme. The intention of the study was to identify and, wherever possible, also to estimate the concentration levels of the organic pollutants (i.e., parent compounds and their degradation products) occurring in the river Nitra. The study was performed by analytical laboratories in Slovakia, Spain and Austria under the framework of EU scientific collaboration project. Supporting analyses were also performed at the Free University in Amsterdam (the Netherlands) within another project. The cooperation enabled a simultaneous use of different analytical techniques. This helped to confirm the identity of organic pollutants occurring in the Nitra river water. The analytical survey was supported by toxicological and hydrobiological measurements with the aim to investigate the impact of the occurrence of organic compounds on the biodiversity of the river as well as on the toxicity of the water and sediments in the river basin.

PROCEDURES

The choice of sampling points was based on the existing network of the water quality monitoring in Slovakia. Nine sites had been selected, most of them were located downstream the most important point and/or diffuse pollution sources (Fig.1). The selection included also the "background" profile in the beginning of the Nitra river and the profile in the Vah river which is the recipient of the Nitra river. The latter profile was selected downstream the mouth of the Nitra river into the Vah river. The sampling frequency was once a month and it was temporary interrupted during winter time. Water samples were collected in the stream line wherever possible, otherwise, they were collected in the distance of 1 m from the bank.

Since the major goal of the survey was to detect all relevant organic contaminants the selection of sample handling and analytical techniques was done with respect to this goal. To isolate analytes with different physico-chemical properties from the aqueous matrix liquid-liquid extraction (LLE) and solid phase extraction (SPE) were performed, simultaneously. In the LLE part of the sample handling 3 L of the water sample were divided into two equal portions and 1.5 L of the sample was extracted with hexane while the other 1.5 L was extracted with methylene chloride. In the SPE procedures a polymeric material was employed as a standard preconcentration medium, however, also the other SPE materials were tested in several experiments during the survey. Membrane extraction discs loaded with C-18 phase and polymer precolumns for on-line SPE-HPLC systems were employed when the transport by the airmail of enriched samples to a collaborating laboratory was needed. The safety of the transport was supported by the results of the tests of the stability of analytes sorbed on various SPE materials. Such tests were performed by several groups and the stability was found satisfactory for most of the analytes (Lacorte et al., 1995; Liska and Bilikova, 1996).

During two years of experimental monitoring GC-MS was the most extensively applied hyphenated analytical technique. Prior to GC-MS analyses the following sample handling techniques were applied in different laboratories: two-step LLE with hexane; two-step LLE with methylene chloride; SPE using PLRP-S co-polymer and elution with methanol; SPE using octadecyl bonded silica or LiChrolut EN polymer and elution with acetone.

To support the results obtained with GC-MS a GC with atomic emission detector (GC-AED) was used after LLE with methylene chloride and after SPE using C-18 and LiChrolut EN.

To detect pollutants with higher polarity (mostly pesticides and degradation products of industrial chemicals) HPLC based methods using spectral detection (DAD UV and MS) were applied. The suitability of HPLC for pesticide analysis has been reviewed by different authors (Barcelo, 1988; Liska and Slobodnik, 1996). During the experimental survey an on-line SPE-HPLC-DAD UV system (LC-SAMOS) was applied for regular monitoring of different groups of modern pesticides. The identity confirmation was enabled using the UV spectral library. This approach had been successfully used for monitoring of polar pesticides in the Rhine river [Brinkman et al., 1994 (a); 1994 (b)]. The limited identification potential of the DAD-UV system was strengthened by the simultaneous use of diverse LC-MS systems. Representative water samples were analyzed by LC-thermospray (TSP) MS in positive ion (PI) and negative ion (NI) mode using time scheduled selected ion monitoring conditions, by LC-MS with high flow pneumatically assisted electrospray (LC-ESP-MS) and by LC-MS with atmospheric pressure chemical ionisation (LC-APCI-MS).

During the survey it was realized that the sensitivity of the SPE-GC-MS procedure with MS operating in the total ion mode was not sufficient to detect analytes at low ng/l level and also that the sample handling techniques employed would probably fail to enrich the highly volatile compounds effectively. Since it was desired to obtain a comprehensive view on the occurrence of those organic compounds which may contribute to the general toxicity of the water even at very low concentration levels it was decided to analyze the water samples also for selected groups of nonpolar and/or highly volatile organic compounds. Nonvolatile chlorinated compounds (PCB represented by their commercial mixtures and congeners, DDT, Lindane, Heptachlor, Methoxychlor and hexachlorobenzene) were extracted with hexane and analyzed by GC with electron capture detector (ECD). Volatile compounds were analyzed using a static head-space technique followed by GC-ECD (1,1-dichloroethene, 1,2-dichloroethane, chloroform, carbon tetrachloride, trichloroethylene, tetrachloroethylene and dichlorobenzenes) or by GC with flame ionization detector (FID) (benzene and chlorobenzene). Concentrations of 6 PAH's (phenanthrene, anthracene, fluoranthene, benzo(b)fluoranthene, benzo(k)fluoranthene and benzo(a)pyrene) were analyzed using extraction with freon and analysis by HPLC with fluorescence detection.

RESULTS AND DISCUSSION

SCREENING ANALYSES

Up to 200 organic compounds were identified by GC-MS during the survey in collaborating laboratories using different sample handling techniques as specified above. The major classes of compounds identified in the Nitra river water included aliphatic and aromatic chlorohydrocarbons, chlorinated ethers, aromatic hydrocarbons, aliphatic and aromatic oxo-compounds (aldehydes, ketones, phenols, carboxylic acids and their esters including fatty acids and phthalates) and heteroaromates (the major representatives were derivatives of benzothiazole and triazines). The uncertainty of the library search depended mostly on the concentration of the analyte. It was assumed that for about 60% of all identified organic compounds the accuracy of their identification was questionable. The accuracy of the recognition of the molecular structure of an analyte was restricted by the limitations of used quadrupole systems regarding the spectral resolution, distinguishing between isomers and the sensitivity. Additionally, despite the high resolution power of used capillary columns, a large number of peaks that appeared on the chromatograms led in many cases to their overlapping. This resulted in the presence of complex mass spectra and decreased the possibility of their correct identification. However, due to a large number of analyses of the samples from a particular sampling site (i.e., all the analyses performed in a particular laboratory during the two year period plus the analyses performed by different laboratories simultaneously), it can be assumed that more than 50 relevant pollutants were positively identified. The confirmation of identification was supported by availability of multiple and/or replicate GC-MS data and also by running a quality assurance program for GC-MS analyses, simultaneously. This programme was based on regular participation in proficiency testing scheme AQUACHECK organized by Water Research Centre in Medmenham, UK. This complex scheme of round-robin tests includes also analyses of water samples containing unknown organic compounds.

In addition to GC-MS analyses a hyphenated system using an atomic emission detector (AED) was applied to confirm the GC-MS results by verification of the elemental ratios of the eluting substances. On the assumption that the elemental responses were directly proportional to the elemental amount and independent of the structure of the molecules, molecular formulas and concentrations were calculated by external calibration. The identification of substances which cannot be confirmed by their empirical formulas due to insufficient peak intensities, integration problems or chromatographic interferences could be facilitated by the presence of one or more heteroatoms and analysis of specific elemental traces. GC-AED analyses were carried out under chromatographic conditions similar to those used in GC-MS analyses to achieve comparable retention times.

For the semiquantitative calculation of the concentrations of detected compounds an external multielemental single-point calibration with diazinon and 1-bromo-3-chloropropane was

performed. This enabled to obtain concentration estimates for tens of compounds including those quantitated by other techniques.
After evaluation of data obtained from GC-MS of Nitra river samples performed in Bratislava, Barcelona, Vienna and Amsterdam and with respect to the GC-AED results the relevant organic micropollutants occurring in the Nitra river were selected. The selection criteria were the frequency of occurrence, concentration level and estimated toxicity. The pollutants were mostly volatile chlorohydrocarbons, chlorinated ethers and pesticides [Table 1]. Several representatives of these groups were selected as target compounds and for their analysis a SPE-GC-MS method with MS working in the selected ion monitoring (SIM) mode has been developed. Atrazine, s-dichloroethylether and sum of two isomers 1,1'-oxybis[3-chloropropane] and 2,2'-oxybis[1-chloropropane] were being determined regularly in river water samples in the final period of the monitoring programme. This provided information on seasonal and spatial distribution of the selected target pollutants in the Nitra river basin.
To obtain a complex view on organic contaminants in the Nitra river and to confirm the results of the GC-based screening, HPLC analyses were performed, simultaneously. For these analyses UV diode array, fluorescent and mass detection was applied.
UV spectra of the analytes were collected using an on-line SPE-HPLC-DAD UV system. In this system the river water samples were preconcentrated on the microcolumn containing polymer sorbent and trapped analytes were eluted by the mobile phase directly into the LC column. Two UV spectral libraries (containing spectra of more than 40 and 80 pesticides, respectively) were used for the confirmation of the identity of the analytes. The limitation of this system was the low information content of UV spectra. To confirm the identity of a particular compound the information on the UV spectrum had to be combined with the respective retention time. Thus, this system was suitable predominantly for target analysis and it helped to confirm results obtained from GC-MS and LC-MS analyses. Moreover, its response was stable over a long period of time what resulted in a very high reproducibility and hence a good quantitation of analytes. Pesticide oriented spectral libraries were the reason why most of the pollutants detected by SPE-HPLC-DAD UV during the survey belonged to different pesticide classes. Phenoxyacetic acids (dichlorprop, MCPA), triazines (atrazine), dichlorophenols and carbamates (carbofuran) were the most frequently found compounds. Their concentration levels ranged from 0.05 to 1 µg/l.
To investigate the presence of carbamates a special HPLC system with fluorescence detection was used for several months during the initial period of the survey. This system included on-line OPA derivatization using solid phase reactor for hydrolysis of carbamate insecticides. Since no positive results had been obtained with this method, it was decided to focus on LC-MS techniques.
The major aim of LC-MS analyses was to identify the unknown organic pollutants not amenable to GC and also to search for specific target groups of modern pesticides. To reach the latter objective, LC-thermospray (TSP) MS in positive ion (PI) and negative ion (NI) mode using time scheduled selected ion monitoring conditions was applied during the initial period of the survey. The target compounds were triazines and degradation products of atrazine, acetanilides, phenylureas, phenoxyacetic acids, bentazone and its degradation products and benazolin. LC-TSP-MS analyses detected Atrazine, deisopropylatrazine, Monuron, Isoproturon, Metazachlor, 2,4-D and MCPB at concentrations between 0.05 - 0.1 µg/l. TSP interface, however, did not enable the identification of unknown compounds. Moreover, the detection limits of the LC-TSP-MS system were not sufficient to detect pesticides at low concentration levels. Therefore, for further monitoring, ESP and APCI interfaces were applied. Using these interfaces several phthalates (among them di-n-ethyl, di-n-butyl, n-butyl and phenyl) were identified. In some samples using the APCI more phthalate compounds were observed as compared to the data obtained with ESP interfacing system. These findings were a good proof for the hypothesis which states that ESP and APCI are complementary techniques and both of them should be used for monitoring purposes. Except of phthalates, with APCI using selected ion monitoring conditions it was possible to identify Atrazine, Simazine and desethylatrazine with a high certainty. Combined use of APCI and ESP interfaces enabled also to identify Malathion, Fenthion and Cycluron.
The most relevant pollutants with respect to their concentrations and frequency of occurence which were identified by HPLC based methods are given in Table 1.

TARGET ANALYSES

The aim of these analyses was to supplement the information on the occurrence of the organic compounds in the Nitra river with those priority pollutants which were expected not to be detected within the screening analyses due to their low concentrations and/or high volatility. During the two year target survey a thorough information was obtained on the time and spatial distribution of selected priority pollutants in the river basin. In general, four types of distribution were recognized. Pollutants belonging to type I had characteristic spatial distribution along the river during the whole survey, but only negligible seasonal variation of these compounds was observed (e.g. benzene). The compounds with the distribution of type II had remarkable seasonal variation but they had no substantial changes of their concentration downstream the river (methoxychlor). Both seasonal and spatial distribution (type III) was typical for chlorinated volatiles and, finally, no characteristic pattern of the distribution in space and time (type IV) was usual for ubiquitous pollutants having many point sources (PAH's).
Since it was apparent that the results of the analyses for selected priority pollutants in water samples had a dynamic character, it had been decided to investigate the levels of immobilized pollutants, simultaneously. To investigate the accumulation of nonpolar priority pollutants within the river basin, analyses of river bottom sediments were performed three times a year. Increased concentration levels of chlorinated pesticides, PCB's and PAH's were found in the lower reach of the Nitra river as a result of sedimentation of suspended solids contaminated in the industrial areas upstream.
Quality assurance of the target analyses was performed via running an internal and external quality control system based on EN 45 000 standards. Internal quality control included the application of control charts and the use of certified reference materials. External quality control was based on regular participation in AQUACHECK proficiency testing scheme.

TOXICOLOGICAL AND HYDROBIOLOGICAL ANALYSES

Toxicity tests and hydrobiological measurements were performed in order to investigate the quality of the aquatic life in the river and toxicity of river water and sediments towards different organisms. In the water samples collected from selected profiles the species diversity of the phytoplankton and mycoplankton as well as diversity of periphyton and macrozoobenthon were investigated. Based on the species composition data the Saprobic Index according to the Pantle and Buck was calculated to estimate the pollution situation downstream the Nitra river. The saprobic situation ranged from xeno - oligosaprobity to beta - alfamezosaprobity in the longitudinal Nitra river profile. To characterize the community structure the Shannon Diversity Index was calculated as well. The most reduced diversity was found in the upper reach of the Nitra river in the area with chemical industry. As expected, the results from hydrobiological analyse were in conformity with the results of chemical analyses.
To evaluate toxic effects of the river environment different toxicity tests were carried out. For water samples the acute toxicity to *Daphnia Magna* (Cladocera, Crustacea; ISO 6341/1989, 48 hours test) and to *Poecilia reticulata* (static test; ISO 7346/1984, 48 hours test) was determined. Besides, tests of the inhibition of the growth of the green algae *Scenedesmus quadricauda* (ISO 8692/1989, 72 hours test) and of the inhibition of the growth of the radix of *Sinapis alba* (96 hours test) were performed. Similar tests were applied also for water leachates of stream sediments from the Nitra river. For most of the water and sediment samples no toxic influence was observed. Therefore, in the second phase of the monitoring water samples were preconcentrated prior to the ecotoxicological analyses using octadecylsilicagel. In this procedure 3000 ml of the water sample were pumped through the column at the rate of 5 ml/min and analytes were eluted with 1.5 ml of methanol. For analysis of the eluates two additional bioassay methods were employed - Thamnoxkit F (acute toxicity to Thamnocephalus platyurus) and Rotoxkit F (acute toxicity to Brachyonus calicyflorus). Using the enrichment procedure positive results were obtained for sampling sites 2-6. The highest effects were found in profiles 4 and 5.

CONCLUSIONS

Combination of different analytical techniques in analyses of the Nitra river samples resulted in a very comprehensive information on the occurrence of organic micropollutants in the Nitra river basin. Approximately 200 organic compounds were identified using different spectral detection techniques. The strategy applied during the whole experimental monitoring was the simultaneous use of techniques providing the same (e.g., parallel use of GC-MS in different laboratories), similar (GC-MS and GC-AED) and complementary (GC-MS and LC-MS) information. Two major advantages of the strategy of the method selection were the confirmative (e.g., GC-AED with GC-MS for industrial pollutants or GC-MS with LC-MS for atrazine) and the complementary (LC with GC in general, LC-APCI-MS with LC-ESP-MS for phthalates) roles of analytical techniques applied. Based on the results from different analyses, priority pollutants specific for the Nitra river basin as to their concentration levels, frequency of the occurrence and toxicity could be selected. Despite the large amount of results obtained, still many occurring compounds remained unidentified. This can be partially ascribed to an experimental character of the LC-MS techniques, at present. It must be also taken into account that some of detected compounds were coeluting under different GC and/or HPLC operational conditions used in the survey and that the superimposed MS spectra became unidentifiable. An additional problem was that the spectra of most of the degradation products were not available in existing MS libraries. Thus, it is obvious that, despite a reasonable progress in the analytical instrumentation that has been done in the last decade, the future developments of multidimensional hyphenated analytical systems have still promising perspectives to make these techniques available to solve the problems of exact characterization of environmental pollutants.

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COMPOUND	Sampling Poing	GC-MS (WRI)	GC-MS (TUV)	GC-MS (VU)	GC/AED	LC-DAD-UV	LC-TSP-MS	LC-APCI-MS	LC-PB-MS (VU)
Atrazin	2,3,4,5,6,7	D	D	D	D	D	D	D	D
1,1'-oxybis[3-chlorpropane]	2,3,4,5,6,7	D	D	D	D				
2,2'oxybis[1-chloropropane]	2,3,4,5,6,7	D	D	D	D				
chloroisopropyl dichloroisopropylether	2,3,4,5,6	D	D	D	D				
s-dichloroethylether	2,3,4,5,6	D	D	D	D				
1,2-dichloropropane	2,3,4,5,6	D							
1,2,3-trichloropropane	2,3,4,5,6	D	D		D				
1,1,3,4-tetrachloro-1,3-butadiene	2,3,4,5	D		D					
naphthalene	2,3,4,5,6	D	D	D	D				
2-ethyl-1-hexanol	2,3	D	D	D	D				
1-[1-methyl-2-(2-propenylox)ethoxy]-2-propanol	2,3	D		D					
benzothiazole	2,3,4,5,6,7	D		D		D			
2-(methylthio)-benzothiazole	2,3,4,5,6,7	D		D					
2,4-D	4,5,6				D	D			
MCPA	4,5,6				D	D			

Table 1: List of organic compounds considered as relevant pollutants in the Nitra river with indication of the hyphenated technique used for their identification and the sampling point where the compound was detected. For numbering of sampling points see Figure 1. WRI = Water Research Institute, Bratislava, TUV = Technical University Vienna, VU = Free University Amsterdam

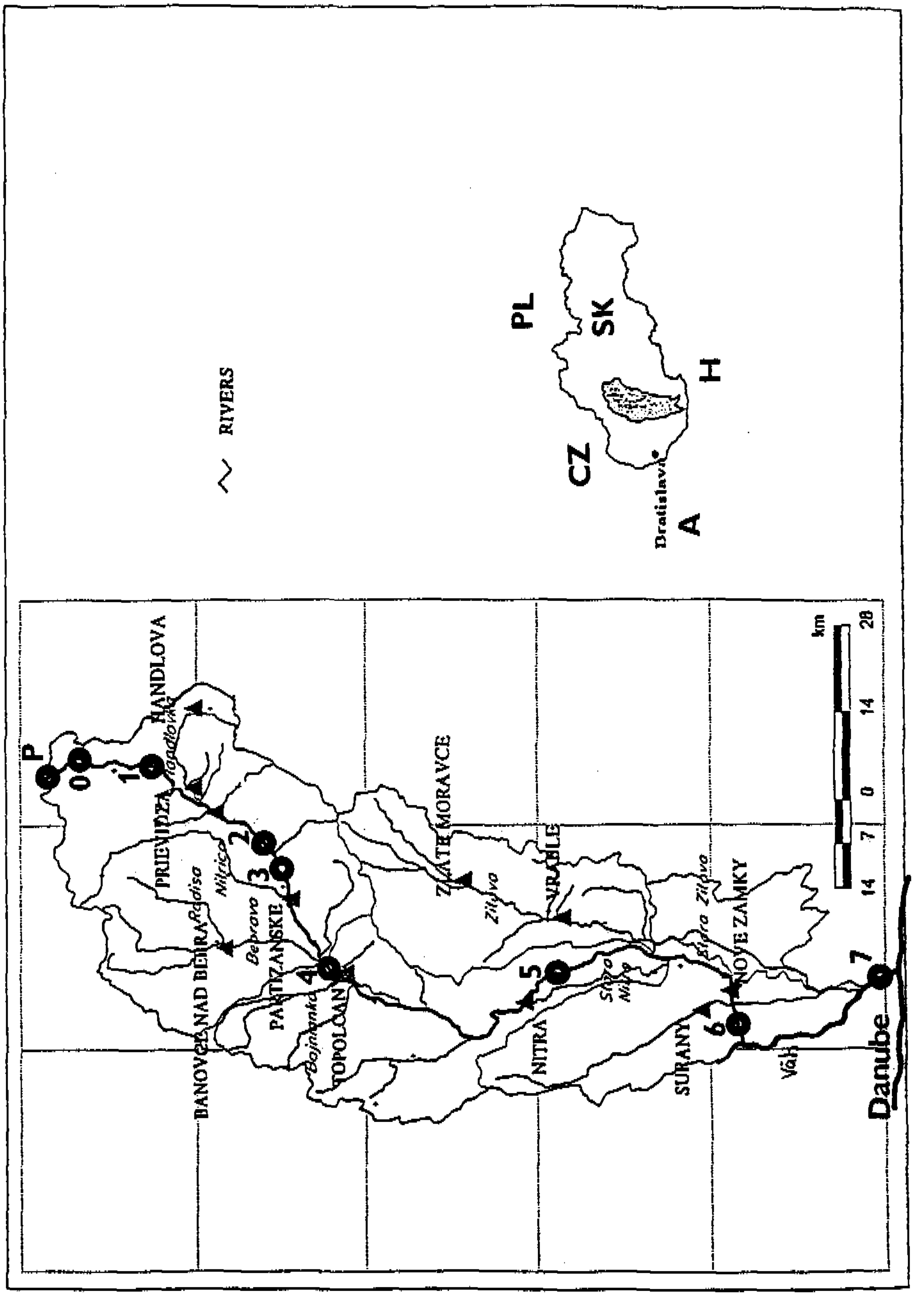

Figure 1: The map of the Nitra river basin with indicated sampling points

RAISING THE QUALITY OF WATER AS A PART OF WATER MANAGEMENT IMPROVEMENT IN THE DANUBE RIVER BASIN - THE MAIN OBJECTIVE OF THE MONITORING, LABORATORY ANALYSIS AND INFORMATION MANAGEMENT SUB-GROUP (MLIM-SG) WORK

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ABSTRACT

This paper aims to describe the work of the Monitoring Laboratory Analysis and Information Management SubGroup (MLIM - SG) carried out in the frame of Environmental Programme for the Danube River Basin (EPDRB) and focuses on one of the major objectives specified at the start of the Programme as well as in the Strategic Action Plan : the development of an international water quality monitoring network, the harmonisation of the sampling and field activities procedures, to enhance laboratory analyses capabilities, and to form the core of an international information management system. An enlarged network of monitoring stations was developed to examine major problems from Danube River Basin including tributaries. A number of determinands and the sampling frequencies for different water environmental factors were adopted as well as unique protocols for exchanging the data between the National Information Systems (NIS) and / or NIS and a Basin Management Centre. The renewing and harmonisation of the field and laboratory technical capabilities, quality control management and necessary training were, and remain the main topics of the SubGroups.
The paper focuses on the actual capabilities with regard to Danube Monitoring and the necessary activities to improve and strengthen the Danube and tributaries monitoring as well as the quality and comparability of the data for the benefits of water management actions.

INTRODUCTION

BACKGROUND

The quality of environment in the Danube catchment area is under great pressure from a wide range of antropic activities. Urban settlements are generating pollution from a large number of activities, for example inadequate waste water treatment process, solid waste disposal facilities and industrial waste emission in to the air, water and on land. The intensification of agricultural practices, animal husbandry and irrigation expansion without a proper control, has produced nonpoint sources of pollution of surface and ground water in the catchment area. The environmental impact of industry, agriculture and urban pollution sources has resulted in significant water, soil and air pollution at local, regional and transnational levels.
There is an overwhelming need for a basin-wide approach to environmental management, and river water quality. Pollutant levels in the main Danube are currently similar to those of the Rhine, whose quality has improved markedly over the last 10 years and due to a co-ordinated

Programme of environmental improvement, although the concentration of oil and some heavy metals is much higher in the River Danube. In the tributaries, the condition is even more severe and concentrations of pollutants exceed acceptable standards. The critical interdependence of upstream and downstream neighbours for managing environmental quality can be seen at local, regional and national levels in the basin. Co-operative action at the international level must therefore be a guiding principle of sustainable management of the Danube River Basin.
A further factor that contributes to the compelling case for a basin-wide approach is the serious deterioration of environmental conditions in the Black Sea, which is significantly influenced by the Danube discharge as well as from other tributaries.
The Environmental Programme for the Danube River basin was jointly established by the riparian states of the Danube in September 1991. To assist in the implementation of the Action Programmes, Sub-Group III - Monitoring, Laboratories and Information Management was set up specifically to address the development of an international water quality monitoring network, to introduce harmonised sampling procedures, to enhance laboratory analysis capabilities, and to form the core of an international information management system.
The Sub-Group, which now consists of one representative from each of the eleven Danube riparian countries (ex-Yugoslavian countries are missing because of war embargo) has been complemented by the formation of three small purpose designed Working Groups (WGs) dealing with three main areas of interest -Monitoring, Laboratory Management, and Information Management. These groups are chaired by members of the MLIM SG who are specialists in these topics, and the WG membership is derived from national experts from the Danube countries. The WGs address problems related from a List of priorities based on the activities from an Implementation Plan approved by the Sub-Group. Wherever possible, the WGs will take on, under the overall supervision of the SG, specific responsibilities for equipment procurement, training and applied research.

The broad objective of the MLIM SG to improve environmental management in the eleven countries participating in the Danube Environmental Programme are settled in the MLIM SG Mission Statement :

Working within the framework of the internationally funded Environmental Programme for the Danube River Basin, to address the targets and actions set in the Strategic Action Plan, and according to the provisions of the Danube River Protection Convention, the Monitoring, Laboraty and Information Management Sub-Group will:

- establish a Transnational Monitoring Network (TNMN) of representative sites for the chemical and biological quality assessment of the River Danube and its major tributaries.
- establish reference laboratories in each Danube riparian country that will analyse all TNMN samples, and that could act as national laboratory accreditation centres in each country.
- provide an information management system for the TNMN, that will alsointerface withnational informationsystems in each Danube riparian country.
- provide training oppotunities at levels appropriate to the needs of each Danube ripariancountry; first priority training areas are biological assessment using macroinvertebrates, assessment of pollutant loads in sediments, and the derivation of data exchange information systems.

  It will achieve these prime objectives, and other tasks identified in the Implementation Plan, through three specialist working groups which will make recommendations for action to the Danube Programme Coordination Unit through the Monitoring, Laboratory and Information Management Sub-Group.

EXAMINATION OF THE CURRENT SITUATION ON MONITORING

THE SYSTEM CREATED IN THE BUCHAREST DECLARATION FRAME

The riparian countries which are signatories of the Bucharest Declaration are to be congratulated in developing an international monitoring programme in a commendably short period of time. The Bucharest Declaration was accepted in Bucharest on 13 December 1985 by the Governments of the Danube riparian countries. The Declaration states that the riparian countries, according to their respective legislation and within their technical and economical possibilities, are ready to take measures that will :

- protect the Danube from pollution, with special regard to dangerous and radioactive substances;
- gradually decrease the degree of pollution of the Danube;
- establish national systems for the monitoring of waste-water discharged into the Danube;

The riparian countries inform each other of measures taken to fulfil the agreed objectives. They exchange information concerning plans for wastewater treatment, water quality standards and research. Furthermore, they inform each other about accidental spills, floods and ice.
To observe the development of the state of the Danube the riparian countries have, in the framework of bilateral and multilateral co-operation, established a monitoring programme based on agreed methods in order to obtain comparable data. Each country has appointed competent national agencies to carry out the water quality and quantity monitoring. At least every two years, these agencies compare their results either on a bilateral or multilateral basis.
Monitoring stations (cross-sections) are primarily located according to national borders :
where the Danube flows from one country to another,
1 where the Danube constitutes the border, stations will be placed at the start and at the
1 end of this reach or in other places within this reach if agreed on bilateral basis.
The riparian countries can, as a secondary option, agree on a bilateral basis to establish monitoring stations at other locations such as up- and downstream of major tributaries, town, reservoirs, etc.

Over ten years after the acceptance of the Declaration, the activities carried out in the framework of the Declaration can briefly be summarised. The implementation of the Declaration has been stepwise and has now resulted in annual meeting (bi- or multilateral) where the monitoring results are discussed.
The monitoring network consists of eleven water quality stations and thirteen flow stations. All stations are placed on the Danube itself and all the water quality stations are placed according to the border criteria.
Sampling and field measurement are carried out on a bilateral basis, where the two countries agree on the date and time. Each country takes its own samples and measurements and analyses the samples in its own laboratory. At each station three samples are taken (left bank, middle, right bank) 0.5 metres under the surface of the water .
As can be seen the current situation on monitoring had a good start at the transnational level through the Bucharest Declaration but this was limited only to the major course of the river, at a limited number of determinands and media of investigation, and with a low level of quality of the data at the national level of development and a very heterogenic level as well.

PRIORITIES IN THE DANUBE RIVER BASIN AND MEANS TO IMPROVE CURRENT PRACTICE ON MONITORING

PRIORITIES IN THE DANUBE RIVER BASIN

There are two main priorities which were established for the Danube River Basin :
1. Sustainability of water utilisation like for :
- drinking water
- industrial water supplies
- irrigation
- fisheries
- recreation

In respect with this the improvement is related to the reduction of microbial contamination, nutrients content, heavy metals and organic micropollutants content (oils, pesticides, etc.)
Protection of the aquatic ecosystem for the Danube and its tributaries, in the reservoirs, in the wetlands, in the Danube Delta and, as a result of this, in the Black Sea.
In respect with this the improvements should to be performed in reduction of inorganic nutrients (N & P), biodegradable organic loads and toxic material inputs.

MONITORING ACTIVITY

As can be seen the water resources are used for several and usually competing purposes. This fact is reflected in the different objectives to be used for selection of the station location and type, as well as determinands and methods. For the Danube Basin three major requirements can be defined :
- Protection of drinking water sources .
- Minimising and monitoring of point sources for effective discharge management
- Monitoring general water quality (for the protection of general aquatic life, fisheries, bathing waters, etc.)

General water quality monitoring was divided into :
- Transport monitoring (load or mass balance of dissolved or particulate substances)
- Concentration level / ecological condition monitoring

Transport, or load, monitoring is a very important consideration. Objectives for this kind of monitoring can be :
- Estimate of inputs to downstream ecosystem (basic studies)
- Trend studies
- Predictive modelling

Objectives for concentration level / ecological condition monitoring were considered to be :
- Definition of background conditions
- Basic studies in relation to standards
- Trend analyses
- Impact studies
- Predicative modelling
- Early warning / emergency warning

The defined objectives and goals were used for a monitoring programme to determine the station / determinands / frequency in the programme.
For covering these objectives the monitoring network was in some areas set up as a multi-objective network involving a large number of methods and determinands and in other areas as more simple networks. The process of determining the areas where the different objectives will be

active starts with an in-depth investigation of all factors and activities, which exert an influence, directly or indirectly, on water quality. The design of an optimal monitoring network was based on careful, preliminary planning and investigations.

IDENTIFICATION OF LOCATION FOR ADDITIONAL MONITORING STATIONS AND ORDER OF IMPLEMENTATION

The design of a monitoring network should build on clear objectives. In the "Environmental Programme for the Danube River Basin - Programme Work Plan" was stated that the monitoring network for the Danube shall :
- strengthen the existing network set up by the Bucharest declaration
- be capable of supporting reliable and consistent trend analysis of concentrations and loads of priority pollutants
- support the assessment of water quality for water use
- assist in the identification of major pollution sources
- also include sediment monitoring and bioindicators
- include quality control

Furthermore, it is stated in the Terms of Reference for the project, that :
- the monitoring network shall provide outputs compatible with other major international river basins in Europe;
- the monitoring network probably in time shall comply with standards used in the western part of Europe;
- the design shall split into immediate and longer term needs - start with practical and routine functions already performed.

As a standard from the western part of Europe, some of them were used several times, so that the monitoring network was designed in the manner that can monitor in compliance with the appropriate EC - Directives, whether or not they become legally enforceable. This will therefore allow a step-by-step approach to be taken in the politico-economic climate that prevail in the riparian states.
Five EC-Directives of relevance were used. Station location and the determinants took the requirements of these into consideration.
Council Directive 78/659/EEC : Quality of fresh waters needing protection or improvement in order to support fish life.
Council Directive 76/160/EEC : Quality of bathing water (now recently modified (COM (94) 36)).
Council Directive 91/271/EEC : Urban waste water treatment.
Council Directive 75/440EEC : Quality required of surface water intended for the abstraction of drinking water in the member states.
Council Directive 79/869/EEC : Methods of measurements and frequencies of sampling and analysis of surface water intended for the abstraction of drinking water in the member states.
Therefore, considering these points, it was recommended that an extension to the existing network of Bucharest Declaration monitoring stations throughout the Danube basin be made for the purposes of :
- Safeguarding the health of humans using the Danube (and tributaries), as a source of drinking water;
- Safeguarding all other agreed uses of the Danube such as commercial and recreational fisheries, bathing and other water contact recreations, as a habitat for flora and fauna, as a source of irrigation water, industrial uses and so on;
- To provide an objective and reliable source of data on water quality against which to judge the effectiveness of point and diffuse source pollution abatement measures.

The Transnational Monitoring Network (TNMN), which builds upon the foundation laid down by the signatory States to the Bucharest Declaration, was developed using the following criteria for the selection of stations :

- Located just upstream / downstream of an international border;
- Located at confluences between the Danube and major tributaries or major tributaries and lesser tributaries ;
- Located downstream of major point sources;
- Located according to the need to monitor for important water usage (e.g. drinking water intakes).

The second criteria was selected to enable estimations of mass balance. The vital linkages of the TNMN with the national monitoring networks are obtained by using existing stations rather than new ones. Under the existing Bucharest Declaration the last three criteria are secondary option to the first criterion to be established on a bilateral basis. Thus far the thirteen stations in the Bucharest Declaration network are those selected under the first criterion. To improve upon the current situation we have recommended, still within the spirit of the Declaration.
Phase 1: A network of seventhy Monitoring Stations, which comprises eleven Meteorological Stations (Type A) and fifty-nine Water Quality / Quantity Monitoring Stations (Type B), of which forty-three will also serve as Sediment Monitoring Stations (Type C). The Water Quality monitoring network will therefore be extended from the existing border-sampling stations to include important sites for the determination of mass balances, sites of significant industrial / urban inputs and sites of important water usage.
Phase 2: A network of one hundred and fourteen Monitoring Stations, which comprises twenty-one Type A stations, ninety-three Type B and fifty-nine Type C stations to bring about the necessary conjunction of flow measurement with quality measurement. The numbers of Type B and Type C stations (and combined B/C stations) are also increased to provide the essential greater linkage with the national monitoring programmes.
All stations recommended for the TNMN currently exist as part of the network of National Monitoring Stations. Therefore, no additional sampling stations need to be established. There is some additional effort required by each country by increasing the frequency of sampling, sampling sediments and biota, and by changing the number of determinands. The judge was to be a minimum additional workload that could be absorbed by the staff and laboratories involved. The recommendations for the TNMN, on this basis, are the most cost-effective and have the endorsement and enthusiastic support of the local experts.
It was recommended that a 5-year action plan are adopted for the implementation of the TNMN. Phase 1 will probably take two years and Phase 2 should be fully implemented by the end of the 5-year period. We have also recommended a major review of the TNMN at the end of Phase 1 to take account of knowledge gained during that period and to adapt to the rapidly changing situation in many riparian states.
The study focused on river water quality of the Danube and its major tributaries; groundwater quality in the Danube catchment area was examined also. The aim was to provide a transnational monitoring network (TNMN) that would produce reliable information to allow cost effective, and affordable management decisions to be taken to achieve the objectives set-out in the MLIM SG Mission Statement. This information needs to be derived from the analyses of representative samples taken at key monitoring sites that reflect meteorological, water quality and quantity, and sediment quality considerations; the biological quality assessment of river water is a priority issue.

LABORATORY ANALYSES

Laboratories had to be identified that were capable of analysing water and sediment samples for trace amounts of contaminants - often at concentrations as low as one part in one thousand million parts of water! This has necessitated the provision of expensive" state of the art" analytical equipment through an approved Equipment Procurement Programme, and training laboratory staff in the use of such equipment. The result is that each country will have a national centre of scientific expertise - a National Reference Laboratory, that can extend its own training programmes and "accredit" other water laboratories and institutes in each Danube country to enable an enhanced system of public health and environmental protection to be achieved in the Danube basin.

INFORMATION MANAGEMENT

The analytical and flow data from each country has to be transformed into information that can be used to satisfy all legal international agreements, such as the Bucharest Declaration and the new Danube River Protection Convention (DRPC), to demonstrate compliance with EC Directives and local discharge permits, and to provide management information to prioritise massive expenditure on environmental improvements from scarce financial resources. The MLIM SG is setting-up a data exchange file facility that links, in a cost effective manner, national databases in all Danube countries to a common file for the TNMN that can be accessed by all countries for all these purposes. In the same time an appropriate and harmonised National Information Centres Network was designed and developed in all riparian countries.

TRAINING ACTIVITY

A further area of activity for the MLIM SG is to oversee an Integrate Training Programme that covers all aspects of the SG's activities. The training programme, supported by EU funds, is provided mainly by approved Danube country institutes with some assistance from western European experts. A key training programme will include senior government officials and directors of Danube institutes and statutory water bodies. It is very important that such "decision makers" understand and appreciate the value and importance of the work of the MLIM SG and the objectives of the EPDRB. In the same time, an harmonised, and similar as level and type of methodologies, series of courses relating to sampling, analysis, and data management are the major part of the ITP

MLIM-SG LIAISON LINKS

A key feature of the MLIM SG programme has been to establish important liaison links with other workers, institutes, and international organisations in order to share knowledge and expertise, prevent duplication of effort, and to save money.
There is a close working arrangement with the AEWS Sub-Group and the Applied Research Programme (ARP) of the EPDRB; in fact research proposals submitted by the MLIM SG are part of the ARP programme, and the two SG's are involved in the specification and future installation of continuous automatic monitoring stations for alarm and routine monitoring purposes.
The recent establishment of the interim International Commission for the Protection of the Danube River (DRPC) in Vienna has resulted in joint meetings between the Interim Secretariat and the MLIM SG, with the objective of incorporating the SG activities into the DRPC structures when all the Danube countries formally ratify the new Danube Convention. All future SG activities will operate in close liaison with the Interim Secretariat in order to achieve common goals in a cost effective manner.
Also at the international level, the MLIM SG has established routine contact with the Rhine Commission with its 40+ years history of improving the quality of the River Rhine. Additionally, liaison arrangements are in place with the Elbe Commission, the Rhine Basin Programme, the Black Sea Environmental Programme (BSEP), the Helsinki Convention Task Force on Monitoring and Assessment, the WMO - Hydrological Forecasting Commission. The importance of the fragile aquatic ecosystem of the Danube Delta and the north western shelf of the Black Sea (B.S.) dictates an integrated approach by the BSEP and the EPDRB, and the MLIM SG work

programme addresses this issue.
The MLIM SG liaisons with the UNEP Global Environment Monitoring System (GEMS) based in Canada, and with the EU Project Management Units that operate in a number of Danube countries. Such contacts enable the MLIM SG members to stay abreast of the latest developments in water management at the national and international level.

CONCLUSIONS

In summary (see Box) the MLIM SG is on track to achieve its objectives in the implementation of the EPDRB as approved by the Task Force. Over the next year the SG work programme will be integrated into the activities of the interim International Commission set-up under the DRPC. The members of the SG will then know that their voluntary efforts to achieve a sustainable environmental improvement of the River Danube and its tributaries will operate under the force of a new international legal framework. The beneficiaries of their efforts will be all the inhabitants of the Danube basin and the north- western coastal region of the Black Sea.

SUMMARY BOX

The MLIM SG work programme is no "head in the clouds" scientific exercise; it is a logical development that links good sampling techniques and sound analytical water quality data to an information management system that is focused on maintaining and improving the environmental quality of the river Danube and its major tributaries, and reducing pollution loads to the Black Sea.
Extensions of the principles of the programme in all Danube countries will improve the quality of life for all inhabitants of the basin by protecting public health, and sustaining the ecosystems of the basin to the benefit of a wonderfully diverse flora and fauna in Europe's second largest river system, The EPDRB needs and deserves the support of all the peoples of the basin to help in achieving its aims and objectives.

AKNOWLEDGEMENTS

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DESIGN OF A FRESHWATER MONITORING NETWORK FOR THE EUROPEAN ENVIRONMENT AGENCY AREA

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ABSTRACT

The European Topic Centre on Inland Waters (ETC/IW) was appointed in December 1994 by the European Environment Agency (EEA) to act as a centre of expertise for use by the Agency and to undertake part of the EEA's multi-annual work programme. The key Task of the first year's programme was to design a freshwater monitoring network for the EEA area. This Task is summarised in this report.
The main task of the Agency is to provide the European Union and the EEA Member Countries with:

'objective, reliable and comparable information at a European level enabling them to take the requisite measures to protect the environment, to assess the results of such measures and to ensure that the public is properly informed about the state of the environment'.

Information is thus required on:

- *the status of Europe's water resources (status assessments); and,*
- *how that relates and responds to pressures on the environment (cause-effect relationhips).*

The proposed network for the EEA to obtain the information it requires is designed to give a representative view or assessment of water types within a Member State and also across the EEA area. It will ensure that similar types of water body are compared. The need to compare like-with-like has led to a stratified design with the identified and defined strata containing similar water bodies. The use of the same criteria for selecting strata and water types across Member Countries will ensure that valid status comparisons will be obtained. The EEA network will be representative of the size, number and types of water bodies in the EEA area (e.g. small rivers), variation in human pressures (e.g. population density and land use), and, will include a number of reference and flux stations.

INTRODUCTION

The European Topic Centre on Inland Waters (ETC/IW) was appointed in December 1994 by the European Environment Agency (EEA) to act as a centre of expertise for use by the Agency and to undertake part of the EEA's multi-annual work programme. The key Task of the first year's programme was to design a freshwater monitoring network for the EEA area. This paper describes the need and use of such a network and explains why existing international sources of monitoring information are not adequate to meet all of the EEA's needs. The basic concept of the network is outlined; more detailed technical descriptions and justification of the proposal are given in other reports (e.g. Nixon et al 1996). The proposal has been peer reviewed by the EEA's National Focal Points and was presented at a workshop in Madrid in June 1996. It was agreed at the workshop that the network should be progressively implemented across Europe and the way forward to achieving that goal is also described.

THE NEED FOR INFORMATION ON EUROPE'S ENVIRONMENT

WHY IS INFORMATION NEEDED?

Over the past two decades four European Community Action Programmes on the environment have given rise to about 200 pieces of environmental legislation. While a great deal has been achieved, the general state of the environment continues to slowly deteriorate. This assessment was made in The Fifth Environmental Action Programme based on a Report on the State of the Environment. The Action Programme highlighted the need for 'a more far reaching and more effective strategy' which could only be assured if the quantity and quality of information was good enough. Several deficiencies in the available environmental information at a European level were highlighted:

- a serious lack of base-line data, statistics, indicators and other quantitative and qualitative material required to assess environmental conditions and trends, to determine and adjust public policies and to underpin financial investments;

- an almost complete absence of the more precise, quantitative data on human interventions and influences on the environment which are needed for meaningful modelling exercises and the optimisation of European policy and large scale investment decisions;

- information which is available is often not processed or presented in a suitable form for potential end users, administrations, enterprises and the general public and does not take into account the different levels of sophistication or simplification required, nor the fact that different types of decision require different types or levels of information.

Against this background, it was decided to establish a European Environment Agency.

ROLE OF THE EEA

The European Environment Agency (EEA) was established by Council Regulation (EEC) No. 1210/90 of 7 May 1990. The Regulation describes in detail the role and tasks expected of the Agency. The main task of the Agency is to provide the European Union and the EEA Member Countries with:

***'objective, reliable and comparable** information at a European level enabling them to take the requisite measures to protect the environment, to assess the results of such measures and to ensure that the public is properly informed about the state of the environment'.*

In 1995 the EEA published a major 'State of Europe's Environment' report (the Dobrí_ report) based on the best available data/information at that time, mainly from national and international monitoring programmes. The aim of the report was to present an assessment of environmental status in relation to the pressures and human activities impacting on it. The report supported goals to:

- develop a comprehensive Environment for Europe programme addressing in particular trans-boundary environmental problems;

- provide a sound basis for effective measures, strategies and policies to address environmental problems nationally and regionally; and,

- *inform the public and raise awareness about the common responsibility for the environment.*

Information was presented on the status and trends of Europe's water resources, and comparisons made, where possible, of differences between Member Countries. It also identified key issues involving water resources, which should be addressed by the policy makers if improvements were to be achieved.

Though a very comprehensive report it highlighted significant gaps as well as large discrepancies in the quality of environmental data and information. One of the most significant gaps was the almost complete lack of comparable and reliable data on groundwater quantity and quality. In general comparison of surface water quality across Europe was very difficult because of the lack of comparable and reliable data. There was in particular a lack of data on small rivers and lakes.

The EEA has the duty to update the Dobríš report in 1998 and is also required to produce monographs on specific issues such as groundwater quality/quantity and eutrophication.

WHAT ARE THE KEY WATER RESOURCE PROBLEMS AND RELATED POLICIES?

The Dobríš report defined Prominent Environmental Problems (PEPs) for Europe. The PEPs in relation to water resources are:

- **Management of freshwater** in terms of **availability** and **quality** (for example, **groundwater pollution, eutrophication** and **organic pollution** including pathogens), and in terms of **physical changes** of water bodies.

- **Acidification.**

Further problems were identified in a state of the environment report for the European Union (EU) produced by the EEA to update the one presented in 1992 by the European Commission, and to contribute to the review of the 5th Environmental Action programme. In terms of water resource issues, the problems identified include, **acidification** and **nitrate levels** in drinking water. Current policies were also considered not to be sufficient to tackle issues such as **water abstraction**, the **quality** of **groundwater** and **chemicals** in the environment.

These and other issues and problems have been further detailed in policy developments such as the Groundwater Action Programme, the proposed Directive on the Ecological Quality of Water and the 5th Environmental Action Programme (EAP). The latter was published in March 1992 and states that community policies must aim at:

- prevention of **pollution of fresh and marine surface waters and groundwater** with particular emphasis on prevention at **source**;

- restoration of natural ground and surface waters to an **ecologically** sound condition, thus ensuring a suitable source for extraction of drinking water;

- ensuring that water demand and water supply are brought into equilibrium on the basis of more rational use and management of water resources.

These problems raise questions which when answered will provide the information on whether policies are working and problems are being solved. It is also necessary to identify measurable indicators in terms of the information needed to answer these questions.

WHAT INFORMATION IS REQUIRED?

Information is required on:

- the status of Europe's water resources (status assessments); and,
- how that relates and responds to pressures on the environment (cause-effect relationships).

Status assessments need to be based on the most up-to-date information (ideally only a year between collection and reporting) and be comparable in terms of the water types (e.g. lakes, small rivers, large rivers) and in terms of how and what aspects are being measured. A judgement of how 'good' or 'reliable' the status information is must also be made. A statistical framework for collecting and presenting information is therefore required. The status information must also be collected and presented at the correct scale or level of aggregation. For example, many water-related problems are caused from within the catchment and, for proper assessment, information must be collected at this level. There will also be a need to aggregate such information at a later stage so that statements about European water resources can be made.

The assessment of the effectiveness of policies and legislation also needs information on the 'pressures' (causes) that might be effecting the 'status' of water resources. Thus the aim will be to establish 'cause-effect' relationships. This assessment will be across media. For example, atmospheric deposition can cause problems such as lake acidification, and land-use can also significantly impact water quality/quantity. Pressures include sources of contaminants entering water bodies at single points or more diffusely, agricultural and land-use practises in catchments, human interventions such as flood defence/engineering, reservoir construction and overexploitation of groundwater resources.

EXISTING SOURCES OF MONITORING INFORMATION

NATIONAL MONITORING NETWORKS

Member Countries monitor water resources according to their national requirements (e.g. legal and operational) and international obligations (e.g. European Commission (EC) directives and International Agreements). The information arising from this monitoring is potentially a major source for the EEA.

Although national monitoring networks are designed to meet their national and international obligations, statutory or otherwise, they also must meet other needs and objectives. For example, general surveillance data from a larger proportion (compared to that required by international statutory requirements) of the total national water resource may be required. Operational data, often at a sub-catchment level, may also be needed, for example, to monitor the impact of specific discharges on water quality. There will be obvious benefit at national level, where possible, in replicating the purpose of sampling points and also in usage of the information obtained. It is likely, therefore, that monitoring networks associated with directives and international obligations will not represent the total monitoring networks of individual nations. For surveillance purposes, sample sites may be located in relation to changes in water quality, perhaps associated with point discharges or tributaries. Where there are gradual rather than discrete changes of quality, for example along a river, the optimum number of sample locations needed to define overall quality would be quantified through an assessment of the spatial and temporal variability of the determinands of interest in that river.

MONITORING REQUIRED UNDER EC DIRECTIVES

The information required by the European Commission from Member States is primarily for assessing implementation of, and compliance, with directives rather than for the provision of information on the general status or quality of water resources. It is this latter type of information, provided in a comparable way from a representative sample of Europe's water resources, that is required.

As an example the European Commission has recently produced a report on the Freshwater Fish Directive (COM 1995). The directive requires a national summary of:

- total number of designations of salmonid and cyprinid fisheries and how many of those comply with the standards associated with the directive.
- total length of rivers and area of lakes designated and complying with the directives requirements.
- total area of lakes designated/complying.

Fourteen parameters are required to be monitored but no numerical data are required to be reported to the Commission. Table 1 summarises Member State's implementation of the requirements of the directive. It shows that the information on salmonid and cyprinid fishery designations has been presented in 3 ways: as numbers, as percentages (of totals) and as lengths of river. Similarly some Member States monitor all 14 parameters whereas others do not indicate which parameters are monitored or monitor different sized sub-sets. It will, therefore, be difficult to compare designations between States and there will be no quantitative information on the status of designated waters.

	B	DK	D	GR	ES	F	IRL	I	L	NL	PT	UK
Implemented	Y	Y	Y	Y	Y	Y	Y	N	Y	Y	Y	Y
Salmonid	25	49%	147	9	27	294	33	6	12	2	?	50,400 km
Cyprinid	130	30%	189	16	113	120	-	2	3	352	?	5,600 km
Parameters monitored	ND	All?	Sub-set	Core of 8	ND	ND	Core of 7	ND	All?	Core of 10	ND	All?
Reported	1	1	1	1	0	0	2	0	3	4	0	2

Notes:

B	*Belgium*	*DK*	*Denmark*	*D*	*Germany*	*GR*	*Greece*
ES	*Spain*	*F*	*France*	*IRL*	*Ireland*	*I*	*Italy*
L	*Luxembourg*	*NL*	*Netherlands*	*PT*	*Portugal*	*UK*	*United Kingdom*

Table 1: Summary of implementation of the Fisheries Directive in EU12 Member States (COM 1995)

Implemented	Have the requirements of the directive been implemented into national legislation
Salmonid	Designations as salmonid rivers
Cyprinid	Designations as cyprinid rivers
Parameters monitored	14 parameters given in directive
Reported	Number of times reported to European Commission

Thus, information from directives is not suitable as:

- The data are not comparable because the degree of comparability will depend on the interpretation of the designation rules and national differences of how these are implemented.

- The data are not representative because in the directives which require routine monitoring the requirements are generally site specific, either at sites designated for a specific use, sites affected by a specific discharge, or, for the Exchange of Information Decisions, agreed sites in main rivers. As the choice of sampling location is, for some directives, related to areas designated by the Member States rather than by the European Commission, it is unlikely that, for those directives, a comparison of quality across Europe of these designated waters will give a complete picture of quality.

Even for the Exchange of Information Decision where sites are supposed to be selected on the same basis the information is not representative because only nationally large or significant rivers (and one lake) are included.

INTERNATIONAL AGREEMENTS

There are a large number of international agreements concerning surface waters, however, not all of these make monitoring requirements. Many agreements aim to protect a specific water body and are made between countries within the catchment of that water body. For large rivers and seas this can involve many countries, for example, the agreements made at the North Sea Conferences are made between all countries bordering the North Sea. By contrast, there are many agreements which exist between just two countries. Thus the scope of application for international commitments varies greatly.

Information from International Agreements will be of use to the EEA. However:

- to be of use data will have to be comparable between the different agreements;

- data will represent only those waters covered in the agreements that is the major water bodies/catchments in Europe.

CONCEPT OF THE PROPOSED NETWORKS

Previous sections have described the need for objective, reliable and comparable information on Europe's water resources, and illustrated that it is unlikely that the information supplied by Member States for compliance with EC directives and other International Agreements will be suitable or adequate to meet that need. There is, therefore, a need for a network through which the information required (by the EEA) can be obtained. However, not all the Agency's needs can be covered by monitoring efforts only. These also require research on cause-effect relations and on policy implementation, but the proposed freshwater monitoring network will contribute largely to the EEA-needs. The best option for developing a new network, in terms of cost-effectiveness and feasibility, is to obtain information from a **representative** sub-sample or portion of the total national and European water resources.

REPRESENTATIVE ASSESSMENT OF WATER RESOURCES

The proposed network is designed to give a statistically **representative** view or assessment of water types within a Member State and also across the EEA area. It will ensure that similar types of water body are compared. For example at present comparisons are made on the basis of loose selection criteria such as important rivers, large rivers etc. The proposed design will ensure, for example, that small rivers are compared with small rivers, deep lakes with deep lakes, and so on. The need to compare like-with-like has led to a **stratified design** with the identified and defined strata containing similar water bodies. The use of the same criteria for selecting strata and water types across Member Countries will ensure that valid status comparisons will be obtained.

OVERALL OBJECTIVE

The overall objective of the monitoring (information) network is:

"To obtain timely, quantitative and comparable information on the status of inland waters (groundwater, lakes/reservoirs, rivers and estuaries) from all EEA Member Countries so that valid temporal and spatial comparisons can be made, and so that key environmental problems associated with Europe's inland waters can be defined, quantified and monitored".

To minimise cost implications, where possible **the monitoring network will be based on existing national and international networks**, use existing sources of monitoring information and create, only if necessary, an EEA database of aggregated data and information rather than of raw non-processed data.

It should be emphasised that the information provided by the network will NOT be for the assessment of compliance of Member States with the requirements of European Commission directives.

REFERENCES CONDITIONS AND STATIONS

The process of quantifying the effect of human activities (to establish cause-effect relationships) on water resources requires the separation of natural factors and their effects from the impact due to human activities. This will be aided by the identification of reference rivers, lakes and aquifers. In these locations or catchments there will only be minimal or no human activities, there would be natural land cover and would be in most cases be representative of pristine conditions. The quantification of chemical, physical and biological conditions at these sites would give a measure of reference conditions or levels against which 'impacted' rivers etc. would be compared. In this way anthropogenic impacts can be separated from natural variability so that valid comparisons can be made. In some parts of Europe and for some water types reference conditions would not occur because landscapes and water bodies have a long history of human activities and changes. In others such as in the more remote Nordic regions there would be many examples of reference conditions. The statistical aim of the network is to detect significant differences between 'similar' areas (or strata) and the trends with time within areas (or strata). In this way improvements due to investment and the extent of problems will be detected and quantified.

STRATIFICATION CRITERIA

The natural criteria that might be appropriate for the selection of strata include bio-geographical, hydrological, catchment altitude and area, river stream order (size), lake depth and aquifer geology. In this way similar water bodies would be identified, reference conditions quantified and valid comparisons made. The anthropogenic factors which must also be taken into account, particularly if the relationship between state and pressures are to be established, include catchment use and activities (for example agricultural forested catchments, urban/industrial catchments), population density, abstraction rates and discharges of pollutants from point sources (pipes) and

from run-off from land or atmospheric deposition (diffuse sources).
Once appropriate strata have been defined, rivers, river reaches, lakes and aquifers would be allocated to the strata. Ideally there would be existing monitoring stations for each of the stratified water bodies. In reality in many cases there will not be. It is therefore proposed as an initial step that existing monitoring stations should be stratified. At some stage the stratification based on water body will be compared with that based on monitoring station. If there are large discrepancies between the two *(for example if small rivers are not sampled representatively)* then additional or re-assigned monitoring may have to be undertaken. In some Member Countries there will be large gaps in the proposed network because, for example, there are currently no national lake or groundwater monitoring programmes.

The allocation of rivers etc. into comparable strata also offers the possibility of reducing the amount of sampling or sites that need to be included in the network without the loss of information. This is because the amount of overall variability in indicators (for example a chemical or biological measure) might be potentially reduced. There is also potential for reducing sampling bias by selecting monitoring sites from similar water types/catchments etc., thus obtaining fairer comparisons between regions on a stratum basis. If, however, too many strata are produced there is always an opportunity of recombining data from different strata into larger strata perhaps using some form of stratum weighting (e.g. on the basis of frequency of occurrence of particular 'water type' in the whole water resource).

WHAT IS THE BENEFIT TO MEMBER COUNTRIES IN PROVIDING THIS INFORMATION?

At present there is not enough comparable information to obtain a quantitative assessment of water resources across Europe. This can lead to unfair or incomplete comparisons being made and wrong conclusions drawn. By submitting information within this proposed framework a 'level playing field' will be obtained so that Member Countries will have confidence in the conclusions being drawn. In addition the information will enable environmental policies to targeted correctly *and cost-effectively.*

SUMMARY

In summary the EEA network will:

1. Be **representative** of the size/numbers/types of water bodies in the EEA area (e.g. small rivers), variation in human pressures (e.g. population density and land use), and, should include a number of reference and flux stations.

2. For **rivers**, have **reference, representative, impact** (part of representative network) stations, and **flux** monitoring stations at discharge into sea, or at international boundaries.

3. For **lakes**, have a general surveillance network comprising **reference** and **representative** lakes, and if necessary, (in the light of experience) an **impact** network with lakes selected on the basis of population density. In addition the largest and most important lakes (nationally) will be *included and possibly a specific cause/effect network of lakes.*

4. For **groundwater**, have a general surveillance network comprising **representative** stations on ***all nationally important aquifers*** *(groundwater in porous media, karst groundwater and others)* ideally at a density of **1 station per 20 km² to 25 km²** of aquifer. In addition the feasibility of establishing **reference** stations in aquifers not affected by human activities will be assessed.

THE WAY FORWARD

The proposed network design has been piloted in four volunteer Member Countries during the early part of 1996. The four countries were Austria (groundwater quality network), Denmark (lakes and rivers network), Spain (rivers water quantity network) and UK (rivers network in England and Wales). The results from these initial studies were presented at the Madrid workshop and will be reported separately. At the workshop most Member Countries (Germany expressed reservations) endorsed the implementation of the proposed monitoring network across the EEA area in a step-by-step process.

The plan for the next phase of the ETC/IW workprogramme under the 1996 subvention period is to further develop and test the next in the first four countries and to start the implementation process in other volunteer countries. The latter countries are Belgium, Finland, France, Ireland, Norway and Sweden. During these initial phases of the implementation the basic design of the network will be tested using real national monitoring data, and if required, the network design will be modified. During the first phase emphasis will be placed on establishing a uniform approach to site selection using the proposed stratification design. This will serve to highlight additional differences and gaps in exiting national and international monitoring programmes. At a later date issues such as determinand comparability and sampling methodology will have to be addressed so that comparable information can be obtained. For many countries there will be a need to consider developing new monitoring programmes where major gaps exist (e.g. absence of a lake or groundwater monitoring network) and/or modifying current analytical procedures so that comparable data can be obtained. The results of these studies will reported to Member countries periodically - the next progress report is due in February 1997.
Ultimately the aim is to have the network in place so that it can quickly provide up-to-date and comparable information on the state of water resources in the EEA area to be used in future updated reports on the State of Europe's Environment or in key issue monographs such as those to be produced by the ETC/IW on Groundwater Quality and Quantity, and on Eutrophication during 1997.

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EC DIRECTIVES AND THEIR IMPACT ON EUROPEAN WATER POLICY

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ABSTRACT

Most of the early EC legislation on water was based on use-related directives laying down standards for individual parameters for the protection of specific uses of the water (e.g. abstraction to drinking water). The adoption of the Dangerous Substances Directive (76/464/EEC) introduced the dual approach allowing Member States to apply either Uniform Emission Standards (UESs) or Environmental Quality Objectives (EQOs) for the control of the particularly dangerous List I substances. More recently a number of industry sector specific directives have been adopted in particular the Urban Waste Water Treatment Directive (91/271/EEC) laying down emission values for sewage discharges depending on the sensitivity of the receiving water.

The realisation that neither the UES nor the EQO approach on its own is adequate for the control of pollution to the aquatic environment has led to the introduction of the combined approach in the recently adopted EC Integrated Pollution Prevention and Control Directive (96/61/EC). This requires the application of "best available techniques" (BAT) and strict Environmental Quality Standards (EQS) for the control of the most polluting industries.

In addition proposals have recently been made for a new Water Management Framework Directive, which will require the classification of waters and water quality improvements programmes integrating both quality and quantity considerations. This directive will incorporate the existing use related directives including the proposed Ecological Water Quality Directive with the exception of the Bathing Water Directive (76/160/EEC), which is currently being revised. The proposed directive will also require chemical and biological monitoring of the aquatic environment, which is essential, whichever method is used for the control of discharges, to ensure the uses of the aquatic environment are protected. The product directives and the eco-labelling scheme are the main instruments to reduce diffuse pollution of the aquatic environment.

INTRODUCTION

In the original Treaty of Rome adopted in 1957 no reference was made to the environment. However, increasing concern about the state of the environment in Europe and the concern that different environmental requirements could lead to distortion of trade resulted in the Council of Ministers agreeing in 1972 that measures to protect the environment can be agreed under Article 100A and 235 dealing with prevention of distortion of competition. A separate Environment Title (Articles 130 R-T) was introduced for the first time in 1987 in the Single European Act. The Maastricht Treaty signed in 1992 extended the role of the European Parliament and, with a few exceptions, decisions can now be reached by qualified majority.

EARLY EC LEGISLATION

Most of the early EC environmental legislation on water was based on use-related directives laying down standards for individual parameters for the protection of specific uses.

USE-RELATED DIRECTIVES

The different use-related directives are:
Surface Water (Abstraction to Drinking Water) Directive (75/440/EEC)
Bathing Water Directive (76/160/EEC)
Freshwater Fish Directive (78/659/EEC)
Shellfish Water Directive (79/923/EEC).

Member States are required to designate waters to which the directives apply. Once a water is designated for a particular use the standards laid down in the EC Directive for the protection of the use are legally binding.

Implementation of the use-related directives has been patchy in many countries particularly in those which base their pollution control on the use of industry specific uniform emission standards. In addition, generally only those waters (except bathing waters), which already met the requirements of the directives were designated under the directives. However, the standards laid down in the directives are frequently used as reference values to ensure that the limit values set for effluents are stringent enough for the protection of the water use.

DANGEROUS SUBSTANCES DIRECTIVE

The adoption of the Dangerous Substances Directive (76/464/EEC) in 1976 introduced for the control of particularly dangerous substances (List I) the dual approach allowing Member States either to apply Uniform Emission Standards (UESs) or Environmental Quality Objectives (EQOs). The Dangerous Substance Directive also required that the less dangerous substances (List II) should be controlled by emission standards based on Environmental Quality Objectives. Minimum legally binding Standards (UESs and EQOs) for the control of a number of List I substances were laid down in daughter directives of the Dangerous Substances Directive whereas Member Governments were required to set EQOs for List II substances.

DEFINITION OF ENVIRONMENTAL QUALITY OBJECTIVE (EQO) AND UNIFORM EMISSION STANDARD (UES)

Although the two terms, EQO and UES, are broadly understood it is nevertheless important to clarify the definition and application of the two concepts in order to fully appreciate the implications in practice.

ENVIRONMENTAL QUALITY OBJECTIVES (EQOS)

In most EC Member States the term Environmental Quality Objective (EQO) has generally been interpreted as long term aim (Qualitätsziel) to be achieved some time in the future often without legally binding commitment to achieve the EQO at a specified time. Frequently the "long term aim" is defined as the background concentration.

In contrast, in the UK the term Environmental Quality Objective (EQO) is reserved for the use of the water (e.g. abstraction for drinking water) whereas Environmental Quality Standards (EQSs) are set as minimum statutory (legally binding) standards for the protection of the identified use. Effluent discharge standards are derived to ensure that the EQSs outside the immediate mixing zone are not exceeded taking into account other point and diffuse inputs and the flow regime of the receiving water. In broad concept this makes allowance for some dilution of the effluent with the receiving water, however, the concentration limits for the discharges are set to ensure that the uses of the water are protected.

Thus there are fundamental differences in the interpretation of the terms EQS/EQO in the different countries and it is important that these differences are clearly understood.

The derivation of the EQSs requires a critical examination of all available data on the substance. Particular emphasis is placed on toxicity data including chronic exposure (e.g. reproduction, early life stages). Based on the toxicity data available appropriate uncertainty factors are applied for the derivation of the standards. The standards are compared with any field data available which may lead to a reassessment of the standards. Thus the aim of the standard is to protect all species likely to be present from long-term exposure to the particular chemical. This standard is usually expressed as annual average concentration. If the chemical is likely to enter the environment also as a result of episodic events (e.g. pesticides during application or first run-off) standards are also set as maximum allowable concentrations.

The EQS approach relies on the existence of adequate data particularly on the aquatic toxicity for the target species. The uncertainties in the available data is compensated for by the size of the uncertainty factors applied.

Periodic assessment of the health of the ecosystem present near discharges is important to minimise the uncertainties. Thus the EQO/EQS approach requires an effective chemical and biological monitoring of the aquatic environment. Standards also need to be regularly reassessed in the light of new data.

UNIFORM EMISSION STANDARDS (UESS)

The UES approach requires that industries belonging to the same industrial sector must achieve the same effluent standards. These standards are usually expressed as concentration in the effluent discharged and as the load of pollutant discharged per unit of production.
For industries discharging dangerous substances in their effluent emission standards are generally based on "best technical means available". In practice this means that the UESs are based on the best practice available for the particular branch of industry. Thus some economic considerations are generally included in the assessment of the UESs. These standards are applied as minimum legally binding standards.

Emission standards for a number of List I substances have been adopted in daughter directives of the EC Dangerous Substances Directives. The standards are usually based on monthly averages, which may be considered as corresponding to the EQSs expressed as annual average concentration, and as daily maximum concentration, which may be compared with the EQSs expressed as maximum allowable concentrations. The method used for assessing compliance with a standard and the type of sample taken have an important impact on the stringency of the standard set.
The UESs are applied as minimum legally binding requirements. More stringent standards may be set if this is required for the protection of the use of the receiving water.

ASSUMPTIONS AND LIMITATIONS OF THE TWO APPROACHES

Uniform Emission Standards
The derivation of UESs takes into account economic and technical considerations thus standards vary for different sectors of industry. The standards are minimum requirements and are not always stringent enough to protect the environment. However, UESs have advantages for the control of discharges to large river catchments (e.g. large national or international rivers) particularly if they are controlled by a number of regulatory authorities. UESs are therefore favoured in countries where the administration is decentralised. They are easy to apply by the

regulators and once adopted in legislation there is no need for the regulator to justify the standards to the discharger. Another potential advantage of the application of UES is the avoidance of market distortion. However, other factors have much larger impacts on competition (e.g. transport and labour costs) and if more stringent standards need to be applied for the protection of the receiving water the benefit of avoiding market distortion no longer applies. UESs are only applicable for the control of point source discharges. Environmental monitoring is required to ensure that the minimum standards are stringent enough to protect the uses of the receiving water.

Environmental Quality Standards (EQSs)

The EQS approach relies on the existence of adequate data to derive the standards. For many compounds only sparse data are available requiring the application of large uncertainty factors. This approach therefore requires effective environmental monitoring (chemical and biological) to ensure that the EQSs are stringent enough to protect the uses of the receiving water, which includes ecological valuable systems. The approach is easier to apply for catchments under the control of a single regulatory authority.

EQSs are essential for small rivers to ensure that the emission values set are stringent enough for the protection of the receiving water. They can be applied for the control of both point and diffuse inputs, they take into account multiple sources and they can be used for the setting of priorities for improvements. EQSs are more economically efficient, where resources are limited, by relating the required improvements to the capacity of the receiving water.

Comparison of UESs and EQSs

A comparison of the dilution required for the different UES values set in the various Dangerous Substances Daughter Directives to meet the corresponding EQSs is given in Table 1. The data are for the protection of freshwater life. The table shows that for many substances the EQSs are more stringent than the UESs as the dilution required to avoid exceedance of the EQSs are, for many substances, much larger than available in most EC rivers. The Standards set were based on the technology and scientific knowledge available at the time and were based on political compromise. However, the Table highlights the importance of EQSs to ensure the aquatic environment is protected.

Substance	Emission Standard	EQS	Dilution Required for UES to meet EQS*
Mercury	50 µg/l	1 µg/l	50
Cadmium	500 µg/l	5 µg/l	100
HCH-Lindane	2,000 µg/l	0.1 µg/l	20,000
'Drins'	2 µg/l	0.03 µg/l	60
Carbon tetrachloride	1,500 µg/l	12 µg/l	125
Chloroform	1,000 µg/l	12 µg/l	83
DDT	200 µg/l	0.025 µg/l	8,000
Pentachlorophenol	1,000 µg/l	2 µg/l	500
Trichlorobenzene	1,000 µg/l	0.4 µg/l	2,500
Perchloroethylene	500 µg/l	10 µg/l	50
1,2 Dichloroethane (EDC)	1,250 µg/l	10 µg/l	125
Trichloroethylene	500 µg/l	10 µg/l	50
Hexachlorobenzene	1,000 µg/l	0.03 µg/l	33,000
Hexachlorobutadiene	1,500 µg/l	0.1 µg/l	15,000

Table 1: Comparison of EQS and UES values for List I Substances

IMPLEMENTATION OF THE DANGEROUS SUBSTANCES DIRECTIVE

The implementation of the Dangerous Substances Directive has been slow. Standards for only 17 substances have been adopted so far. Proposals had been made to designate a further 15 substances as List I which required unanimity under the Single European Act. This proposal has now been withdrawn partly as a result of the Maastricht Treaty ,which eliminated the need to designate these substances as List I by unanimity, but also in response to the EC Integrated Pollution Prevention and Control (IPPC) Directive and the revision of the Dangerous Substances Directive (see below).

Few EC Member States have derived EQS for List II substances as required by the directive. Particularly those countries which use the industry sector specific effluent control had difficulty in implementing the directive for List II substances especially as they considered the EQSs as long-term aims to be achieved some time in the future.

INDUSTRIAL SECTOR EC LEGISLATION

Although initially the emphasis has been placed on the use-related approach for the protection of the aquatic environment, the Titanium Dioxide Directive (78/176/EEC) was an early attempt to adopt industry specific directives and the Dangerous Substances Directive can also be considered partly a sectorial approach for specific substances. More recently the UES approach has been favoured by the EC resulting in the adoption the Nitrate (91/676/EEC) and particularly the Urban Waste Water Treatment Directives (91/271/EEC). However, in the future the most polluting industries will be controlled under the new EC Integrated Pollution Prevention and Control (IPPC) Directive (96/61/EC). This requires the combined approach of applying "best available techniques" (BAT) and strict EQSs for the control of the most polluting installations and certain polluting substances, and the minimisation of the overall impact on the environment. A similar approach is already being applied in the UK, France and The Netherlands for the control of the most polluting industries and selected dangerous substances.

PRODUCT DIRECTIVES

The product directives prohibit the use of certain products or restrict the types of products which may be used. For instance, the Detergent Directive requires a certain degree of biodegradability before a surfactant may be marketed in detergents. Similarly, the Pesticides Directive (91/414/EEC) restricts the pesticides which may be approved for use and the Marketing and Use Directive (76/769/EEC) and its amendments control the use and marketing of certain dangerous substances. These product directives are likely to become more important for the control of diffuse sources of pollution. However, even though the directives may prohibit or restrict the use of substances these substances may nevertheless still be present by being imported with raw materials or semi-finished products (e.g. pesticides on imported wool).

ECO-AUDITING AND ECO-LABELLING

The recently adopted EC environmental auditing and eco-labelling schemes may become significant instruments to achieve reductions in emissions of both point and diffuse sources to the aquatic environment. Although voluntary it is likely that increasing numbers of companies will participate in the schemes.

REVISION OF EC LEGISLATION

The adoption of the IPPC Directive requires revision of the Dangerous Substances Directive. Any future Dangerous Substances Directive will focus on the smaller (non-IPPC) point source discharges and may include different control options such as abatement equipment specifications.

The realisation that an assessment of the receiving water quality is required to identify priorities for improvements and to ensure that the discharge consents set are stringent enough to achieve the desired water quality the EC Commission has made proposals for an Ecological Water Quality Directive.

However, a fundamental re-examination of EU water policy is currently taking place and a Commission Proposal for a Council Directive Establishing a Framework for European Community Water Policy has recently been published. The Directive will require a river basin management approach and the establishment of a monitoring programme for ground and surface waters. It is envisaged that, with the exception of the Bathing Water Directive, the current use-related directives will be incorporated into the new directive. The Directive will also require the development of water quality improvement programmes and a wider consultation with and information of the public.

CONCLUSIONS

With the adoption of the IPPC Directive the EC is moving towards a combined approach of using UESs and EQSs for the control of the most polluting industries and certain polluting substances. To achieve a general improvement in the quality of the aquatic environment of the Member States the EC Framework Water Policy Directive will require a river basin management approach and the classification of waters and water quality improvement programmes. It is also envisaged that the EQS values laid down in the different water use directives will be incorporated into the new Framework Directive. The new directive will also require chemical and biological monitoring of the aquatic environment. This is essential, whichever method is used for the control of discharges, to ensure the uses of the aquatic environment are protected. The product directives and the eco-labelling scheme are the main instruments to reduce diffuse pollution of the aquatic environment. However, for the reduction of diffuse inputs of nitrate to the aquatic environment the EC Nitrate Directive is the main tool especially for the protection of groundwater used for the abstraction to drinking water.

ACKNOWLEDGEMENT

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TAILOR-MADE GUIDELINES: A CONTRADICTION IN TERMS?

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ABSTRACT

Tailor-made monitoring and assessment imply providing the right information to make management decisions, for a reasonable price in the given economic context. Guidelines can be developed for the strategy of tailor-made monitoring. The policy life cycle and the monitoring cycle are guiding principles. A quantification of the weaknesses in the process of monitoring and assessment may learn that the specification of information needs and institutional aspects dominate over the technical weaknesses.

INTRODUCTION

This paper deals in brief with the strategy for tailor-made monitoring and assessment. It will address the following points:
1. Tailor-made or not tailor-made, that is the question;
2. Tailor-made guidelines, a contradiction in terms?
3. Attempting to develop tailor-made guidelines;
4. Weak links in the process; and
5. Conclusions.

TAILOR-MADE OR NOT TAILOR-MADE, THAT IS THE QUESTION

Tailor-made monitoring implies making monitoring more effective and efficient. In what context do we use the word "effectiveness"? Effectiveness in monitoring, within a tailor-made context, is often measured in terms of the information deemed necessary to answer management questions. What information is needed by management to make decisions? What if the information is not available when the decision must be made? The manager will make the decision using the best knowledge available, with or without information from monitoring! A "wrong" decision may result. However, it may be deemed more efficient to make a wrong decision than no decision at all. Tailor-made implies "*providing the right information* to make management decisions".
Tailor-made also stands for cost-effectiveness. Effective monitoring is never a waste of money. However, it may be too expensive in a given economic context. The tailor may prefer to use other materials, less expensive ones. That is also characteristic of tailor-made monitoring: It stands for the right information at a reasonable price and for reasonable investment, operation and maintenance costs in your own economic situation. It stands for communication between the customer (the information user) and the designer of the monitoring process (the tailor). This makes the monitoring system sustainable.
The opposite of tailor-made is "ready-made monitoring". This is monitoring without strategy, without vision. It is just a matter of keeping up with your neighbours, and only consists of tools that are ready for use. You can buy them, order them. *It has been designed without taking account of* your specific situation. Much more can be said about ready-made monitoring. I will only mention

two characteristics:
- It is trendy: high-tech is "in", western style, you should have it, whether you need it or not. It looks professional to have automatic monitoring stations, to have at least GCMS, HPLC, AAS in your lab and GIS and DSS for the processing of your data.

- It does not consider the socio-economic situation: The ratio between the costs of automation and labour costs make many tools only affordable and accountable in countries where labour costs are very high. As a gift IBM delivers the computers and Hewlett Packard the laboratory equipment, but who argued that it was necessary, and who will pay the high costs for spares, maintenance and consumables?

I would not dare to say that your monitoring is ready-made and mine tailor-made. We have all more or less of the one and of the other. But whom the cap fits, let him wear it.

TAILOR-MADE GUIDELINES, A CONTRADICTION IN TERMS?

Guidelines for tailor-made monitoring appear to be a contradiction in terms.
We found that tailor-made stands for unique, specific, nonconformist. On the other hand, guidelines have the character of standards, rules, conformity, the universal best solution. This sounds like a contradiction.

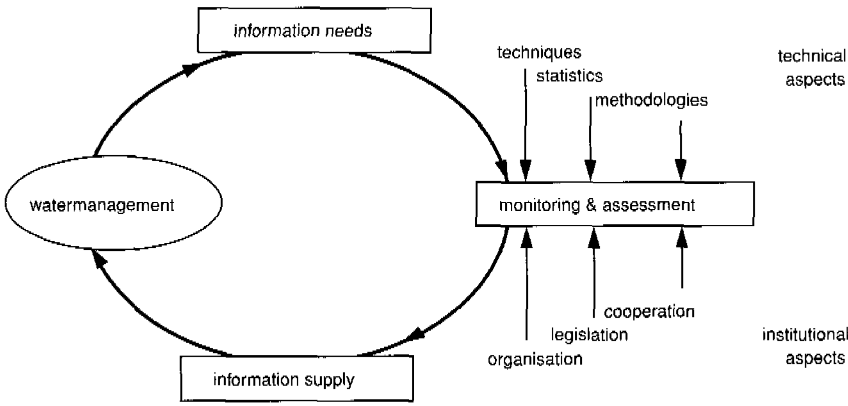


Figure 1: tools for monitoring

Technical as well as institutional aspects play a role in the design of a monitoring programme (Figure 1). We can distinguish between techniques, methodologies, statistics, and even training courses for specialists who use them. These are the tools that we use. For the selection and application of the right tools we use standards, manuals, handbooks. Besides, there are the strategies: What kind of tools should be used in a specific context?
Tailor-made or ready-made, the same tools and skills are used for the clothes we wear. You do not need a professional tailor to teach you how to use the tools. However, the proper tailor can teach us how to use them in a unique situation. He is using principles for his expert judgement. These are "strategic" guidelines, rather than technical ones. With these characteristics tailor-made guidelines are not a contradiction in terms.

TAILOR-MADE GUIDELINES, IS THAT POSSIBLE?

Last year guidelines for the monitoring and assessment of transboundary rivers were drafted by the Task Force on Monitoring and Assessment under the UN/ECE Convention on the Protection and Use of Transboundary Watercourses and International Lakes (ECE, 1996). Are those the "guidelines for tailor-made monitoring"? At least they were an attempt in that direction. They will be tested in pilot-projects for European river basins during the next three years, and they will be revised later, on the basis of the findings of the pilot-projects. They are not handbooks or operational manuals. You may find such handbooks and manuals in the excellent publications of WHO, WMO and other organisations (WHO, 1992; WMO, 1994; Chapman, 1996). You may also find state-of-the-art information in the yellow reports of the series "Monitoring Water Quality in the Future" (Adriaanse *et al*, 1995; Van Loon and Hermans, 1995; De Zwart, 1995; Tonkes *et al*, 1995; Groot and Villars,1995) and in the 1996-series of red reports by the before-mentioned Task Force (Breukel and Timmerman, 1996; Van Helmond *et al*, 1996; Knoben *et al*, 1995; Timmerman *et al*, 1996; Niederländer *et al*, 1996). All these reports preceded the drafting of the guidelines. Upon completion of these reports, we asked ourselves: Is it possible to draft strategic guidelines, to be used as a code of practice by riparian countries in transboundary river-basins, countries that signed the Convention? Are there guidelines that establish the principles for tailor-made monitoring?

I will not repeat the contents of the guidelines (ECE, 1996). You can read them. I will only mention some principles and basic elements.

First we have to realise that the information needs of today are not the information needs of tomorrow. That is because problems like political issues have a certain life cycle. In 1986, one of the former Dutch Ministers for the Environment, Pieter Winsemius wrote his political testament, "Guest in own house" (Winsemius, 1986). He introduced the policy life cycle (Figure 2).

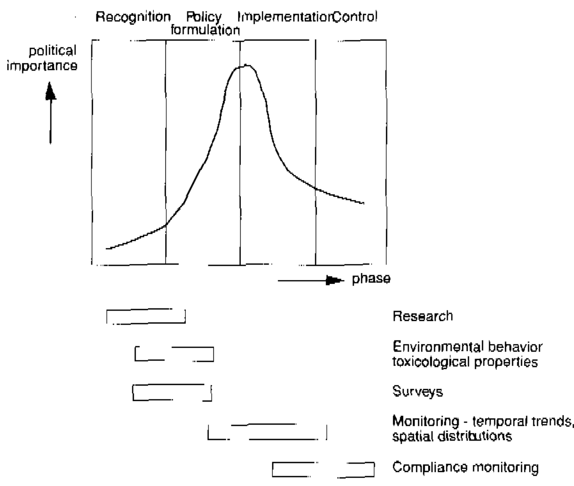


Figure 2: policy life cycle

Initially, there is no awareness of a problem. The first signs of a potential problem may come from the society (from researchers, eco-freaks, environmental organisations like Greenpeace, etc.) or from other countries. There is a period of discord and quarrel; people may even deny that there is a problem. After some time governmental organisations begin to recognise that the problem exists. The political importance increases and policies and regulations are prepared. After implementation of the policy measures, we reach the situation that theoretically the problem is under control. And after some time we may even see a period of deregulation. Information needs grow and diminish with their political importance. The character of the required infor-

mation and the monitoring activities will change completely in time (Winsemius, 1986). At first we need research, insight in the problem. What is our problem? What is the harm, what are the risks? What is happening to our water resources? We need in-depth surveys to become aware, and to analyse the problem.
From that insight we learn how the problem can be managed, and which information makes sense. *Objectives and standards for water quality are formulated and testing of compliance with* these standards becomes a routine activity. Thus different types of monitoring can be distinguished, which are given more or less emphasis during time:

Strategic monitoring
- status and trends
- preliminary surveys

Testing for compliance
- water quality standards
- discharge permits

Operational monitoring
- process control
- early warning

The second principle behind the guidelines is the monitoring cycle (Figure 3).

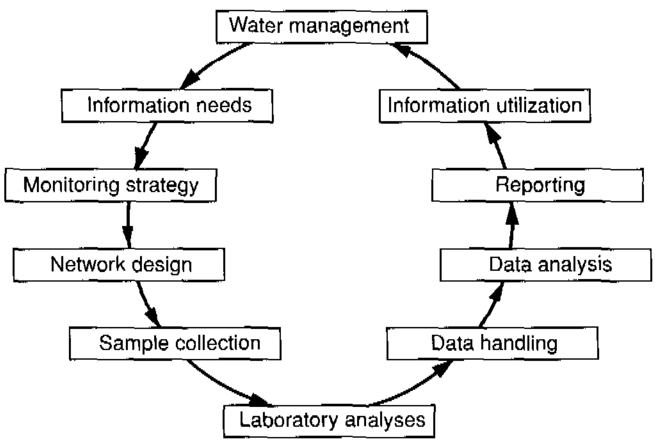


Figure 3: monitoring cycle

It is a guiding principle that the process of monitoring and assessment *should be seen as a* sequence of related activities that begin with the definition of information needs and ends with the use of the information product. Successive activities in this chain should be specified and designed on the basis of the required information product as well as on the preceding part of the chain. *You must realise that a chain is only as strong as its weakest link.* What are the weakest links? I will discuss this later. The top of the cycle is the water management. This means that the decision-maker decides what our information-product should be. What water use and water functions have priority, what are the problems and the threats in water management, what issues are decisive in formulating the water policy, what are the measures taken?
Information is for action, for use.
You will claim: The public must also have a say in what is important, publication of data via INTERNET for example. Members of the rafting club, or fishing club or other water users are eager for data on the INTERNET, and they will tell their representative in parliament that they

appreciate this information very much. To be elected again, their representative supports the continuation of funds for this data program. Obviously, (indirectly) the decision maker once again decides what should be our information product. He decides about the funding of our activities. Monitoring is part of a process dominated (that should be dominated) by the upper part of the cycle. Let me mention some basic elements in the guidelines.

a. To begin with, the awareness of the issues and information needs are key elements for the proper design of the monitoring programme, as was already explained before.

b. Indispensable insight in processes in river-basins requires in-depth investigations into processes by preliminary research, risk assessments, etc. It should become clear what are meaningful indicators for the specific river basin.

c. Determine an assessment strategy before the monitoring effort:
   - Realise that the status of a water body cannot be assessed by a set of physical-chemical parameters only. Apply integrated assessment by a triad of approaches, including physi cal-chemical analysis, ecotoxicological assessments and biological surveys.
   - Inventories should be made, including a general screening of all available information. What human activities take place in the region, in industries, in agriculture? What may be expected in the water? What are the hot-spots?
   - Perform preliminary surveys, including a screening of hot-spots in the water body and in the effluent discharges .
   - Make tiered, step wise approaches, going from course to fine assessments (Figure 4). Such tiered testing strategies can ultimately lead to a reduction in information demands for further routine monitoring.

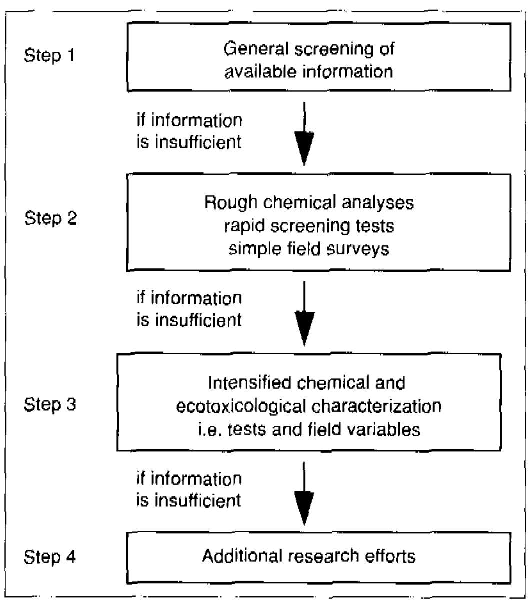


Figure 4: tiered approach

- Decide by what indicators or indexes the status of the waterbody could best be characte rised.

d. Quality management. Of course for the comparability of data the laboratory practices are important. We all are aware of significant inaccuracies in analytical results (Van Veen, 1990).

Parameter group	Surface water	Waste water	Sediments
Macro parameters	14	6	30
Suspended matter	24		
Metals (total)	41	20	17
PAH		58	51
PCB			60
OCP			96

Table: Comparability of analytical data

However, quality management should cover all elements of the monitoring cycle. The importance of employing protocols for sampling, data handling and data presentation should be mentioned.

e. Institutional aspects
Monitoring is not an area somewhere behind the real world. Information plays a role. This role is highly clear in a process of improving the protection and use of water. For the public to know what is going on for their daily use of water, monitoring is one of the players. The game has to be won. Monitoring has to perform well. But it cannot perform properly as long as the roles in the team, the coaching, the responsibilities, are vague.
Let me give some examples:

- It is essential that countries agree on: (1) the political decision that problems have to be managed on a river basin scale; (2) that river basins have to be used and protected in the context of transboundary cooperation; (3) that river basin authorities should set up river basin agreements; (4) that monitoring activities are coordinated on a regional or river basin scale.

- It is very important that the responsibilities in water management are clearly defined and implemented. Who is responsible for the water-quality planning of the region? Who is responsible for the enforcement of the plans? Who is responsible for the formulation of the information needs.

- It is important that countries improve outdated legislation to meet today's requirements for tailor-made assessment, taking into account aspects of an integrated watershed approach. I refer to such developments as the drafting of the Water Framework Directive of the European Union (European Commission, 1996) and to the World Bank study to reach recommendations for new Water Quality Standards and Objectives for the Russian Federation. Of course these developments may imply a struggle of many years.

Technical solutions for improvement of monitoring will fail, will be a waste of money, as long as the essential institutional arrangements have not been clearly established.

WEAKNESSES IN MONITORING

All elements of the monitoring process have their own pitfalls and traps, in other words, their specific weaknesses. We could easily write books about them. Quality management has to cope with them. Can we score the weakest spots in our monitoring activities? I know this is like comparing apples and oranges. However, I made an attempt to indicate the contributions in percentages to the ineffectiveness of the information product. Or, to put it more directly, the "contri butions to the misery" in information acquisition (Figure 5)

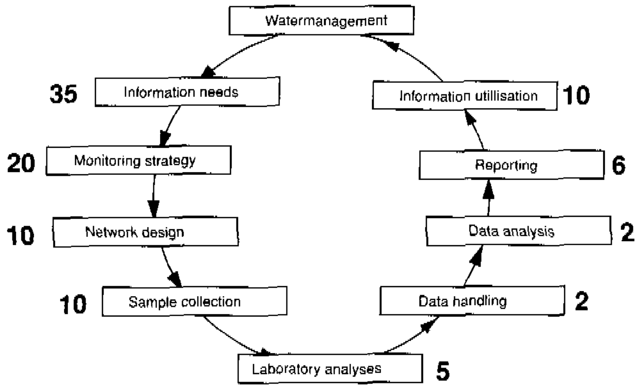
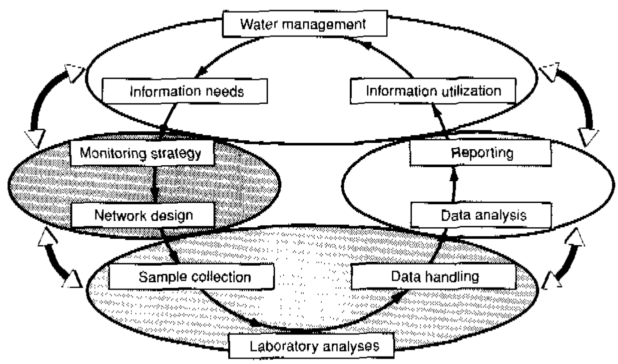


Figure 5: Contributions to ineffectiveness in information product

Of course it is subjective, but I feel that it may be indicative. I asked my colleague Peter Stoks to do the same. The results were comparable, only slightly different, and Figure 5 represents our mean values. I must say, that they apply to a situation that can be characterised as not-too-bad.



This means: reasonable availability of funds for monitoring, sufficient equipment, certified labo-

Figure 6: Worlds of expertise

ratories. We discussed this and felt that this is not the whole story. The ten elements of the monitoring process could be grouped in four parts, as shown in Figure 6.

In fact these parts are four separate worlds of expertise. Within one world activities are performed rather well, but between the fields there is restricted communication. Results of one world are thrown over the wall to the other world. Network designers often feel that within the field of water management and information needs there is much confusion and vagueness in specifying what information is really needed. Improvements in communication should be directed to the four indicated arrows.

CONCLUSIONS.

1. Tailor-made monitoring, first of all, aims to increase the effectiveness of information.
2. Secondly, tailor-made aims at efficient solutions in the given context.

3. Tailor-made monitoring and assessment can be guided by strategic principles, which can be formulated in a code of conduct, or guidelines.
4. Institutional aspects, like the lack of clear responsibilities in water management and water *policy, hamper tailor-making of monitoring considerably.*

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CONCLUSIONS AND RECOMMENDATIONS OF THE INTERNATIONAL WORKSHOP "MONITORING TAILOR-MADE II - INFORMATION STRATEGIES IN WATER MANAGEMENT", NUNSPEET, THE NETHERLANDS, 9-12 SEPTEMBER 1996.

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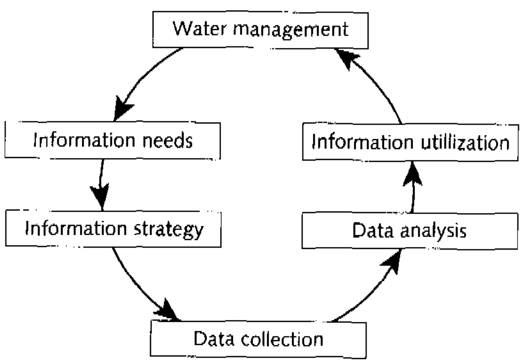
CONCLUSIONS

1. An increasing demand for methodologies to enable the specification of information needs as well as the transfer to the actual users of information resulting from monitoring is observed. Also the idea, that monitoring without specification of information needs prior to the actual network design will be a waste of money, is becoming common knowledge. This development has been supported by the results of the workshop Monitoring Tailor-made I and these ideas are currently finding their way into practice as we can conclude from the presented papers during Monitoring Tailor-made II. However, it appears to be difficult to deal with the upper "information parts" of the monitoring cycle (Figure), since methodologies are lacking.



2. Other aspects/factors are also forcing us to make better use of our monitoring information. Many organisations in the field of monitoring have to deal with budget cuts and the cost-effectiveness of our monitoring activities is becoming more important. From the presentations we can also conclude that a growing number of countries embrace the tailor-made approach to water monitoring.

3. We could also conclude that to be successful in tailoring monitoring networks towards information needs, a growing number of disciplines will be involved. Not only monitoring engineers will work on the network design, but also information analysts, economists, sociologists and specialists in visual presentations can be involved. This creates new challenges, especially for developed countries. However, new problems will arise also, since communication between these different specialists will become more complex.

4. It is concluded, that there is no universal monitoring programme or even a universal strategy to develop monitoring networks. Especially when problems are severe and numerous, priorities

will not allow an extensive study on the specification of information need before starting to measure anything at all.

5. Another conclusion from this workshop is a tendency to shift from retrospective to predictive monitoring. This includes for example the special purpose monitoring of substances from resource to supply.

6. Within the field of monitoring strategies and network design, integrated monitoring is a rising star. Especially the combination of chemical and ecotoxicological monitoring will give the resulting information an added value. The development of indicators or indices within this field enables a streamlining of information, which will reduce costs. These are also very useful in the quantification, presentation and communication of information. Ultimately, indicators could be used to direct or adapt monitoring activities to new, arising issues. Monitoring cannot anymore be seen as an isolated activity, but also other sources of information have to be included (e.g.: socio-economic data). The use of models will enable a dynamic combination of the different aspects of environmental quality and these can also be used to change retrospective monitoring towards predictive monitoring.

7. On data management and presentation we conclude that the growing public awareness on environmental issues has resulted in a responsibility of monitoring organisations and governments to communicate the collected environmental information to the public. This could be a presentation of understandable information on Internet in a simple and meaningful way. Simple indices or ratings will help, like the use of a kind of Dow-Jones index for monitoring information. It is also recognised that these indices should not be too complex. It is not useful to combine all aspects in a statistical sound way, to fulfil this responsibility towards the public.

8. Also, the responsibility is shifting on one hand from national to local authorities and on the other hand from national to multicountry organisations (e.g.: EEA). The latter facilitates the river basin approach in the fight against water pollution. Still, both local and multicountry approaches have to be tuned somehow

9. Most remarkable results gave the combination of groundwater and surface water specialists at this workshop. For groundwater the same concepts are applicable as in surface water monitoring. Monitoring activities of groundwater also closely follow the already mentioned "monitoring cycle". Especially the information parts of the cycle as "specification of information needs" and information use are fairly similar for both groundwater and surface water monitoring networks. Differences are mainly due to physical characteristics and thus become apparent on an operational level. Therefore, it is encouraged to co-operate when specifying information needs for both groundwater and surface water monitoring. However, the actual monitoring programmes should be executed separately in most cases.

10. Presentations during this workshop gave an opening in finding methodologies for the specification of information needs. For a useful specification, policy designers and monitoring specialists should meet and a forum for regular communication should be established. Another prerequisite in successfully defining information needs will be to include institutional aspects in monitoring agreements. Another development in order to make valid analyses of the state of the environment throughout river basins possible is the evolution from general standards to more differentiated quality standards (e.g.: EU-Framework Directive).

RECOMMENDATIONS

The workshop led to the following recommendations:

1. A stronger participation of policy designers and public in meetings like this workshop will improve the resulting monitoring information itself and also communication with the public.

2. The involvement of other (than usual/traditional) disciplines is a prerequisite for a sound specification of information needs, for an up-to-date monitoring strategy that can include information needs arising from new issues, and for a responsible way of managing data and resulting information.

3. To improve quality of monitoring activities, the establishment of an international monitoring forum will open a way to make this rather complex communication between different disciplines and organisations possible. It will also increase the transparency for the public of measures taken, since it will become more clear on what basis (monitoring information) these measures are taken.

4. More effort should be put in research or development of methodologies to specify information needs and the development of transparent meaningful indicators/indices. Also a decision tree should be developed, which could be based on the specific priorities of a country or river basin and that will result in different levels of complexity of the designed monitoring network.

The above conclusions and recommendations have been drafted with the help of the chairpersons of the different sessions. Input was also received from authors contributing papers and posters, and from those participating in the plenary discussions.

DESIGN OF A GROUNDWATER-MONITORING-NETWORK IN THE NEW FEDERAL STATES OF GERMANY

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ABSTRACT

Groundwater protection is one of the most elementary tasks of any state, especially when its groundwater resources provide most of the drinking water. The groundwater monitoring system was designed in a way that information can be gathered on groundwater quality and trends at any time. The system provides evidence on the scope and causes of variations of groundwater quality. Based on typical key parameters groundwater quality was allocated to different types of impact.

STARTING POSITION AND TARGETS

The Federal Environment Agency, under the environmental research programme of the Federal Ministry for the Environment, Nature Conservation and Nuclear Safety, launched a research project in the New Federal States to develop a groundwater monitoring network. The project was undertaken jointly by the environmental agencies of Mecklenburg-Western Pomerania, Brandenburg, Saxony-Anhalt, Thuringia and Saxony in association with the Gesellschaft für Umwelt und Wirtschaftsgeologie mbH (UWG). In the former GDR the implementation of a country-wide groundwater monitoring was planed but not realised due to the political change in Germany. The monitoring methodology conceived in the former GDR and the concepts proposed by the Länder Arbeitsgemeinschaft Wasser (LAWA) for the old federal states revealed a large measure of agreement. In close association with the authorities in charge of groundwater monitoring of the New Federal States the existing concept was updated.

INVESTIGATIVE CONCEPT

The investigative concept is based on the idea that it should provide a representative survey of groundwater quality. Existing pollution loads and their causes have to be determined. The research project focused on the following objectives:

1. Provision of a survey of groundwater quality in the New Federal States based on comparable and reproducible analytical data;

2. The project calls for investigations designed in delimiting or more closely defining uniform hydrogeological regions and in deriving reference data suited to identify anthropogenetically conditioned variations and to fix target parameters for protective or remedial measures.

3. Enhancing the precision of the measuring network by functional testing of the integrated measuring points, supplementing measuring point master data and opening additional representative measuring points.

4. *Enhancing the precision of the measuring programme in terms of type and number of parameters*, frequency of measurements and measures to improve internal and external qualicontrol.

FOUNDATIONS

The research programme covered the period from March 1992 till July 1995. Groundwater sampling was scheduled for spring and for autumn. The federal states involved came to an agreement about the catalogue of substances to be analysed, the methods of analysis and criteria for the selection of measuring points. Measuring points found to be in poor technical condition were removed from the network. During the period of investigation the number of measuring points increased from 215 to 360.

RESULTS OF INVESTIGATIONS FOR THE PERIOD 1992 - 1994

During that short period of investigation the hydrochemical status of groundwater did not tend to change significantly. It is mainly iron and manganese, which exceed the limit values laid down in the Drinking Water Ordinance. The portion of nitrate-concentrations, exceeding the limit value, remained comparatively stable at between 3 % (spring 1992) and 9 % (autumn 1994). Ammonium is registered at a constant rate of approx. 20 % of the investigated groundwater bodies above the limit values. pH-values below 6.5 were frequently encountered during the entire period under review (25 % in spring 1992, 21 % in autumn 1994).

groundwaterimpacts	*identified by concentration-anomalies of the following parameters*
salinization	conductivity, chloride, sodium, sulphate, potassium, AOX
agricultural influence	nitrogen-components, potassium, sulphate, chloride, /potassium/sodium
pesticide-impact	nitrogen-components + pesticides
waste-water-impact	sulphate, sodium, phosphate, nitrogen-components, borate, AOX, oxygen demand, chlo ride, metals
atmospheric dust impact	calcium, sulphate, trace elements
acidification	pH, aluminium
industrial influence	AOX, PAC, HVCH, pesticides
diffuse influence	influences not strictly to identify (mostly salinization or waste water influences)

Tabel 1: Evaluation scheme to categorise main impacts to groundwaters

For the evaluation of the hydrochemical data collected an evaluation scheme was designed. A certain combination of anomalous concentrations of typical key parameters (Tab. 1) is used for the allocation to one of the following types of impact: salinization, agricultural influence, pesticide-impact, waste-water impact, atmospheric dust impact, acidification, industrial influence and diffuse influence. On the basis of these criteria 1803 analyses were allocated to a certain type of impact.

Figure 1 shows the distributive pattern as reflected in the data set. It has to be mentioned that there are in part considerable state-to-state differences in terms of the share of unaffected groundwater bodies and the importance of different types of impacts. Often this is directly related to differing natural hydrogeological conditions between the states concerned.

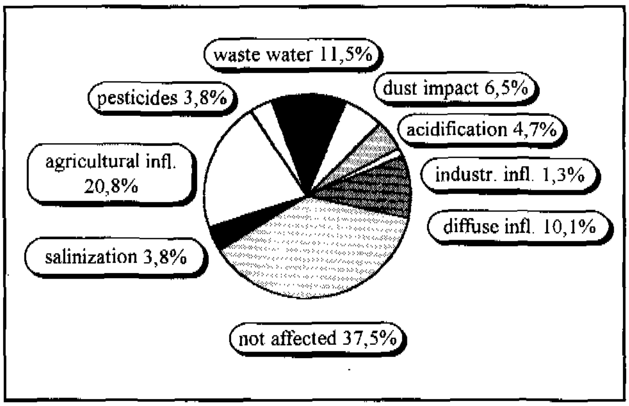

Figure 1: Relative share of groundwater impacts for all analyses

CONCLUSIONS

1. Future investigations should include the determination of groundwater age, relative discharge components, groundwater flow velocity and resultant discharge delays.

2. Measuring points bearing heavy anthropogenetic loads should be eliminated from the network. Measuring points registering marked loads resulting from specific impacts should be included in special-purpose measuring networks.

3. There should be greater flexibility in terms of frequency of analysis and range of parameters.

4. Measurable concentrations of anthropogenetically conditioned groundwater substances (NO3, SO4, Al) are very much subject to meteorological and hydrological influences. They need to be sampled more often than twice a year.

5. A method of dealing with measuring data below detection limits is needed.

6. Close links should be established to the meteorological and hydrological as well as the specific hydrodynamic conditions of the aquifer concerned and taken into consideration for the interpretations of groundwater quality data.

7. The use of lithofacial units permits the determination of anthropogenetic impacts on a given groundwater quality in consolidated rock formations.

DESIGN OF A STATE GROUNDWATER NETWORK

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ABSTRACT

For building-up its groundwater quality network the state of Baden-Württemberg decided to aim not only at some overall information but

- *to get reliable estimates about the effect of certain protection measures, and*
- *to get a representative picture about the extent of effects from different kinds of land use.*

To satisfy these requirements 3.300 measuring points are necessary, of which 2.700 could be probed in 1995. This was possible by cooperation with many users of groundwater and by use of existing wells etc. only. The network is operated on a yearly basis. Extensive quality assurance measures, the build-up of a flexible data base and of a reporting procedure, which guarantees actual information, were further prerequisites.

However, at present, the resources allocated are cut back. Therefore, the design of the network had to be optimized again. The number of stations was not changed in this process, since it is the opinion, that the spatial variability still is larger then temporal variability and errors from sampling and analytics. Instead, the frequency of measurements, the number of parameters analysed and the effort spend in operational measures (quality assurance, quality improvement, data mangement facilities etc.) had to be reduced.

INTRODUCTION

The state of Baden-Württemberg depends to approx. 75% on ground- and spring water for its public drinking water consumption of 706 Mio m3 (1995). An additional 181 Mio m3 from groundwater is used by industry. Recognizing the vulnerability of the resource groundwater, which was previously considered to be well protected, the state has implemented an "groundwater protection program". Part of this program is a groundwater monitoring network.

The main goals of this network are (LfU, 1996a)

- to ensure compliance with laws and regulations
- to monitor the state and possible developments of the resource

The first goal has to be dealt with on a local basis, depending on the characteristics of the particular installation and the hydrogeological setting. This job is taken over by local agencies, which operate local observation- or early warning wells as necessary.

The second goal is realized by a state network. To ensure an environmentally sound usage and management of the groundwater resource this network has to

- monitor not only the quantity but also the quality continuously
- identify processes which influence the groundwater
- control the effects of protective measures and regulations.

In the beginning of the build-up of the network, the strategic decision had to be made, whether to concentrate on a few pilot areas or study units as, for example, in the USGS "National Water Quality Assessment Program" (Hirsch et al., 1988) or to try to cover the total area of the state.

Based to a certain extent on political reasoning the second approach was chosen.
In order to design an appropriate network the financial and personal resources have to be properly distributed between four cornerstones: network density (rsp. number of stations) - measurement frequency - parameter list - quality assurance and operational measures.

The factors influencing this allocation are

- consequences from the definition of goals
- the ratio of large and small scale spatial as well as temporal variations in relation to errors of measurement
- pragmatic points of view as usage of existing facilities and data.

The major factor certainly is the definition of the goal. This point can be quantified by 3 "levels of information":

- overall description for the entire state by a few hundred stations
- a representative description, enabling some regional differentiation and statistical evaluation of processes, effects of regulatory actions etc.
- detailled and nearly complete surveillance.

The design of the networks corresponding to each of these levels and procedures to establish them will be different. For the main state network the second level is expedient.

DESIGN CONSIDERATIONS

The scales of variation to be considered range from between-aquifer over in-aquifer-variations to spatial variations within areas of, for example, a few tens of square kilometers. For a first estimate on large scale spatial variability in Baden-Württemberg 42 aquifers (29 without mineralized groundwater) have been delineated according to the geological situation. A part of these aquifers have been aggregated to "groundwater provinces" (GLA/LfU, 1985, see Appendix 1).
Temporal variations considered range from year-to-year fluctuations down to the variations in continuously recorded quality data from springs.
These variations have to be compared with the magnitude of measurement errors. Such errors can result from improper sampling, handling, analysis, data transmission, etc. The analytical and sampling errors are quantified by results from multiple sampling, a special quality assurance program etc. A comparison of the different types of variation show the dominance of spatial variations. Examples are given in Figures 1 to 4.

Fig. 1 shows for data from the basic network and 4 groundwater provinces, that the temporal variations at the measuring locations over a period of 6 years (CVt) is considerably smaller than the spatial variation between the mean values (CVr) for the same wells.

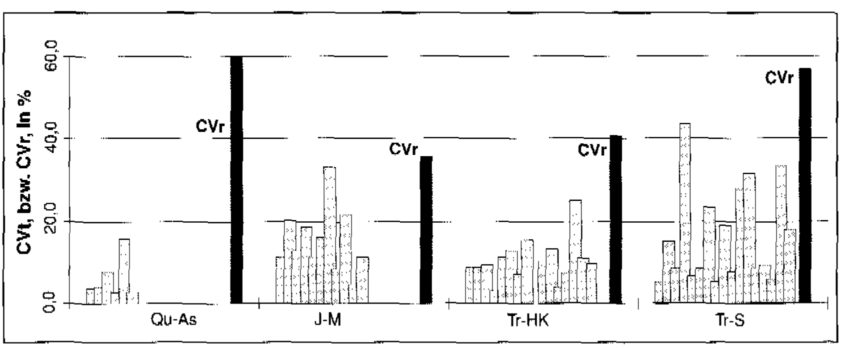


Figure 1: *Temporal variability at individual wells (hatched) and spatial variability within selected groundwater provinces (full)* (Data: Nitrate, BMN, 1985/91, Abbreviations for groundwater provinces see Appendix 1)

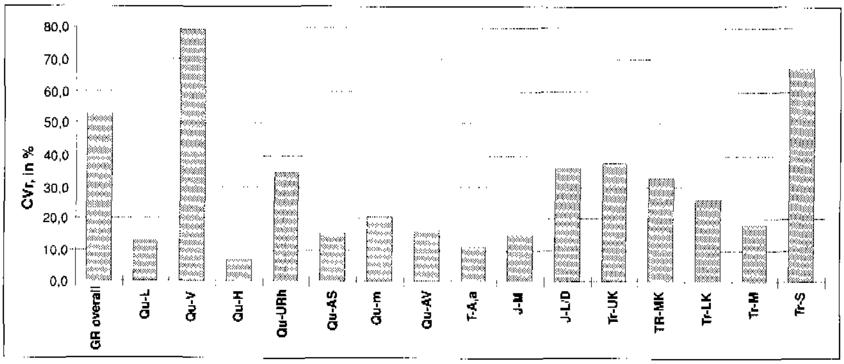


Figure 2: Spatial variability within groundwater provinces (Data: Total hardness, statewide screening network (GR), abbreviations for groundwater provinces see App. 1)

For each water constituent the spatial variability within the groundwater provinces is rather different (Fig. 2), and the pattern between the provinces is different. Similarly, the degree of overall spatial variability for different constituents varies over a wide range (Fig. 3).
However, generally the amount of spatial variability is largest as compared to the temporal variability but also compared to (except extraordinary) errors of sampling and analysis (Fig. 4).

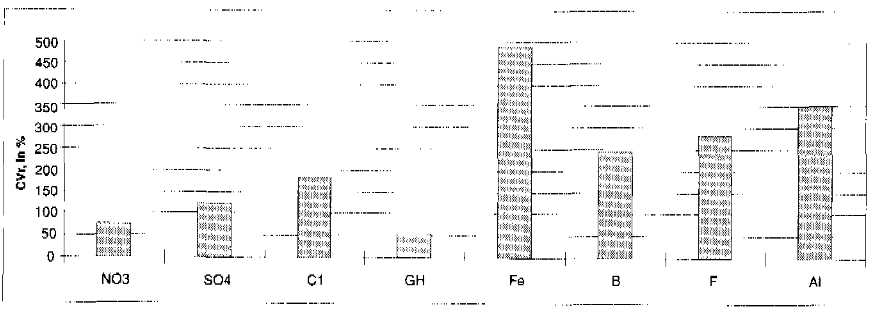
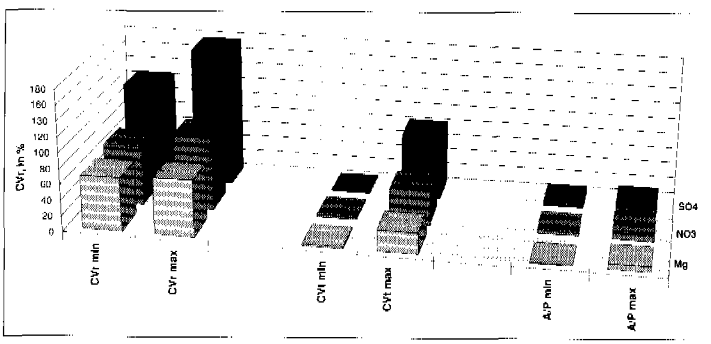


Figure 3: Overall spatial variability, different constituents (Data: Statewide Screening network (GR), 1918/93, means



over 15 groundwater provinces)

IHW-2

Figure 4: Spatial (CVr), temporal (CVt) coefficients of variation and error of sampling and analytics (A/P)

The more technical considerations have to be paralleled by organizational points of view. For example, a basic decision was whether to build new stations as perfectly as possible or to use as much of existing facilities and data as possible. Since Baden-Württemberg choose the second way, a large number of individuals and institutions participate in the program. This results in a large organizational effort in the build-up and operation of the network.

Additionally, at present it is rather possible to get additional monetary instead of personal resources. This has resulted in the participation of approx. 30 laboratories and other private contractors for the operation of the network and especially for sampling and analytics. Therefore comprehensive measures have to be taken to ensure a sufficient quality standard of the measurements. Part of the relevant procedures are a regular laboratory quality assurance program, detailled technical rules of operation, training courses for sampling etc.

REALIZATION OF THE NETWORK

The above considerations, the availability of resources and the decision, to get not only an exemplary or overall description of groundwater quality but also to gain some control on the effects of regulatory actions lead to a quality network which is subdivided into:

- a **"basic network"** to determine the anthropologically by local effects possibly not influenced groundwater quality (113 locations)
- a network to survey the quality of water used for **drinking water** supply in connection with **early warning** wells (685+70 locations)
- a **supporting** network to give a representative picture of the groundwater under different kinds of land use (1.600 locations)

A special supplement is a spring network (200 locations) to monitor the water quality in karstic and cristallin aquifers.

Overal the network consists at present of approximately 2700 stations, that is 1 observation point per 13 km2 on the average. This large number of stations can be realized by cooperation with different partners only. The frequency of measurements lies between once and six times per year, with a few exemptions. The complete network is probed within 6 to 8 weeks during September/October. The parameter list was substantially modified in order to describe the state and possible developments over a period of 5 years only (groups P1 to P5, see App. 2) and to satisfy the special objective of different controlling programs (V, S, B, see App.2).

Although there is much effort spend in data collection the need for integration of **additional data** is recognized. Such data are needed for proper interpretation of the measurements.

Additional data to be used are

- information about groundwater levels, results from quantitative groundwater model investigations about flow directions, travel times etc.
- data about drinking water quality collected by the health administration and the state statistics bureau
- hydrogeological data, air quality data, soil data, data from contaminated sites.

First of all, data about the quantitative aspects of groundwater are required. Therefore the long time existing quantitative networks will - after their optimization - be integrated. This implies simi-

lar rules of operation but also a joining of data processing. After integration the groundwater network will consist of four primary components: groundwater level - groundwater quality - springs - lysimeter/precipitation.
A very important aspect to be included in any evaluation of quality sampling is the information to be gained from quantitative groundwater models. This implies for example information about groundwater flow velocities and direction or water balances.

A particular example for additional data are soil nitrogen data, which are collected on a regular basis from approx. 70 000 locations yearly in order to control compliance with a state regulation to reduce the nitrate input from agricultural practices into groundwater. These data are used to evaluate anticipated developments of nitrate concentration in groundwater.

The data exchange with relevant agencies is to be established on a routine basis.

However, regardless of how careful such a network is designed, it will not be successful without an adequate reporting. At present, the reporting in Baden-Württemberg is aimed mainly at the public. This requires quick, attractive but perhaps somewhat simplified reports. That requirement is fulfilled at present by yearly state-of-the-resource reports, appearing approx. 3-4 months after the end of the sampling campaign (e.g. LfU, 1996b), and different short information notes about particular aspects.
Nevertheless, as data accumulate, it will be possible to present more detailled evaluations. First examples are recent reports about the geogene groundwater quality, acidification in groundwater and about the effects of the mentioned regulation for the reduction of nitrate input originating from agriculture.

CONCLUSIONS

If the goals of the network require early detection of trends with a predetermined degree of significance and an evaluation of the effectiveness of protective measures a considerable number of measuring points, frequent measurements and a large parameter spectrum have to be used. Due to monetary restrictions the Baden-Württemberg groundwater quality network had to be optimized only three years after reaching full operation. Because of the dominance of spatial variability it was decided to retain the number of measuring points and to stretch the sampling periods instead. Also the parameters were reduced to only those of immediate relevance.

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APPENDIX 1: Groundwater provinces Baden-Württemberg (GLA/LfU,1985)

Abbreviation	
Qu-L	Quaternary-Loess/Rhine Graben Rim
Qu-V	Quaternary-Glacial Sand and Gravel/Valley Fills
Qu-H	Quaternary-Hillside Debris
Qu-URh	Quaternary-Glacial Sand and Gravel/Upper Rhine Valley
Qu-AS	Quaternary-Glacial Sand and Gravel/Suebian Alb South Rim
Qu-m	Quaternary-Glacial Sand and Gravel, Tillcovered
Qu-AV	Quaternary-Glacial Sand and Gravel/Alpine Foreland
T-A,A	Tertiary / Suebian Alb South Rim, Alpine Foreland
J-M	Upper Jurassic, Malm Foremation
J-L/D	Lower & Middle Jurassic, Lias & Dogger Formation
Tr-HK	Upper Triassic, Upper Keuper Formation
Tr-GK	Upper Triassic,Middle Keuper Formation
Tr-LK	Upper Triassic, Lower Keuper Formation
Tr-M	Middle Triassic, Muschelkalk Formation
Tr-S	Lower Triassic, Buntsandstein Formation

APPENDIX 2: Parameter Groups

Abbr.	Name List of Parameters	
G1 (P1)	Reduced Basis Program	Alkalinity, Total hardness, DOC, Ca, Mg, Na, K, NH4, Fe total, C1, NO3, NO2, SO4, B
P2	Industry/Solvents	VOC, BTXE, dissolved or emulsified hydrocarbons, NO3
P3	Agriculture	NO3, N-Herbicides
P4	Urban	Alkalinity, Total hardness, NO3, C1, SO4, K, B, As, Cd
P5	Organics	NO3, ad-hoc list
VO	On-Site	Discharge, Color, Turbidity, Odeur, Temperature, Conductivity, pH, O2, O2-saturation
V	Acidification	VO + special program
S	Controlling Program Fertilisation Reduction	VO + NO3. NO2, NH4
B	Balancing	Discharge, individual list per station

THE GEOHYDROCHEMICAL APPROACH IN OPTIMIZING GROUNDWATER QUALITY MONITORING

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ABSTRACT

Systems for groundwater quality monitoring should be primarily based on geohydrochemical information and expert judgement. In reality one is often faced with a rather erratic network of wells with a too unspecific monitoring program. With the geohydrochemical approach proposed here, costs are minimized and the benefits are maximized.
The approach consists of: (a) a geohydrochemical screening procedure for analysis on expensive trace elements and organic micropollutants; (b) an advanced hydrochemical mapping technique for selecting the right observation wells and tailoring both the analytical program and frequency of sampling; and (c) chemical analysis of reactive solid phases in the aquifer matrix for further tuning the monitoring program.

INTRODUCTION

There are three major issues of groundwater quality monitoring:

a. prediction of future groundwater quality as a source of drinking water, in order to anticipate in (1) the abatement of ongoing deteriorating processes and/or (2) the implementation of additional purification techniques;

b. environmental protection: water quality is the most economic, sensitive and relevant pollution sensor; and

c. independent validation and calibration of hydrological flow models, through chemical groundwater dating and the chemical visualisation of flow paths.

The first two are generally accepted (Baggelaar, 1992). The latter is gaining in importance as it increasingly realized that hydrological flow models, being the basis of hydrochemical predictions, are inaccurate without hydrochemical validation and calibration.
Traditionally, systems for groundwater quality monitoring evolved from casual problems and the decision of hydrological and chemical engineers, while occasionally a statistical analysis was made to reduce the oversized monitoring program. Too often a 'waste of money' feeling arises, especially where very expensive analytical data are frequently below the detection limit (Fig.1). It is about time that, besides the hydrological, chemical and statistical approaches, a geohydrochemical approach is admitted for optimizing groundwater quality monitoring.

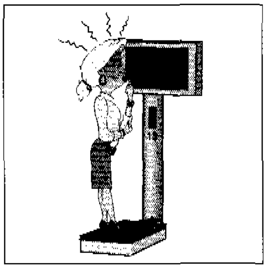

Figure 1: The 'waste of money' feeling, also pertaining to most systems of groundwater quality monitoring. Prevent a database from loading with too many expensive analytical data below either an acceptable or, worse, a too high detection limit.

THE GEOHYDROCHEMICAL APPROACH

The geohydrochemical approach consists of the following 3 steps:

1. application of a geohydrochemical screening procedure for the more expensive analyses, like those on trace elements and organic micropollutants. The procedure, which is based on experiences with groundwater in many geohydrochemical environments, is described in Table 1;

Well condition	do not analyze for TEs# and OMs@ in recently drilled wells, wait at least 3 months (bailer drilled) or 12 months (uncased drilling methods).
Origin (detection by Cl^-, $\delta^{18}O$, EJ*)	analyze specific OMs@ and TEs# only in specific waters (Na-dikegulac in Rhine bank filtrate; Ag, Hg, Mo, Pb, Se, Sn, U in leachate of mine tailings)
pH	if pH>6, do not analyze for Al, Be, Lanthanides, Sc, Ti.
Redox class (by O_2, NO_3, SO_4, CH_4, EJ*)	if sulphate reducing or methanogenic, do not analyze for: Ag, Cd, Co, Cu, Hg, Ni, Pb, Se, Zn, AOX/VOX, chloro-alkanes and chloro-alkenes.
Age (detection by ^3H, ^{14}C, Cl, CFCs, EJ*)	if pre-industrial (>150 y), do not analyze for OMs@; if pre-tritium (>40 y), do not analyze for recently developed OMs@ like CFCs.
AOX and VOX	if below detection, do not analyze for OMs@ without specific reasons.
The most mobile persistent xenobiotics	if not present do not analyse for the less mobiles or less persistent
The most mobile heavy metals	ditto. If for instance Ni insignificant (EJ*), do not analyze for Pb, Cu, Zn, Cd.
Current minimum detection limit (MDL)	if too high do not analyze. For instance a MDL of 1 µg/l for Ag, Be, Cr, Hg, Pb, Sb, Se, and Sn is too high in 95%
Geohydrochemical twins	analyze for the difficult-to-analyze ion only if the easy-to-analyze ion is significant (EJ*): As-Sb, Ca-Sr, Cl-Br, Hf-Zr, K-Rb, Na-Li, PO_4-As, SiO_2-Ge, Zn-Cd.
Geohydrochemical antagonists	ditto, however if insignificant (EJ*): Ca-F, Fe(II)-O_2, NO_3-Fe(II), O_2-CH_4, OH-Al

: TEs = Trace Elements; @ : OMs = Organic Microcontaminants; * : EJ = by Expert Judgement

Table 1: Geohydrochemical screening program for groundwater analysis, in sequential order. Skip useless analyses before you pay for them, but add those indispensable analyses which cannot be acquired any more when the delayed evaluation starts.

2. evaluation of the present groundwater quality by hydrochemical mapping, in order to select the most representative observation wells and to tailor both the analytical program and frequency of sampling. An example is presented in Fig.2, using the mapping method of HYdrochemical Facies Analysis (HYFA) as proposed by Stuyfzand (1993; 1994); and

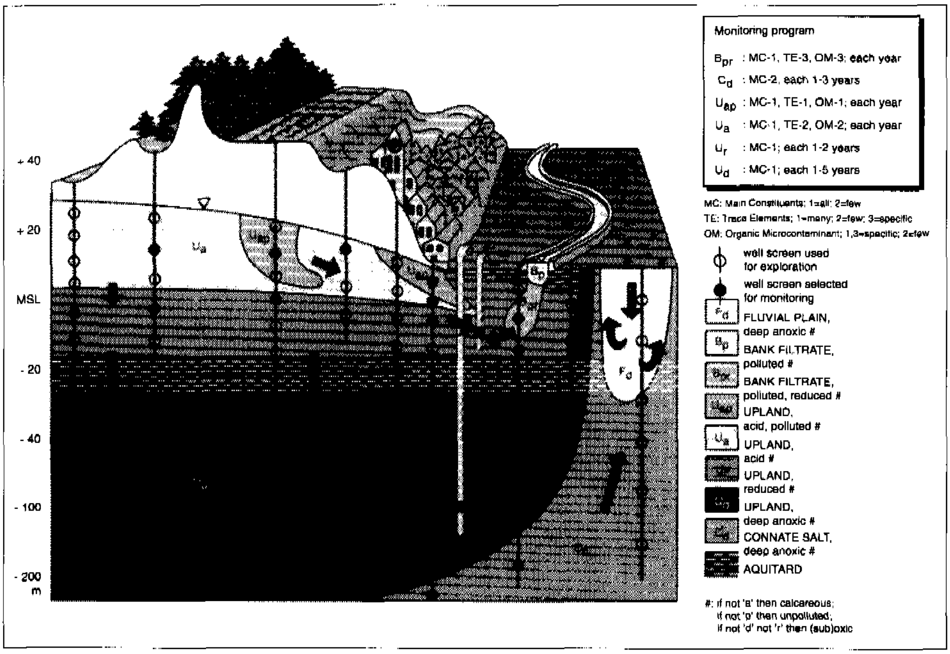

Figure 2: Hydrochemical mapping by HYdrochemical Facies Analysis (HYFA) as a solid basis for structuring the monitoring program. First hydrochemical maps need to be constructed using exploratory multilevel monitoring wells. They display the areal extent of groundwater bodies (differing in origin) and hydrochemical facies within (zones differing in degree of pollution, redox, calcite saturation etc.). Subsequently the best monitoring sites, specific analytical programs and sampling frequencies can be adapted to expectations, by combining these maps with groundwater flow maps and application of Table 1.
During monitoring, changes in origin (for instance from Upland water to Bank filtrate) or facies (for instance from 'reduced' to 'acid') in a well need to be followed by adaptations of its monitoring program.

3. quantification of the reactive solid phases in the aquifer, in order to anticipate in the most probable quality changes (thereby focusing on the relevant parameters) and the time scale involved (thereby tuning the frequency of sampling). An example is elaborated in Fig.3.

It should be mentioned that 'screening' may affect the normal sample routing in the laboratory, and may be less effective where multi-element analysis is applied without additional costs.

CONCLUSIONS

In financial terms, early investments in geohydrochemical research (including additional well installation, water and soil sampling, analysis and expert judgement) are earned back by a net lowering of the whole exploitation costs of groundwater quality monitoring (Fig.5). Besides financial profit there are other gains: the hydrochemical data acquired are more useful, the geochemical data are indispensable in predicting future groundwater quality, and the expert report on the spatial distribution of watertypes is needed to evaluate the size of high quality groundwater reserves and the importance of the various threats. It contributes to the understanding of the process of groundwater quality deterioration, without which the right abatement strategy cannot be selected, as in case of salinization (Fig.4).

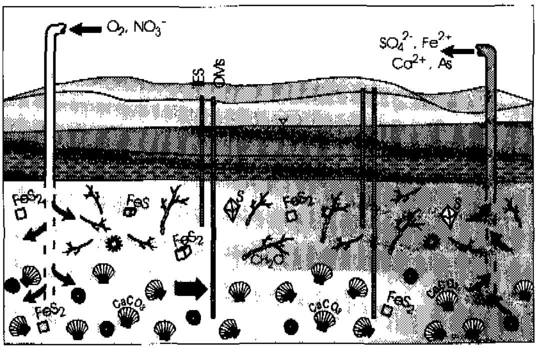

Figure 3: Geochemical inspection of the aquifer matrix can help to further tune the monitoring program. In this example, oxygenated water is injected through a deep well in an anoxic aquifer with geochemical stratification. The upper zone requires intensive monitoring of trace elements and a low frequency monitoring of organic microcontaminants, the lower zone requires the opposite.
The decalcified upper formation contains much organic material and pyrites which are being oxidized without acid buffering. This yields large amounts of dissolved trace elements, and a strong delay or decomposition of the organic microcontaminants injected. The calcareous lower formation is very poor in organic material and low in pyrites. The acid generated upon pyrite oxidation, is well buffered here and trace elements are hardly mobilized. The organic microcontaminants are hardly delayed (lack of organic sorbent) and many of them may resist decomposition as soon as the redox buffer pyrite is depleted.

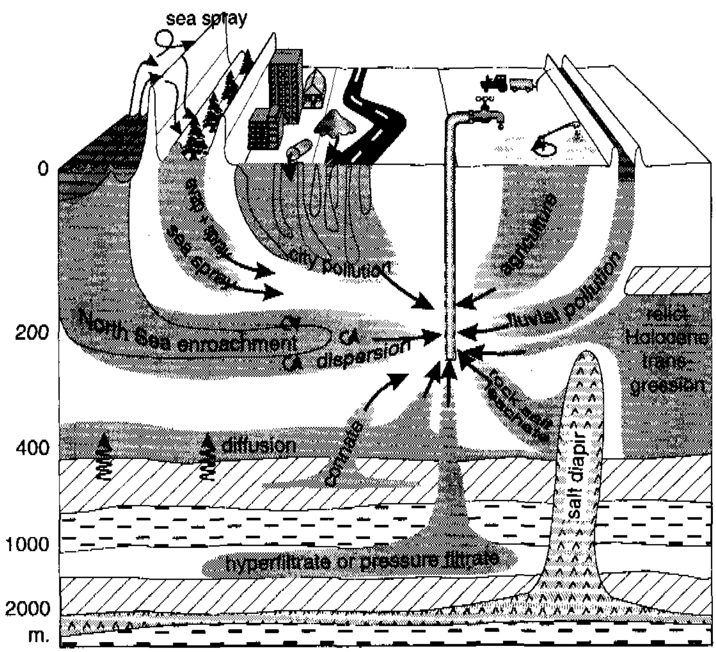

Figure 4: *The abatement of a rising chlorinity of raw groundwater requires the salinizing process to be identified. And without knowing this process the ultimate chlorinity cannot be predicted.*

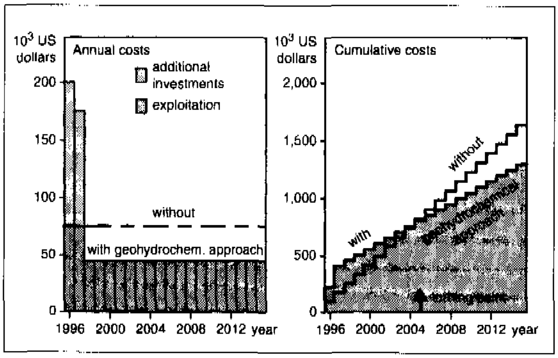

Figure 5: Early financial investments in geohydrochemical research are easily earned back after ca. 10 years of exploitation of the monitoring network, mainly through: (1) a lower number of sampling points due to a well-defined representativity of each point; (2) restriction of the extensive analytical program including organic microcontaminants and trace elements to selected sampling points with a raised chance to catch them; and (3) a reduction of the sampling frequency for sluggishly reacting wells.

The case shown, pertains to a 5 Mm3/y well field for drinking water supply, composed of 10 shallow and 15 deep pumping wells. The additional investments (ca. 225,000 US dollars in 2 years) cover expenses connected with the installation of additional observation wells, geochemical analyses of aquifer samples, hydrochemical analyses and an expert report on HYFA including a design of the monitoring network and program.

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DESCRIPTION OF THE MOZART GROUNDWATER MODEL

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INTRODUCTION

Knowledge about the behaviour of deep groundwater and the relation between surface water and groundwater is very important when tackling environmental problems such as excessive fertilization and desiccation. The Institute for Inland Water Management and Waste Water Treatment uses the MOZART groundwater model (model for the unsaturated zone for national analyses and regional applications) to describe the hydrological processes in the unsaturated zone (the shallow groundwater). MOZART plays a key part in the computational framework which describes the complete water system and consists of various computer models (figure 1). MOZART is especially designed to describe impacts of water distribution policies not only on a national scale but also on a regional scale. It is a flexibel model, which can easily communnicate with other models. In order to be able to relate groundwater management and surface water management to each other as part of integral water management, the model has been coupled to a model for deep groundwater (NAGROM) through an interface facility MONA. MOZART provides the relevant hydrological data for the models that predict the effect on agriculture (AGRICOM), ecology (DEMNAT) and waterquality (ANIMO).

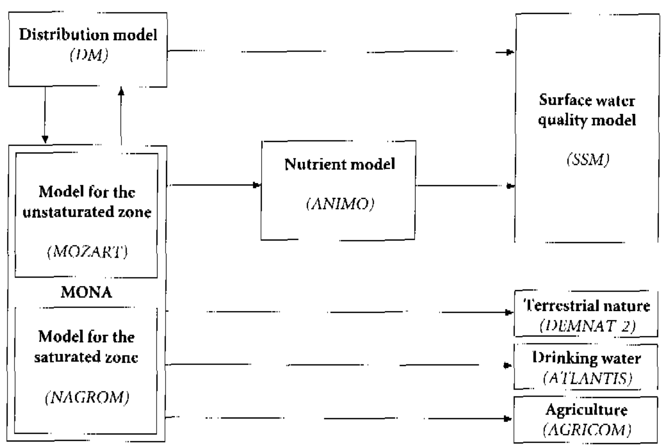


Figure 1: Computational framework

DESCRIPTION OF THE MOZART MODEL

MODEL CONCEPT

MOZART simulates one-dimensional vertical water transport in the unsaturated zone for unique hydrological units (plots). A plot is a one-dimensional system with unique values for evaporation, soil physics and geohydrology. A plot consists of an effective root-zone and a subsoil (figure 2). In each plot, the soil is subdivided into an effective root-zone and a subsoil, which can subsequently be divided into segments with specific soil-physical characteristics for each segment. A horizontal and a vertical flux and the moisture content can be calculated per segment. The interaction between groundwater and surface water is described by (broken) linear relations between drainage fluxes and the groundwater table. Each drainage system is presented by a line segment in the relation (figure 3). The slope of the three sections reflect the drainage resistances; the intersections of the segment reflect the drainage levels of the systems. The lower boundary condition is given by the seepage flux between the first aquifer and the plot. For the calculation method, there is the choice between a semi-stationary method (on a decade basis) and a numerical non-stationary calculation scheme or transient flow (with a variable time scale).

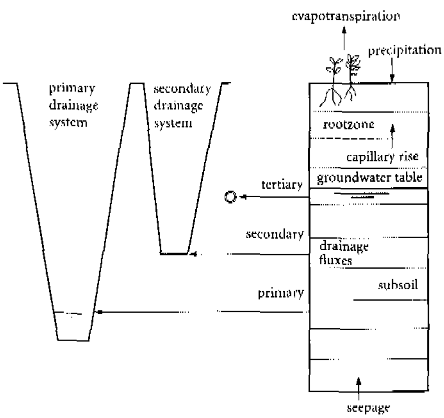


Figure 2: Plot schematization in MOZART

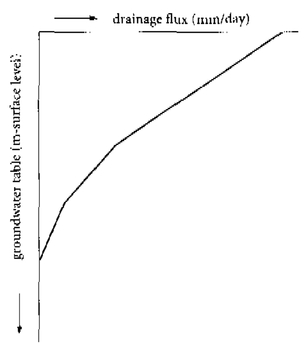


Figure 3: Drainage relation within MOZART

INPUT AND OUTPUT DATA

The input data of the model are precipitation, potential evaporation and possibly sprinkling in agriculture and groundwater extraction for the benefit of agriculture, industry and drinking water. The input parameters are based on a unique combination of a meteorological region, vegetation, soil type and geohydrological situation. The output data are actual evapotranspiration, the moisture content of the root zone and subsoil as a function of time and place, fluxes, groundwater levels, discharges, sprinkling demand, etc.. The model also calculates changes in the supply of water which is foreign to the system and the supply and drainage of water as a carrier of substances. For the Dutch situation all these data are stored and presented in a GIS.

SCHEMATIZATION

For the areal three-dimensional schematization, MOZART uses geographically based nation-wide available information from various data bases:
- PAWN-districts
- WIS (information system for the Dutch watermanagement system)
- DHN (digitally based contour map)
- BIS (soil physical units)
- LGN (land use)

SCALE LEVEL

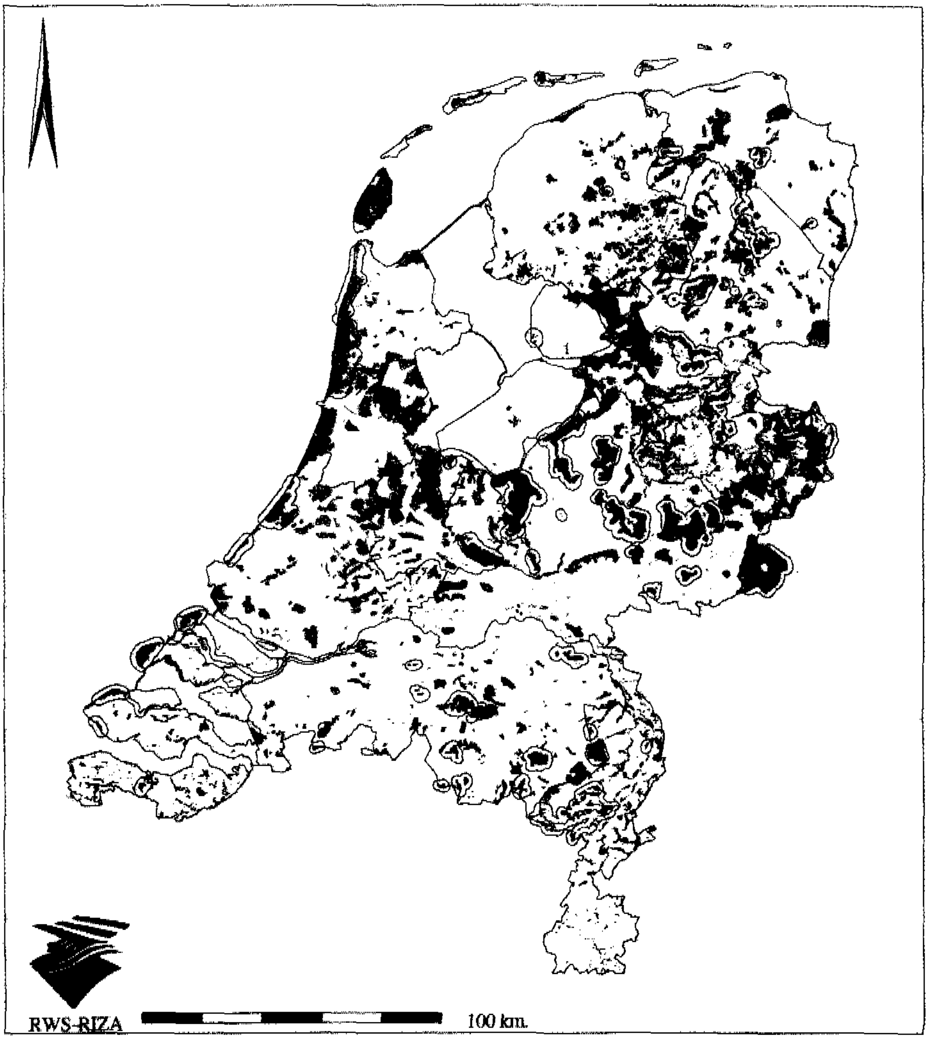
The combination of these data bases results in a large number of unique hydrological units, consisting of grid cells of 500 x 500m (approximately 130.000 units for the entire Netherlands). The 'basic schematization' is a good starting point for regional calculations, while this scale is also in line with the chosen process description. For national calculations, and in particular for fertilization calculations, 'aggregated' schematizations will be sufficient. These less detailed schematizations are derived from the basic schematization.

APPLICATIONS

An example of a national study is the so-called 'Aquatic Outlook'. The aim of this study is to obtain an insight into the biological and chemical, physical and economic values of the Dutch water systems. MOZART played an important role for the evaluation of the current policy with respect to the problems of desiccation (figure 4).
One of the scenarios in the national study is the reduction of desiccated areas with 30% before the year 2015. For this scenario a set of measures had been taken like rising of surface water levels, incrementing drainage resistances and reallocation of groundwater extractions. The output of MOZART (groundwater levels, discharges, sprinkling demand, etc.) can be elaborated to input for the effect models like the ecohydrological model DEMNAT and the AGRIcultural COst Model AGRICOM. AGRICOM needs a.o. changes in mean highest and mean lowest groundwater levels to calculate costs and benefits for agriculture. In the same way the ecohydrological effects (losses and gains in changes of nature value) and change in water quality can be calculated. Figure 5 shows the effect on the mean highest groundwater level for a set of measures and as a result the effects on water logging for agriculture (figure 6).

CONCLUSIONS

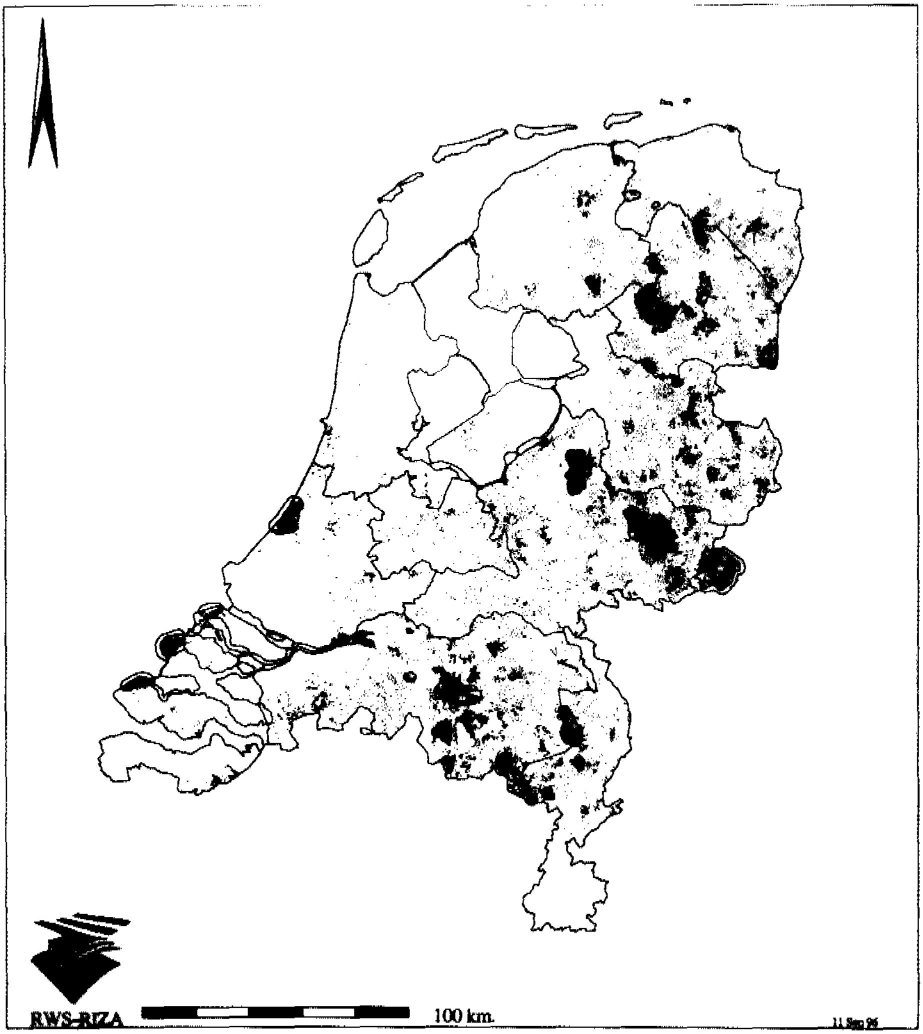
- The user can customize the model to the specific characteristics of the study area.
- The user can choose the desired process descriptions, and can easyly switch between the different numerical schemes.
- MOZART runs on a pc and on a UNIX workstation.
- MOZART supports not only the national analyses, but also provides the opportunity to focus on smaller pilot regions. The required data can be stored in a GIS.
- MOZART proves to be a powerful tool in supporting current and future decision making and in the translation of national policy to regional implementation.



Desiccated nature area "only nature"

Desiccated agricultural area with nature value

Figure 4: Desiccated areas in the Netherlands



more than 30 cm lower
0 –30 cm lower
no change
0 –30 cm higher
more than 30 cm higher

Figure 5: Changes in mean highest groundwater levels

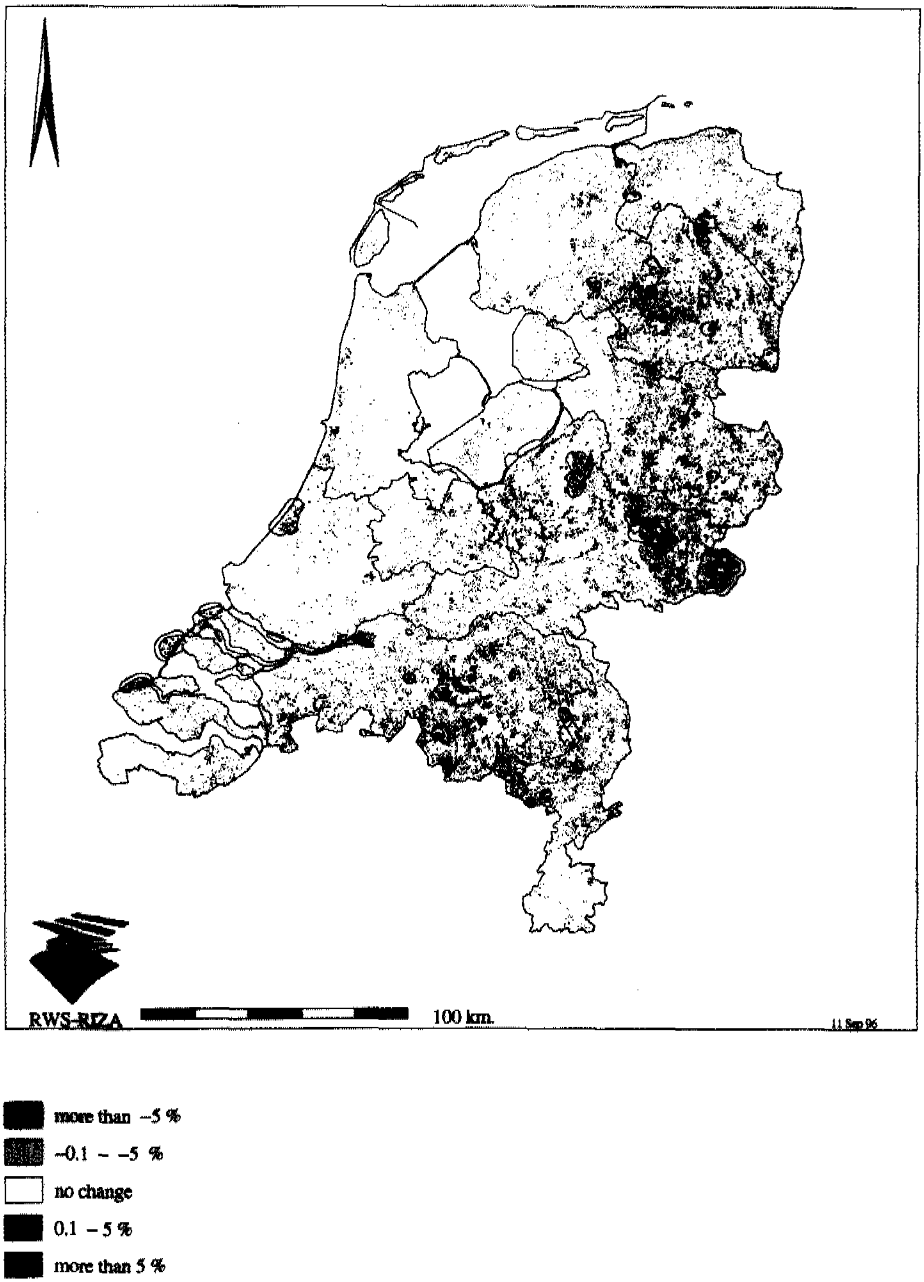


Figure 6: Changes in yield depression due to water logging

STRATEGIES FOR GROUNDWATER MONITORING DESIGN IN THE PO RIVER VALLEY (ITALY)

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ABSTRACT

The methodological and operational design process for groundwater monitoring in the Po River Valley is described. The activities are related to the characteristics of the study area and to the national legislative framework. The strategy selected by the River Basin Authority follows the progressive implementation of the monitoring within a River Catchment Plan. The case of the Po River basin appears to be a reference example of a rational design for improved groundwater monitoring in territories with complex institutional structure, segmented responsibilities and a composite monitoring history. The strategy takes advantage of the intersection between management needs, transfer of appropriate scientific knowledge and design of a suitable institutional framework.

THE FEATURES OF THE GROUNDWATER SYSTEM

The Po River Valley, is the largest alluvial plain in Italy and extends over 25,000 km2 (i.e. 35% of the catchment area) from the piedmont of the Alps and the Apennine chains to the Adriatic coast. About one third of the economic potential of the country is concentrated in it and over 12 million people are settled there. The territory concerned covers some six Regions.

A very important aquifer system made up of Quaternary alluvial deposits of the main river and its tributaries lies beneath the Po Plain. According to recent evaluations, the total volume of groundwater pumped for different purposes (domestic, industrial, agricultural) amounts to 5.5-6.0 109m3/y, which is very close to the total system resources. Locally the maximum potential is exceeded and this has some consequences for the geo-equilibrium (land subsidence). Groundwater satisfies about 95% of drinking water demand in the Po valley for an amount of about 3•109m3/y. It feeds over one thousand aqueducts which account for 80% of the total domestic supply (Passino, 1993).

In view of its hydrogeological and geostructural conditions, the upper plain is characterized by a very high degree of vulnerability to pollution (fig. 1). The heaviest groundwater withdrawals for domestic supply are localized here owing to the high aquifer yield and the good water quality. However, the larger urban centres, along with the most intensive industrial and farming activities, also fall in this section of the Plain, which results in a very high pollution potential and risk. The situation concerning groundwater pollution is particularly serious in many parts of the Plain and has often necessitated relaxing the drinking water standards provided by the legislative act (DPR 236 issued in 1988) embodying the E.U. Directive 80/77. Consequently, distribution of poor quality water for human consumption has been allowed.

The most typical problems of anthropogenic pollution of groundwater are related to three main groups of substances: organohalides, nitrate and pesticides (Giuliano, 1995). Some "natural" pollutants, such NH4, Fe, Mn and As are frequently recorded at concentrations higher than allowable drinking water standards in groundwater pumped from aquifers with reduced redox-conditions. Saline intrusion occurs in coastal areas liable to local aquifer over-exploitation.

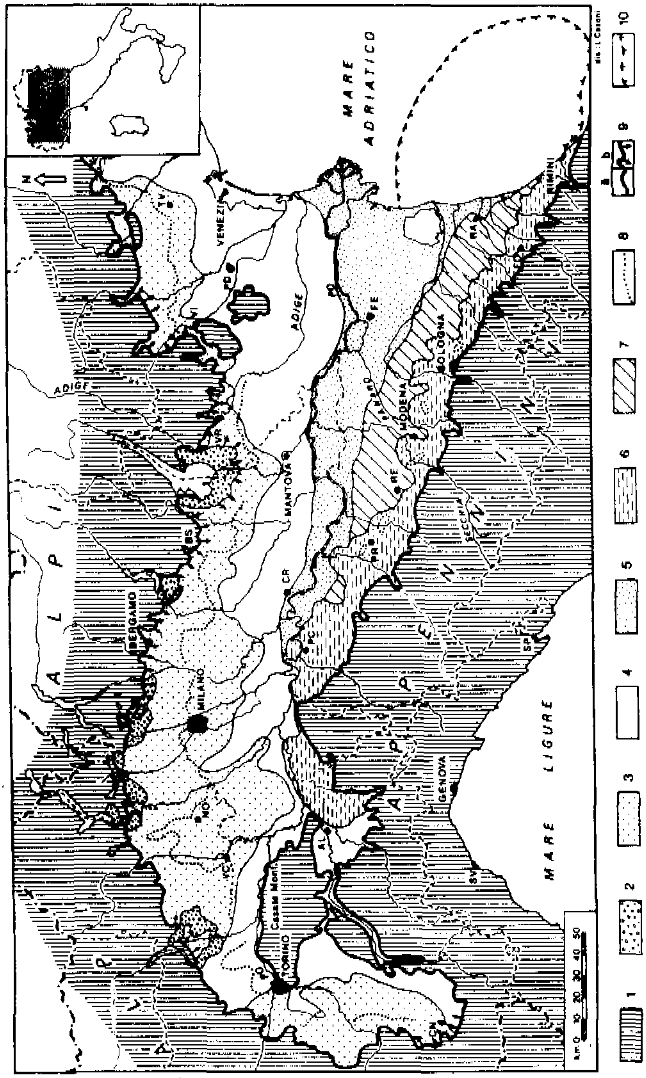

Figure 1: Outline of the principal hydrogeological units in the Po Basin. Legend: 1) Apennine and Alpine ranges; substratum of the Po aquifer; 2) Principal morainal deposits of the pre-Alpine glacial apparatuses, of variable permeability, with aquifers marked; 3) Fluvio-glacial and fluvial deposits of the high pre-Alpine plain, mainly with very high transmissivity, and free groundwater potentially vulnerable in the upper portion; 4) Fluvio-glacial and fluvial deposits of the Piedmontese plain, with limited transmissivity and vulnerable free groundwater; fluvial deposits of the middle and low Lombardy plain, with medium to high transmissivity, mainly free and potentially vulnerable groundwater; fluvial deposits of the middle and low Veneto plain with variable confinement and limited vulnerability; 5) Fluvial deposits, mainly from the Po, with medium transmissivity and confined groundwater with limited potential vulnerability; 6) Fluvial deposits of the Apennine watercourses, with medium to limited transmissivity, vulnerable free groundwater in the high plain, and restricted to groundwater in the vicinity of the alluvial fans; 7) Fluvial deposits of the floodplain of the Apennine watercourse, with very low transmissivity and confined groundwater of very limited vulnerability; 8) Line of seeps; 9) Boundary of the Po aquifer towards the mountain ranges; a) not replenished; b) replenished; 10) Off-shore boundary of the Po aquifer.

THE PRESENT FRAMEWORK OF GROUNDWATER MONITORING

The activity of monitoring groundwater in Italy is dependent on a rather composite frame of national legislation and on a hierarchical commitment of control responsibilities.
Observations on groundwater quantity aspects have long been the responsibility of the bodies involved in the supply of water for civil uses and in agricultural development (irrigation and land drainage). An nation wide instrumental network has been also operated by the National Hydrographic Service for 50 years with the improvement of agricultural management as its main concern. It covers the main alluvial plains in the country and the Po Plain in the first instance.
The endorsement of a comprehensive water pollution control Act (Law 319/1976) assigned to the Regional Governments the responsibility of setting up suitable permanent networks to evaluate quantity and quality characteristics of groundwater bodies, as well as of establishing regulatory monitoring programmes. The networks were allowed operate within centralized or decentralized schemes according to the functional organization of Regional Governments, with the operational support of territorial bodies (Provinces, Public Health Units, public water companies). The task, however, was not fulfilled by all the Regions, and in some cases the networks were shut down after only a short period of operation.
Regulatory obligations linked to waste disposal activities (landfills, water treatment plants, etc.) have played a role of primary importance in the development of groundwater monitoring. Here, too, control is in the hands of the local Public Health Units.
From 1988 on, the above-mentioned legislation covering water destined for human consumption (DPR 236/88) strongly reiterated the need for groundwater quality monitoring at supply sources. Most of this information is independently collected by the bodies (Municipal Boards, Consortia, public companies, etc.) responsible for water supply, as well as by Public Health Units. Other more recent legislative acts following E.U. Directives in the water sector are also of interest in understanding the segmentation of monitoring activities in the Po Valley.
In addition, a network for producing large-scale information on groundwater resources was very recently set up by the Ministry of Environment (Ministero dell'Ambiente, 1993; EEA, 1996) as a component of the National Environmental Information System (SINA). A substantial part of the information comes from the Po Plain and makes use of selected stations from existing regional or local networks.
As a result of the complexity of the legislative framework and the high concentration of economic activities, a considerable number of groundwater monitoring activities take place in the Po Plain. A survey carried out by the River Basin Authority has revealed the existence of several hundreds of observation points scattered very unevenly over the plain (fig. 2).
The networks often cover a limited area and may have special objectives. Many of the networks are concerned with the regulatory control of polluting point sources, while consolidated networks operating at a regional scale to gather general purpose information on both quantity and quality aspects of groundwater currently exist in only two Regions. In the larger remaining portion of the Basin (indicated by dark shading on the map of fig. 3) similar networks are still to be defined or re-designed on the basis of previous experience.
The existing networks dealing with piezometric head show characteristics which are largely non homogeneous as regards their structure, tasks and procedures, while they do not refer to a consolidated conceptual model of the groundwater system. The networks dealing with quality aspects are generally referred to specific contamination problems but have little relevance to the analysis of groundwater at the basin scale. Indeed, there is no unique, structured monitoring network dedicated to the study of resource dynamics and the evolution of quality in the Basin groundwater system. The data collection and operational procedures adopted in existing networks are largely inhomogeneous, so that the systematic use of available information could be unreliable.

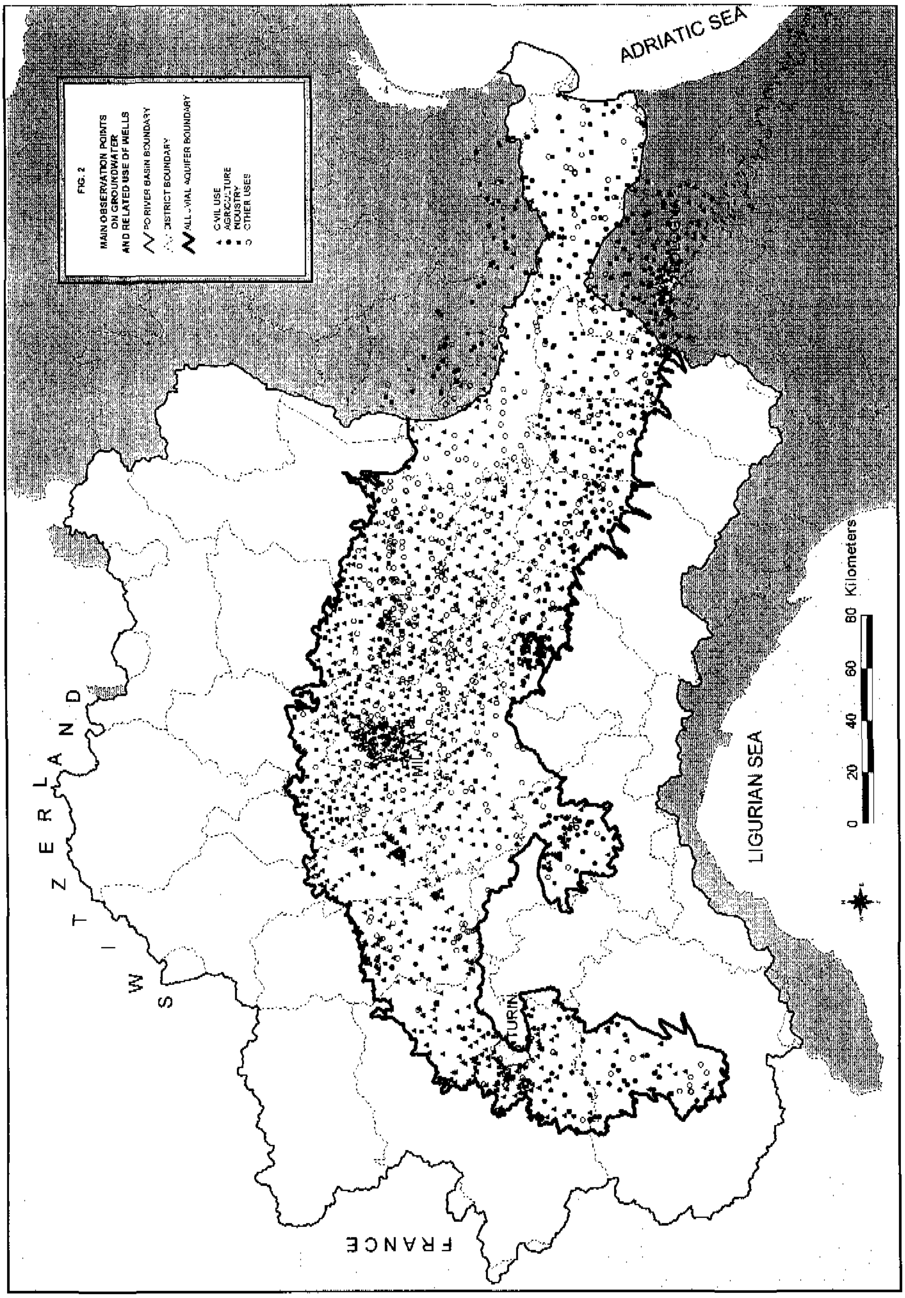

Figure 2: Main observation points on groundwater bodies and related use of wells

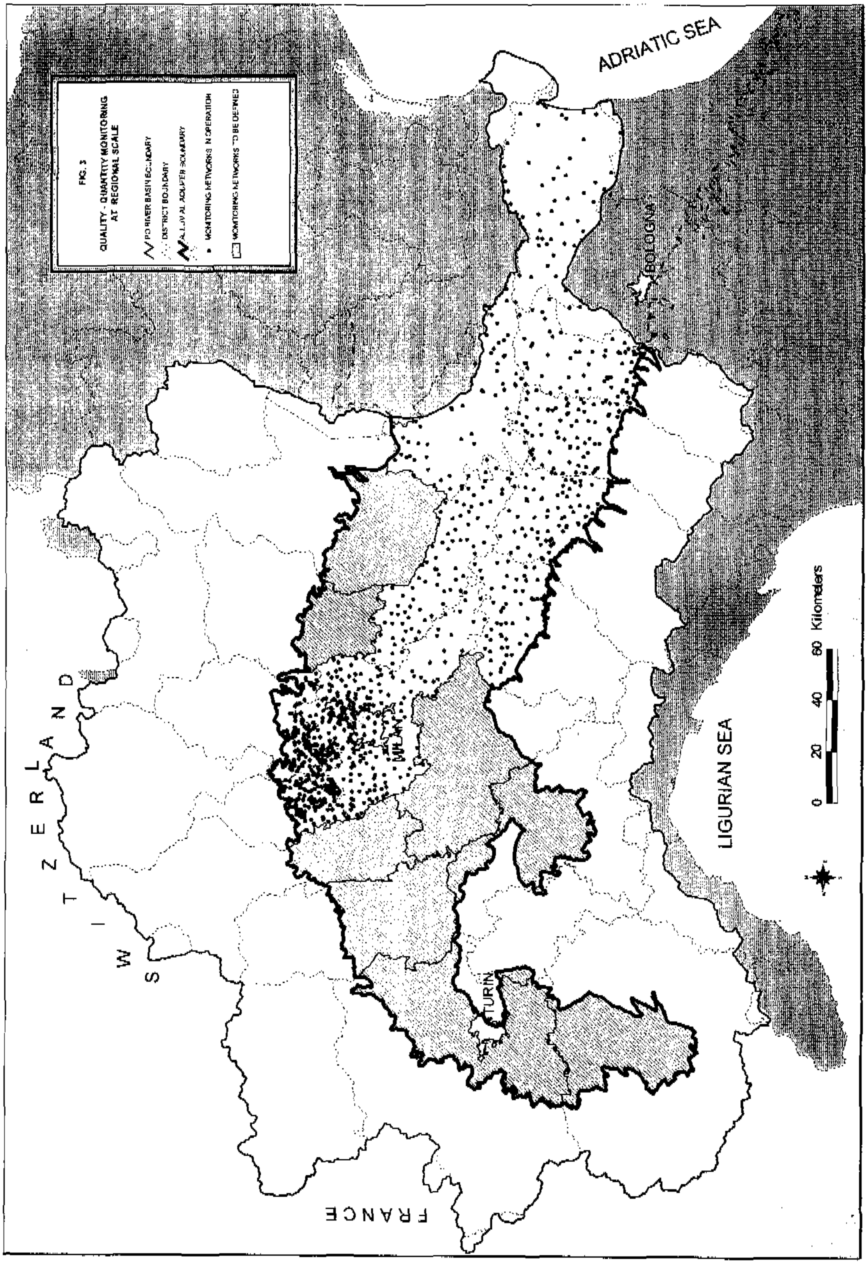

Figure 3: Situation of quality-quantity monitoring networks at regional scale

MONITORING STRATEGIES

In the Po River Valley the outstanding importance of groundwater in the satisfaction of drinking water requirements, along with the coincidence of hydrogeologically vulnerable areas, location of pumping centres and clustering of pollution risk sources (Giuliano, 1995), call for careful planning in the use and protection of groundwater. The design and operation of suitable forms of groundwater monitoring occupies a central position in the River Catchment Plan being developed by the River Basin Authority. The creation of a dedicated basin-wide network is seen as a medium-term objective to be attained by following an iterative pathway within the Plan and based on validated procedures.
The monitoring strategies for groundwater have been outlined in their essential features and aims, identifying the implementation pathway and related phases but their implementation is, however, still in the very early stages.
It is planned to use a quality network as the basic tool to evaluate groundwater utilization policies and to verify the efficiency of quality recovery actions in degraded aquifers, with specific reference to the E.U. drinking water standards. This network will be able to exploit accumulated scientific knowledge concerning the groundwater system (geostructural, hydrogeological and geochemical features and pollutant dynamics) in order to attain a high degree of representativeness and efficiency. The network will be designed with due consideration being given to providing the required information:

- to define a complete framework of groundwater resources and dynamics in relation to uses and climatic variability;
- to identify trends in groundwater quality related to uses and to the influence of non-point contamination factors;
- to identify critical areas related to quantity and quality aspects;
- to determine the limits of groundwater resource development and appropriate intervention strategies.

From a conceptual standpoint, groundwater monitoring is considered as part of a large, unique project dealing with the development of systematic (meteo-hydro-quality) monitoring in which data collection and information management in the Basin are functional to planning requirements. The basic aim is to optimize the trade-off between the quantity of data collected and the quality of information thus gathered before proceeding to create new networks. The approach adopted in this project calls for the following steps:

- identification of information needs in relation to the users;
- characterization of existing monitoring networks in terms of efficiency and operational features;
- definition of optimal data flows between the Basin Authority and the bodies (central and regional) concerned with producing the information;
- definition of design and operating standards.

The requirements are evaluated on the basis of the analysis of the physical processes involved, the relative space-time scales, the key variables, the technological feasibility of their measurement (direct and remote mode) as well as of their use in prediction and simulation models. The definition of information fluxes starts from the critical analysis of management requirements and the key features of operational structures located in the basin. The analysis is carried out separately for ordinary (routine) operating conditions and for emergency situations arising out of the multiplicity of the bodies involved in the decision-making process.

By comparing the present monitoring situation with the conceptual reference framework, integration and qualification needs are defined for data collection and transmission, network patterns, and technological and analytical tools. Several scenarios referring to different levels of hierarchization/aggregation of networks and centralization/decentralization of monitoring responsibilities are evaluated, leading to the selection of one which will allow present structures, expertise and resources to be preserved and co-ordination between different structures to be maximized, as well as ensuring efficient data collection.
The groundwater monitoring strategy will be developed according to the structure of the above project and will satisfy the requirements arising out of the specific needs of the sector and its role within the Po River Catchment Plan.
The River Po Basin Authority, with the scientific support of several research institutions, recently launched a number of background initiatives aimed to rationalize monitoring activities and to ensure compliance of data collection with the planning goals.
A research project in collaboration with NRA-WRc (UK) and Water Research Institute is currently under way to elaborate optimal design and operation of procedures for surface and underground water bodies (Pagnotta et al., 1995). The procedure will be tested in two areas representative of typical morpho-hydrological conditions of the Basin.
A special project is being launched to evaluate the intrinsic vulnerability of groundwater in the Basin (plain and pre-alpine hills) at the operational scale (1:50,000). It is focused on the identification of areas of high natural sensitivity where the principal aquifer is of specific interest for drinking water supply purposes. The assessment is based on a ranking procedure which uses several parameters (soil and unsaturated zone features, hydrogeological conditions of aquifer, depth to groundwater, hydraulic conductivity) describing the hydrogeological settings in relation to potential pollutant migration. The vulnerability mapping is carried out in accordance with the six grade classification proposed and extensively tested by the National Group for the Defense from Hydrogeological Hazards (GNDCI, 1990)
The suitability of groundwater quality for drinking purposes is preliminary evaluated at the basin scale by means of a screening technique jointly developed by IRSA and the GNDCI (Civita et al. 1993). The proposed classification is based on several parameters deemed fairly representative of general quality conditions for groundwater (table I). In particular, chemical and physical parameters (hardness, electric conductivity, sulphate, chloride, nitrate) and undesirable substances

Quality Levels		PARAMETERS							
		Group 1 (chemical and physical parameters)					Group 2 (undesirable substances)		
		TH (°F)	El. Cond. (mS/cm)	SO_4 (mg/l)	Cl (mg/l)	NO_3 (mg/l)	Fe (μg/l)	Mn (μg/l)	NH_4 (μg/l)
optimal	A	$15^{(1)} \div 30^{*}$	<1*	<50**	<50	<10*	<50	<20	<50
acceptable	B	$30^{(1)} \div 50$	1* ÷ 2	50** ÷ 250	50 ÷ 200	10* ÷ 50	50 ÷ 200	20 ÷ 50	50 ÷ 500
poor	C	>50	>2	>250	>200	>50	>200	>50	>500

Notes:
(1) Minimum recommended value.
* Intermediate value between the Maximum Allowable Concentration (MAC) and the Guide Value (GV) (DPR 236/88).
** GV times two.

Judgment relating quality level and use

A: Suitable for drinking use with no treatment; acceptable for the majority of industrial and agricultural uses.

B: Suitable for drinking use with no treatment, with some limitation for industrial and agricultural uses.

C: Unsuitable for drinking use, with some limitation for other uses.
C1: requiring specific treatment.
C2: requiring simple or advanced oxidation treatment.

Table 1: Classification criteria for groundwater quality (9 classes)

(iron, manganese, ammonia) are aggregated in two groups for consideration. The classification criteria identify three levels of decreasing quality, i.e. optimal, acceptable, poor. The intersection of the quality levels with the two groups of parameters gives a nine class ranking system (e.g. A1A2, A1B2, A1C2,...). The quality levels have some relation to the threshold values of the parameters set out in the Italian law as guide values (GV) and maximum allowable concentration (MAC) for drinking purposes and to good technical practices of treatment of drinking water.
The first phase in the implementation of the new monitoring strategy for groundwater relates to the development stage of the River Catchment Plan and includes the following items:

- identification of areas for proper groundwater use in relation to management options of the River Catchment Plan;

- identification of critical areas to be intensively monitored and of specific priority contaminants in relation to pollution activities (rice and corn growing, cattle farming, toxic waste land disposal etc.);

- development of improved monitoring procedures;

- selection of pilot areas where design operational guidelines may be tested and the collected information validated;

- definition of a conceptual model of the groundwater system and (point and non point) pollutant patterns to which the monitoring should be tailored;
- identification of a basin-wide quality/quantity network devoted mainly to aquifers actually used for drinking water supply as well as to emergency groundwater resources;

The second phase of monitoring activity will coincide with the operational stage of the River Catchment Plan and include:

- application of protocols for data collection, information analysis and reporting;

- validation of information produced by the network with reference to Plan goals and to the evaluation of recovery actions;

- reporting on trends in groundwater quality improvement at the basin scale and on climatic impact on groundwater resources;

- validation of the exchange of information flows between the Authority and the Regional Governments as well as the central Government informative systems.

The present stage of implementing the new monitoring strategy for groundwater consists in a partial completion of the above mentioned first phase, supported by a consolidated conceptual reference framework and by the results of the background research projects. The issues are progressively discussed with the concerned Regions in order to define the action plan for the execution of the second phase of the monitory activity. This plan will be conjunctively executed by the River Authority and the operating agencies of the Regions.

CONCLUDING REMARKS

The creation of a unique, dedicated monitoring network for groundwater in a highly developed area, such the Po River Basin, can only be achieved progressively via a process which ensures the application of accumulated knowledge to the system and preserves the expertise and resources associated with existing operating structures. Such a network will play an important role in the development and implementation of the River Catchment Plan for water resources basin management.

High grades of efficiency in the network design may be gained by projecting existing structures and procedures into a scientifically sound reference monitoring framework in which data collection and information management are strictly functional to the requirements of planning implementation. This framework is defined according to the analysis of physical processes, to the technological feasibility of observing key variables and to the maximization of information flows between the bodies involved in the water management process.

The case of the Po River basin appears to be a reference example of a rational design for improved groundwater monitoring in territories with complex institutional structure, segmented responsibilities and a composite monitoring hystory. The new strategy takes advantage of the intersection between management needs, transfer of appropriate scientific knowledge and design of a suitable institutional framework.

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THE ROMANIAN NATIONAL SYSTEM OF GROUNDWATER QUALITY MONITORING

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ABSTRACT

The aim of the paper is to present the structure, operating mode and results obtained from the National System of Groundwater, Quality Monitoring (SNSCAS) as implemented in Romania.Elements concerning evolution of shallow grondwater pollution are rendered evident, together with their implications on the operation of centralised water supply systems, especially along the line of establishing and observing some sanitary protection areas.

MAJOR ITEMS

The material approaches the following main problems:

- specific aspects connected to the shallow groundwater resources;
- presentation of the objectives, structure and operation of The National System of Groundwater Quality Monitoring (SNSCAS) as an integral part of The National System of Water Quaility (SNSCA) in Romania;
- the role of The Environmental Engineering Research Institute (ICIM) in elaborating the system's conception, the methodologies for bore hole monitoring, and the manner in which the in-field data are processed and interpreted;
- the data processing and interpretation by the ICIM teams of specialists, enabling the presentation of the shallow groundwater quality evolution and forecast. It should be mentioned that monitoring the specific pollution indicators during a significant time period was necessary, as well as monitoring those areas of the country that are of great importance to the water users;
- emphasis is put on the negative impact of certain social-economic activities with the use of computer simulation and maps of the pollution sources of shallow groundwater;
- the need to establish and specify the characteristics of the sanitary protection areas for those strata used as water sources for the centrlised drinking water supply systems.

CONCLUSIONS

The complex activity carried out up to now in the field of groundwater quality monitoring has led to the gathering of extremely valuable information. On the one hand, this has contributed to the understanding of the process of groundwater quality formation and evolution, depending on the hydrogeological characteristics of each aquifer. On the other hanbd, it gives the possiblitiy to substantiate the strategies that must be adapted in the near future in order to provide an adequate protection of the groundwater resources. The technology of the methods used in the shallow groundwater quality monitoring activity implies an interdisciplinary approach of the issues and a better understanding and awareness from the part of the users of these resources. The results of this activity are extremely favourable for the improvement of the living conditions and for optimising the use of groundwater as an important resource.

JOINT COMMISSIONS ON TRANSBOUNDARY WATERS BETWEEN HUNGARY AND THE NEIGHBOURING COUNTRIES

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ABSTRACT

There are seven countries around Hungary. Nowadays we have six agreements in force and for the implementation of them there are six joint commissions on Transboundary Waters (later JCTW). Until now, we do not have an agreement with Yugoslavia. With Croatia and Slovenia, Hungary signed the new agreements on JCTW in 1994 while the agreement with Ukraine was signed in 1993. New agreements with Slovakia and Romania are under preparation and will be based on the "Convention on the protection and use of transboundary watercourses and international lakes (Helsinki, 1992) and on the "Convention on co-operation for the protection and sustainable use of the Danube river" (Sovia, 1994).

At the first stage, the agreements mainly covered flood control questions, at the second stage, surface water management problems are included too. later, water quality questions and the problem of water resources management were included while the next stage is planned to deal with groundwater related issues. In the beginning, the agreements were valid only for several kilometres from both sides of the frontiers, but today there is a shift to include the whole catchment area.

On the poster the organisation and the activities of these six JCTW are shown, including the short story of the Agreements.

RIGA CITY SURFACE WATER CONTROL

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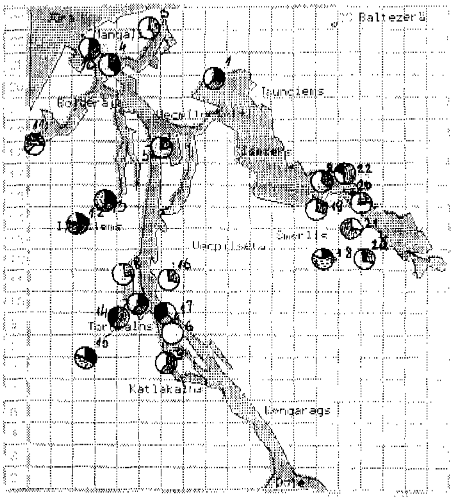
ABSTRACT

To make well-founded decisions self-government of Riga needs understanding of the current, local urban environmental quality. This is of particular importance with regard to the use water for drinking water supply, recreation and fishing.

For Riga City surface waters quality control the monitoring system developed by Prof., Dr. Sc. Ing. Guntis SVIKIS (Riga Technical University, Institute of Quality Engineering) was introduced [1]. specialized data base management system (uses dBase data base structure) is integrated with Regional Map and gives possibility to chart present water pollution on 25 observation stations which location were select in accordance with the needs of the urban development policy of Riga Council (fig.1).



O - moderate pollution

◐ - high pollution

● - very high pollution

Figure 1: Characteristic of total pollution in Riga city surface water

The report has been prepared under the auspices of Riga Council Environmental Protection Board. It is based on information submitted from the Ecological Laboratory of Riga Council. In addition, available data on the subject were used including publications of the Ministry of Environmental Protection and Regional Development.

SITUATION

Riga, which is located at the mouth of the largest Latvian river Daugava, is rich in inland water resources. Rivers, small tributaries of the Daugava, lakes and artificial water reservoirs covers approximately 54 square kilometres or 17,6% from the total space of the city territory (307 km^2). The northern boundary of Riga is the southern shore of Riga Gulf. Total length of Riga's coast line exceeds 15 km of marvellous sandy beaches with great recreational importance.

WATER QUALITY MONITORING STRATEGY

Main objectives are:

- to provide the local decision makers, city administration, urban planners and Riga inhabitants with easily understandable and obvious information;
- to identify the key areas for concern;
- to get reliable information for making decisions concerning land use, water remedial measures, maintenance and promotion of fish stock;
- to obtain a feedback of information on the effects of remedial measures taken, and the need for complementary actions;
- to get information for assessing damage caused by pollution;
- to prepare recomendations for establishing network of protected areas in Riga.

Appraisals of water quality are based on mesurements of oxygen consuming substances (COD, BOD), nutrients and contaminants (metals, detergents, chlorides). Oxygen saturation is used for evaluating oxygen conditions. Analyzed metals are lead, cooper, nickel, iron, chromium, cadmium, manganese and zink. The water quality was evaluated with reference to guidelines for classification of water quality in lakes and rivers issued by Latvian Ministry of Environmental Protection and Regional Development.

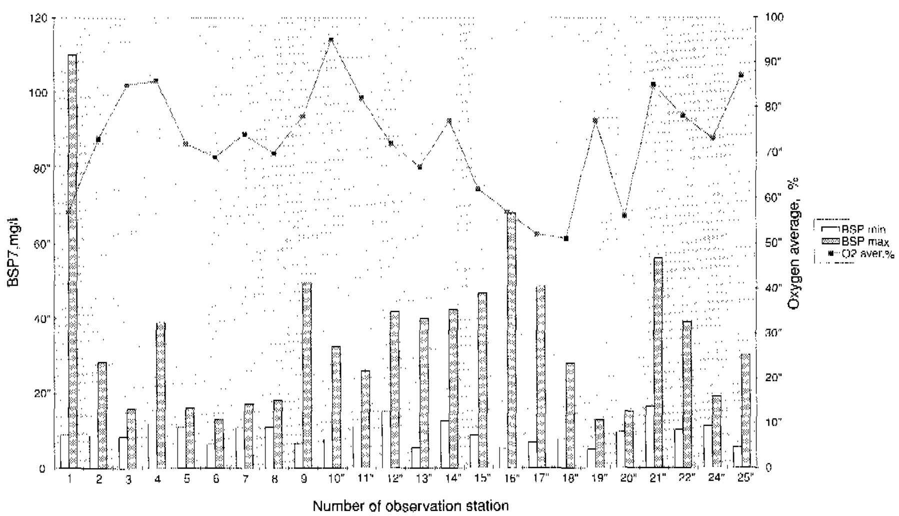


Figure 2: Oxygen situation and load of oxygen substances

FINDINGS

- Nutrients are abundant in the surface water of Riga.
- Although the water bodies is subjected to a high load of oxygen consuming substances, the overall oxygen situation (fig.2) is generally good but varies extensively over the year.
- Concentrations of all metals are low in almost the whole controled Riga surface waters basin.
- According to criteria, which have been used as a foundation for appraisals of water quality in our country, Riga surface water quality might be evaluated as not satisfactory since the pollution level "very high" was periodically found at 56% stations (fig.3). Results varies extensively over the year.

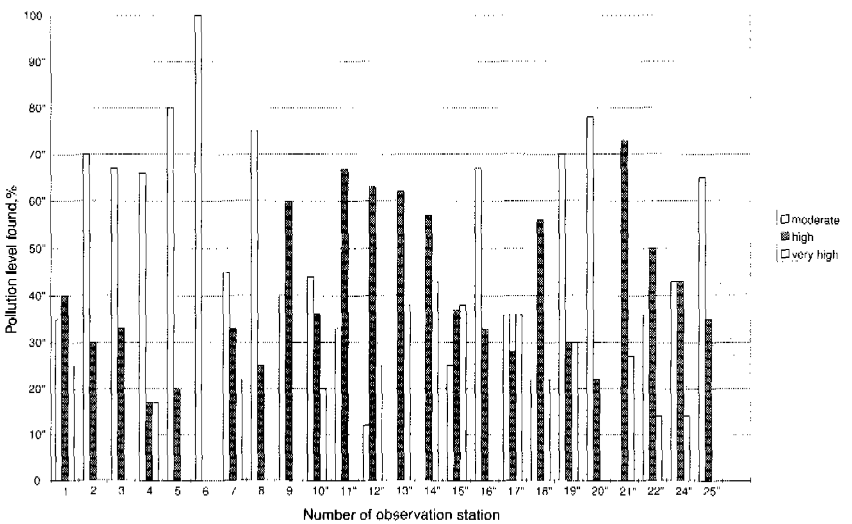


Figure 3: Riga city surface water quality characteristic

Additional indicators which provide indication of the overall health of the aquatic ecosystems is obtained from accounting of fishery results, which is carried out by the Riga Environmental Protection Board due to quantity (catch amount) and species in each water body inside Riga administrative boundaries. In some water bodies yields of economic valuable fish species tends to decrease.

Future tasks

- Investigations of sediments and observations of biological determinands should be included in the monitoring programme.
- Data management after river basin and sub-basins should be implemented instead of district-based data organisation.

CONCLUSIONS

Characteristic features of the water in Riga inland aquatic systems are its satisfactory oxygen content, low concentrations of metals, high colour value, abundance of organic material and high nutrient content. Although monitoring of water quality have, up to now, only been realized in limited part of the Daugava catchment area within administrative boundaries of Riga, it offers the possibility to surpervise pollution level of Riga surface waters.

Internal waters are included in the area covered by the 1992 Helsinki Convention and keeping surface water pollution under continuous observation as well as collecting the most up-to-date and relevant loading data are going to become the duties of all riparian states. Most of the loaded nutrients and organic matter (causing oxygen consumption) enter the Gulf of Riga in the Riga City region, consequently, assessment of the state of the Riga's surface waters takes contribution to the ongoing work to restore the Baltic Sea marine environment.

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A STATUS FOR WATER QUALITY MONITORING IN VIETNAM.

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ABSTRACT

Water Quality monitoring is fairly new in Vietnam and has only been used regularly since 1988, when the first monitoring system was set up on the Mekong River. Since then the monitoring system has been expanded to cover the main tributaries, amongst them the Srepok River in the Central Highlands. The main effort in monitoring has been given to the quantitative aspects. The Srepok Water Action Plan has shown the need for an expansion of the water quality monitoring network for a sound management of the water resources.

WATER QUALITY MONITORING

Vietnam is situated in the sub tropic and tropic zone. Rivers play a significant role in the structure of Vietnam and the two river systems, the Mekong River in the south and the Red River in the north contains the larger part of the Vietnamese population. These rivers are greatly influenced by the annual cycles of the rainy and the dry seasons. Both rivers flood the delta areas annually. The river plains are used intensively for agricultural purposes, mainly for growing rice.

Monitoring on water quality is a fairly new topic in Vietnam.
Traditionally the interest on water has been concentrated on the quantitative aspect regarding possible use for hydro power, irrigation and to a lesser extent water for the industry. Drinking water in Vietnam is mainly taken from ground water sources through shallow wells, but the growing population and thus growing pressure for water has led to an increase in abstraction of surface water.

The Vietnamese strategy on monitoring in surface, ground, sewage and irrigation waters is in its infancy and has not yet been formalised and standardised. Some tentative criterion based on practices in the major cities were published in 1993 and standards for drinking water quality were issued by the Ministry of Health. In 1995 the Ministry of Science, Technology and Environment issued standards for ground and surface water. Classification of water bodies is restricted to two classes whether for drinking water purposes or "other purposes". This is in fact an "Either /Or" system which may be quite easy to use, but will not be able to distinguish between the many different types of water bodies in Vietnam.

At present there are no national plan for a co-ordinated monitoring programme taking care of these aspects. One of the reasons for the lack of monitoring programmes is also the lack of the necessary legislation. Vietnam has at present a Law on Environment that also include aspects of water pollution. A Law on Water is expected to be adopted within the next one or two years and that law will be the used as the framework for national plans for monitoring.

The Law on Environment from 1994 specifically stresses that the state bodies are responsible for assessments of the ambient environment and that the water resources should be protected.

Point-source polluters are ordered to take the necessary measures to reduce and prevent pollution. Violation of the law is very clear: The polluter pays and can be prosecuted by the civil or criminal law.

The Draft Law on Water is mainly a regulation on the use and exploitation of the water resources, thus also including a chapter on the planning and management of the resources. However, one of the chapters stipulates the water quality control and discharge of waste water. Most of the control is in a form of registration, but does not take into account, that the ambient water quality control should be organised and linked to a monitoring programme.

Today surface water quality monitoring in Vietnam is restricted to the programmes running in the lower Mekong basin and its tributaries in Thailand, Laos, Cambodia and Vietnam. This monitoring programme is sponsored and operated by the Mekong River Commission through national laboratories in the riparian countries. Water Quality Monitoring in the Lower Mekong was established in 1985, when stations were set up in the main river in phase I. In phase II in 1992 the monitoring network was extended to cover some of the upstream rivers of the Mekong system.

The data from the Mekong River monitoring network have not been used extensively. However, the national laboratory in Vietnam have extracted information from the national stations and have found increasing trends within the nutrient variables, indicating a growing human impact on the Mekong River.

In the draft Environmental Programme from the Mekong River Commission it has been proposed that a third phase be launched. This phase aims at co-ordinating the different policies and strategies within the Mekong river riparian countries and strengthen the efforts in controlling water pollution. The monitoring will be based on the existing system with some reviews and redesigns according to the new strategy and include upgrading of laboratory facilities.

A monitoring programme on the Saigon and Dong Nai rivers have been running since 1992. It is operated jointly by the Environmental Committee of Ho Chi Minh City and local department of the Ministry of Science, Technology and Environment.

The present programmes are designed traditionally to provide data on a long row of chemical and physical variables. Possibilities for running monitoring programmes have also been restricted by a severe lack of laboratories and equipment necessary for performing water quality analyses. However the use of the data has never been discussed in detail and the abstraction of information has not been standardised and used by the national authorities.

Danida, the Danish international development assistance, has through the Mekong River Commission sponsored the development of a Water Action Plan for the Srepok River Basin. This is an integrated water resource management plan that takes into consideration all aspects of water within a basin. Legal and institutional aspects are also considered. As a part of this water action plan, a new monitoring programme on surface-, ground- and sewage water has been proposed for the Srepok River system, a tributary to the Mekong River. This programme will take another approach towards collecting data. It will start by defining the goals and through the design of the monitoring network concentrate on collecting only those types of data, that will give information on human impact on the water quality. The water quality in the Srepok Basin is at present considered pristine. A substantial growth in the demand for water in the basin, combined with a growth in industrialisation, urbanisation and population growth, will undoubtedly increase the impact on the water resources and quality. The present use of mainly ground water in the river basin is expected to shift towards use of surface water.

A prerogative for this monitoring programme is access to a water quality laboratory. Through a donation from Danida a modern water quality laboratory was set up in the spring of 1996 in the capital of the Dak Lak province in the centre of the Srepok river basin.

MONITORING OF FRESHWATER QUALITY IN FINLAND

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ABSTRACT

The national strategy for environmental monitoring is currently under review in Finland. There has been considerable discussion about the extent to which the present freshwater monitoring networks match the national and international demands for information, as determined by decision-makers, researchers and citizens. The most important environmental issues to be considered in the monitoring of freshwater quality are eutrophication, acidification, harmful and toxic substances, and changes in climate and biological diversity. The present monitoring programmes are considered deficient in some respects, especially concerning biological monitoring and would benefit from the incorporation of new methods and strategies. This is particularly true for the monitoring of biological diversity and littoral zone.

MONITORING NETWORKS

PRESENT NETWORK

In Finland, the monitoring of freshwater quality was begun in the early 1960s with the aim of collecting information on the state of surface waters in loaded and natural areas. The importance of monitoring had increased when the pollution effects of rapidly expanding industrial production started to become apparent in water bodies. The Water Act, which came into force in 1961, resulted in *a statutory local monitoring system.* Industrial and municipal wastewater treatment plants, fish farms, and other polluters are obliged to monitor the quantity and quality of their discharges and the effects of their wastewater on the receiving waters. Currently, there are hundreds of statutory monitoring programmes, which are approved by the regional environment centres and carried out by officially supervised laboratories. In the statutory monitoring, there are about 4000 sampling sites, where physical, chemical, and biological variables are analyzed.

The basic national monitoring networks set up by the water authorities in 1962!1965 covered nationally important rivers and lakes and some lakes with regional importance. There are currently 71 lake sites and 69 river sites where about 30 physical and chemical variables are monitored. The sampling frequency is three times per year in the lakes and four times in the rivers. Although the sampling frequency is not sufficient for reliable flux estimations, it is sufficient to gain a general overview of water quality in Finland and to trace trends in the water quality of the most important rivers and lakes. *The total load of inland waters into the Baltic Sea* is calculated, since 1970, on the basis of 12 annual samples in the mouth of 30 rivers.

Phytoplankton monitoring was introduced in 1963. In the late 1980s, bottom fauna, periphyton, and zooplankton were also included in the monitoring programme to obtain more comprehensive background information on the *biological quality* of lakes. Biological monitoring has been focused only on the pelagic areas of lakes, mostly because of limited financial resources. There is an intention to include macrophytes in the lake monitoring programme in the next few years. Ecological monitoring of rivers, by means of bottom fauna sampling and habitat surveys

(National Rivers Authority, 1992), will also be included in the national monitoring programme.

The monitoring of water quality in *transboundary* rivers between Finland and Russia, begun in 1966 according to a bilateral agreement, has focused on three heavily loaded rivers and one lake. Other transboundary rivers, between Finland and Norway or Sweden, are mainly included in the basic national network referred to earlier.

The transport of suspended and soluble matter from land areas to surface waters in *small drainage basins* has been monitored since 1962. Water quantity and quality are measured in order to study the effects of changes in silvicultural and agricultural land use on material transport in 15 basins.

Heavy metals and organochlorine compounds in fish (pike **Esox lucius**, vendace **Coregonus albula**, whitefish **C. lavaretus**, perch **Perca fluviatilis**) and in lake mussels (**Anadonta piscinalis**) has been monitored since 1978. Monitoring long-term changes in the *acidification of small* headwater lakes due to atmospheric deposition was started in 1987 at 180 sampling sites throughout the country. These sites are monitored once a year. The monitoring network consists of strongly acidified lakes and lakes in different stages of acidification, which is important in the current circumstances in which loading and deposition rates are assumed to be changing.

All the national programmes described above have been designed and coordinated by the appropriate central administration, currently the Finnish Environment Institute. Sampling, most laboratory work and the storing of data are carried out by 13 regional environment centres, which are subject to the authority of the Ministry of the Environment. The regional centres also undertake monitoring activities of their own to satisfy regional needs.

In 1994, the basic national monitoring network for river and lake water quality was substantially reduced because there was a need to develop terrestrial monitoring and financial resources were limited. The most heavily loaded areas are no longer included in the national programme. The monitoring sites withdrawn from the national programme are now included as part of the regional or statutory monitoring networks. This means that particular attention should be given to maintaining and improving the quality of statutory monitoring. In *regional monitoring*, the aim is to reduce physical and chemical water quality monitoring, to increase monitoring of the biological variables and to transfer financial and other resources to terrestrial monitoring, which has already been done at national level.

NEW NEEDS

There is a need to develop a closer relationship between freshwater quality monitoring and *hydrological* monitoring. For example, the station networks should be harmonized and hydrological models should be more efficiently used in data analysis. The need to monitor *biological diversity*, highlighted in the Rio convention, and the effects of climate change must be considered.

During the last decades, the nutrient load from point sources has strongly decreased due to efficient wastewater treatment. This has increased the relative importance of diffuse sources.In the issue of how to implement and finance the monitoring of eutrophication caused by diffuse sources such as agriculture requires special attention. In addition, methods for monitoring the effects of eutrophication and other stress factors in *littoral zones* should be developed.

The project of the European Topic Centre on Inland Waters for designing a freshwater monitoring network for the countries joined to European Environment Agency must also receive attention.

Reporting and assessment of monitoring results, especially those concerning national lake and river water quality programmes, have not, thus far been adequately considered. The state of the surface waters in Finland has been described up to three times after every ten years by dividing rivers and lakes into five classes, from excellent to poor. The classification system (Heinonen & Herve, 1987) has been considered important by decision-makers and citizens. However, the classification criteria and calculation routines need refinement.

CONCLUSIONS

A new strategy for monitoring the total environment is currently under preparation in Finland. In addition, there is a need to critically evaluate the freshwater monitoring. In the national programmes, methods should be developed particularly for the monitoring biological diversity and changes in the littoral zone. Much more attention must be given to improve reporting and assessment of the monitoring data. This will require development of the *databases* and of a map-based user-interface system for a PC environment, for easy handling of data up to sub-basin level.

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POLISH EXPERIENCE IN CREATING AND OPERATING AN AUTOMATIC RIVER MONITORING SYSTEM

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ABSTRACT

Monitoring activity on international watercourses in Poland, among others, is focused on establishing automatic stations for river water quality monitoring.
The program of automatic monitoring system creation for the border water pollution control in the basin of the Oder river has been accepted by the State Inspectorate of Environmental Protection in Poland for implementation in the years 1996 - 2000. According to this program it is planned to create 13 stations for continuous measurement of water pollution. The program was started by establishing 2 monitoring stations: at Widuchowa (on the Oder river) and Porajów (on the Neisse river, tributary of the Oder river).
The monitoring system in the Oder river basin should bring about the economic benefits thanks to the expected decrease in the harm caused by accidental pollution. The precise knowledge on the changes of water quality will allow for more adequate and efficient water management policy.

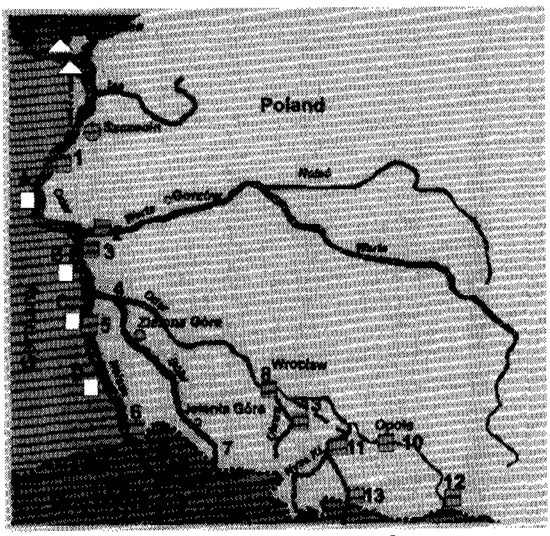
INTRODUCTION

Poland, as a country aspiring to join the European Union, faces a challenge of having to respect all the international agreements on protection of the environment. They include, among others: the Convention on the Protection and Use of Transboundary Watercourses and International Lakes and the Convention on the Protection of the Baltic Sea.
The monitoring of waters is the tool allowing for the assessment of the threat resulting from the bad quality of river waters. Most of the rivers in Poland have been monitored for more then 30 years. However, the uniform principles of river monitoring in Poland were introduced in 1988.
The Oder is one of the biggest rivers in the basin of the Baltic Sea and the second longest river in Poland. Its riverhead lies within the area of the Czech Republic, its total length is 854 km, and its basin area is 118 861 km2. Along 187 km it forms a border between Poland and Germany. The lower part of the Oder river forms a complicated estuary system. The anomalies in the flow of the river, highly dependant on the changes in water levels on the Baltic Sea and meteorology conditions, are visible at Widuchowa.
Being a transboundary river, it receives in its upper part pollution from the heavy industrialised area of Poland, the Czech Republic and Germany, so-called "black triangle". The concentration of the Oder water pollution in the lower part of the river results mainly from the load of pollutants inflowing to the Oder from these areas. From the beginning of the 70s the decrease in the concentration of industrial pollution, and the increase in the communal, agricultural and stock-farming pollution in the river waters have been recorded, especially in the lower part of the river. The immediate result of the latter are high concentrations of nutrients, which cause a strong eutrophication effect, high value of BOD, low concentration of oxygen and unsatisfactory sanitary condition of the river. This limits the use of water, for drinking as well as industrial purposes.
Downstream from Widuchowa, located on the mouth part of the Oder, there is a drinking water intake plant for the city of Szczecin, whose population is about half a million inhabitants. A Petrochemical Complex in Schwedt, Germany, is situated upstream from Widuchowa. The Complex is

supplied by means of the oil pipeline, which crosses the river. This situation, as well as the river transport, may cause accidental pollution of river waters. This is the reason why the measurement station in Widuchowa plays the role of an early warning station. Apart from its early warning function the station supports the monitoring system for the Oder river at its outflow to the Baltic Sea and, strengthens the water quality monitoring program agreed with Germany.
The Neisse is a left-bank tributary of the Oder, with the total length of 252 km and the basin area of 4 297 km2. Its 198 km long part flows within the area of Poland. The upper part of the river flows in the area of the Czech Republic. Along the 60 km long part it forms a border between Poland and Germany. The load of pollution received from the heavy industrialised areas of Poland, the Czech Republic and Germany ("black triangle"), significantly influences the water quality of the Neisse, in a similar way as for the Oder river.
The monitoring data has shown that the Neisse river in Porajów, located where the borders of three countries - Poland, Czech Republic and Germany meet on the river, is excessively polluted with organic substances, nutrients, phenols, sulphurs, dissolved substances and suspended solids. The basic function of this station is to monitor the input of pollutants from the Czech Republic and to strengthen the water quality monitoring program agreed with this country. In Porajów the Neisse is a mountainous river, with a wide range of water level changes.

THE AUTOMATIC MONITORING SYSTEM IN THE ODER RIVER BASIN

The project of the automatic monitoring system was developed in 1996. According to the program it is planned to built 13 automatic monitoring stations. The location of the stations is shown in the map below.



A, B, C, D	- automatic monitoring stations in Germany
1 - 13	- automatic monitoring stations in Poland
a, b	- existing Polish Marine Institute stations

The main objectives of the system are:

- the creating of the early warning system;
- the strengthening of the existing river monitoring networks;
- the calculating of the pollutant loads;

- the conducting of long term observations of the water resources quality and quantity;
- the verification of the existing water management programs.

The following circumstances were taken into account while planning the localisation of the stations:

- the location of important water intake plants;
- the already existing German automatic stations;
- places where the wastewater is fully mixed with the river water;
- technical conditions (easy access by cars, possibility of the connection to electricity and telecommunication networks, safety conditions), etc.

The organisation structure of the system contains:

- the centre of the system - the institution co-ordinating and managing monitoring in the basin, which is planned to be financed by water users;
- the supervising centres (regional) with their own laboratories, responsible for data quality and performing additional analyses (the existing Voivodeship Environmental Inspections).

The creation of the system was started on the base of the PHARE fund financial means. At the first stage two stations were chosen to be built, one in Widuchowa, and the second one in Porajów, Both stations started their measurement program in July 1996.
While choosing the measurement parameters the following factors were considered: the individual function of each station, the current state of river waters pollution, the technical and economic feasibility to perform measurements and the possibility to minimise human activities in the station.
All the factors mentioned above influenced the choice of parameters for both stations. The determinands include: water level, flow, water and air temperature, conductivity, pH, dissolved oxygen, turbidity, ammonium, nitrates, phosphates, chlorophyll (fluorescence), UV- absorption, oil substances (in Widuchowa only). In view of the trace concentration of heavy metals and pesticides there is no need to monitor these parameters. A system of automatic sampling has also been installed at the stations, to make samples available for analysis in the case of any toxic pollution appearance in the river waters.
Total cost of building these two stations amounts to 825 120 ECU, while estimated running cost for one year is 100 000 ECU.

CONCLUSION

The realisation of the system generates demands for the resolving of many technical, organisational, economic and legislation problems.
The monitoring stations at Widuchowa and Porajów are the pilot ones for the automatic river monitoring system in Poland. Both stations have already brought a lot of technical troubles unrecognised before. So it is essential to implement some technical improvements, e.g. of the sampling and filtration installations in order to safeguard them against the pollution with suspended solids. It has been also found that it is necessary to: replace the ultrafiltration module with some other technical solution, introduce the possibility to measure inside the station the parameters hitherto measured in-situ, in view of frequent freezing of the river water table in winter, and large amount of phytoplancton in summer, improve the selectivity of the operation of some analysers and adjust the water sampling installation to the changeable water level in Porajów station, in view of the mountainous nature of the Neisse river, etc.

During the next stage of the stations' operation it is also fundamental to: undertake the investigation of the quality of obtained measurements, investigate the necessity and financial feasibility of introducing the bioalarming system and design the computerised system of alarming the institutions responsible for water management.

The experiences gained until now will enforced the modification and improvement of the system. The very high cost of building and operating the stations at Widuchowa and Porajów can bring about the changes in the number of the planned stations, the range of the parameters and the schedule for the building of the next stations.

Furthermore, the lack of the modern water law (now having been discussed in Poland for a few years) makes the operation of the system rather difficult. The existing Water Law does not allow to find the way of introducing the suitable, financially independent, organisation structure, to make water users involved in water management policy in the river basins. Presently, the operation of the program is financed by the state or regional budget.

The system of the automatic monitoring network should be the common task for the countries situated in the Oder basin. Until now, there has been no discussion on this subject in spite of the existing bilateral agreements with neighbouring countries. The last agreement between Poland, Germany, Czech Republic and European Union for the water protection in the Oder river basin (April 11, 1996) will hopefully give the possibility to make this system a common tool in water management policy in the Oder river basin.

MONITORING OF ALBANIAN WATERS

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ABSTRACT

Albania is a country that is naturally rich in water. High precipitation causes high runoff rates. Three of the country's main rivers, Buna, Drini and Vjosa, have their sources outside the national territory. There are four main lakes: Shkodra, Ohrid, Large Prespa and Small Prespa, and a great number of smaller ones. It is clear that the assessment of water availability for future projects requires the use of criteria for water quality as well as water quantity.

INTRODUCTION

During many years in the recent past, the economical development of Albania has been based on criteria which did not take into account the protection of the environment: use of outdated industrial technologies, concentration of the population in fast-growing urban centres, unrestricted use of chemical fertilisers to improve agricultural yields, total absence of treatment plants for solid and liquid wastes, be those from domestic, industrial or mining origin. All that caused a situation in which levels of pollution in the rivers and other water bodies grew fast, turning surface water in some places inadequate for human use, and even for agriculture. The Hydrometeorological Institute of Tirana, placed under the Academy of Sciences, is in charge of monitoring the quality of surface water in Albania. Starting about 1967, the IHM launched a systematic water sampling programme. Up to 1984, the analyses intended to determine only the physico-chemical properties of the samples. The monitoring network included 47 stations. In each station, 20 to 130 samples have been taken and analysed. Afterwards, it appeared necessary to change the type of analysis to be performed, to assess the pollution levels. Several indicators as dissolved oxygen, biological and chemical oxygen demands, ammonia, and some metallic ions were added. The main problem for these analyses was the insufficient laboratory equipment of the IHM. Important parameters as heavy metals in a country producing chromium and many other metals is smaller amounts, or pesticides in a country which relies on irrigated agriculture, could not be proposed for systematic monitoring. Occasionally only could they be analysed in other laboratories.The network for surface water monitoring was composed, taking into account different factors: strategic points along the main rivers, wherever possible coincidence with a discharge measurement station, presence of a known pollution source (urban sewage discharge, mining or industrial activity). The nation-wide difficulties experienced at the time of political changes in Albania affected the IHM, and no monitoring took place in 1992 and 1993. Since then, sampling and analyses were resumed, although with limited budget. Now prospects are better, due to international assistance projects, and it is expected that in 1996 the number of samples analysed might increase in about 100% compared to 1995. The results obtained in 1994 and 1995 confirmed a clear improvement in the quality of river water following the slow-down of industrial activities. The development of an environmental awareness in national authorities requires now an improved follow-up of the evolution of river pollution.

METHODS

The quality of waters in determined by two factors: the activity of the man and the natural hydrologic cycle. Its changes are mainly determined by the occasional phenomena. For that reason, it is impossible to catch all the probable changes, and in the determination of the number of stations and frequency of sampling some compromises are necessary. The sampling for the evaluation of the quality should be such as to better present its changes in time and space. Meanwhile, it is necessary to evitate the influence of small water inputs before the place where the sampling is made.

The determination of the number of sampling stations in the rivers of Albania was based upon some parameters which characterise the river catchment, and, concretely: their surface, the length and the slope of the rivers (Pomerou & Orlob, 1967). According to these data it results that about 50 stations should be defined in the rivers of our country. The determination of the places for the sampling has been carried out based on the national hydrometric network, as well as on the different polluted discharges. The application of the graphicfor the sampling would allow the evaluation of the contribution of each of these factors concerning the quality of the waters at a given place.

Usually, the determination of the sampling is made statistically and is based upon the measured concentrations, variability of the data, etc. If there is a lack of these data the determination of the frequency of the sampling from the waters of the rivers of Albania is made based on the average ratio between the maximum monthly discharge and minimum one in relation to the catchment area. A ratio smaller than 10 would characterise a well regulated runoff, thus, expecting a small

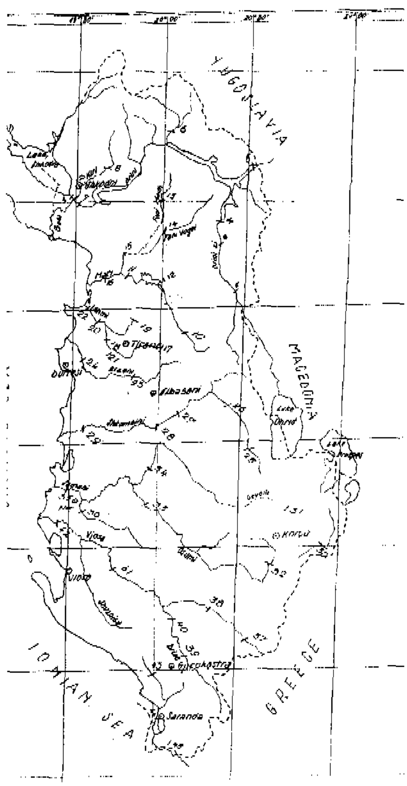

Figure 1: Map of physico - chemical and polution measurement stations.

change on its quality from hydrological regime. The sampling once in two months would make possible catching the change of the quality of the water. If the above ratio is greater than 100 then it is necessary that a larger number of samples is taken during the hydrological cycle (Figure 1). For most of our rivers (Mati, Ishmi, Erzeni, Shkumbini, Semani) it results that in each station of measurement, at least 12 water samples a year should be taken, while in the stations of measurement of the other rivers (Drini, Vjosa) about 6 samples a year are enough (Table 1). The above program entails a high volume of work in the field and in the laboratory. We think that just in the first stage and, gradually, to take water samples in the given station in table 1 at each two months. Together with the problems which may emerge expeditions can be organised for the sampling in some torrents as well, especially in the dry season period (Unesco - WHO, 1978). In this way we can follow up the chemical composition of the rivers of Albania, as well as the impact of the point pollution and diffusing discharges (industrial, urban, agricultural) on its change. For the rivers of Drini Zi, Drini Bardhe and Vjosa, it is proposed to have stations just after they enter the territory of Albania. For all the rivers, stations are planned to be opened in their lower flow in order to know the quantity of the different chemical components they send to the sea.

Catchment Area	Area, km^2	Number of stations
Buna	19573	1
Drini	14173	8
Mati	2435	7
Ishmi	651	6
Erzeni	755	2
Shkumbini	2351	5
Semani	5389	7
Vjosa	6680	6
Kalasa	228	1
Bistrica	108	1
Pavlla	337	1

Table 1. Physico - Chemical and Pollution Measurement Stations.

QUALITY OF THE SURFACE WATERS

It appeared that the most polluted rivers of Albania corresponded to very different pollution sources: paper factory and small food processing plants discharging into Kiri river (the dry season flow of which is very low); uncontrolled discharge of liquid and solid wastes along Tirana river; Shkumbini river is affected by the huge metallurgical complex at Elbasan; and Gjanica river receives flows from industries related to oil: extraction, refinery, treatment of sub-products.As a result of the pollution discharges and decomposition of the organic matter in the waters, the nitrite, ammonia, nitrate have been present and in some cases they achieves very high values. The water of city has the characteristics of a drinking water. After the industrial and public discharges it becomes dark with contenstent of organic matter, nitrite, ammonia, nitrates, iron, etc. This phenomenon happens also with rivers of Kiri, Shkumbini, Gjanica, etc. The enrighing factories have discharged great quantities of matters in suspending state in the waters of Fani Madh, Fani Vogel and Uraka. Gjanica is one of the most polluted rivers Of Albania. The pollution of Gjanica becomes stronger after the discharges of oil-processing Plant of Ballsh.

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FRESHWATER COMPONENT OF THE ARCTIC MONITORING AND ASSESSMENT PROGRAMME

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ABSTRACT

Arctic Monitoring and Assessment Programme (AMAP) was established in 1991 as a major instrument for informational support of the Arctic Environmental Protection Strategy (AEPS) of the eight Arctic countries. The basic methodological approach is to trace fate and effects of the prioritive groups of pollutants from their sources to humans and ecosystems through the environmental compartments. Freshwater component of the programme is considered as an integrated part of the inland environment and includes all basic elements of freshwater ecosystems.

HISTORY AND BASIC APPROACHES OF THE PROGRAMME

AMAP was established as an integral part of AEPS by the Ministerial Conference of the eight Arctic countries (Canada, Denmark, Finland, Iceland, Norway, Russia, Sweden, USA) in 1991 in Rovaniemi, Finland. The main objectives of AMAP are:

- to monitor, assess and report the status of the Arctic environment;
- to document levels and trends of pollutants;
- to document sources and pathways of pollutants;
- to document and assess the effects on the Arctic environment of anthropogenic pollutants, including impact of pollutant fluxes from the sources in both the Arctic and lower latitudes;
- to recognize the importance of the use of the Arctic flora and fauna by the indigenous peoples and to assess their relationship to human health;
- to recommend actions for protecting the Arctic environment.

The AMAP programme is built as far as possible on already existing monitoring and research programme either within the Arctic, or in adjacent areas that affect the Arctic. AMAP is aimed to harmonize these programmes but also to initiate new programmes to fill gaps in knowledge if necessary. By building the AMAP programme on acknowledged procedures it will be possible to compare the results from the different countries and programmes and present overviews that cover the main areas from the mid latitudes to the Arctic.

AMAP has been initiated in a step-by-step manner. As an initial priority, the AMAP is focused on persistent organic contaminants, heavy metals, radionuclides and acidification. To provide an integrated assessment of the Arctic environment, AMAP programme covers the following media: atmosphere, terrestrial, freshwater, marine and human health. Within the first four media the levels of contaminants are to be monitored in different compartments e.g. air, snow, water, soil, sediment and biota. The monitoring programme includes also local food sources and dietary habits.

Methodological approaches used for the AMAP Assessment are aimed to trace fate and effects of the prioritive groups of pollutants from their sources to humans and ecosystems through the environmental compartments as it can be achieved based on available information and level of knowledge. The results of the assessment should allow not only to fulfill the Ministerial request but to highlight gaps in knowledge and to develop proposals for further development of the AMAP monitoring system for filling these gaps.

FRESHWATER MONITORING AND ASSESSMENT

Basins of the Arctic rivers, especially in the Siberian part of Russia, occupy extremely large territories down to 45-50° N. Economic activities in the basins present both potential and actual threat to the Arctic environment. Due to this, rivers are considered as one of the basic sources of pollution for the Arctic region. In the AMAP Assessment, special attention is paid to freshwater pathways of the pollutants including water flow, sediment and ice transport, natural water quality relevant to pollutants' speciation and partition, fate of prioritive pollutants in estuarine zones. Freshwater biota and food webs are considered as accumulators of pollutants and a source of human intake.

CONCLUSIONS

AMAP can be considered as one of the first international attempts of practical implementation of the integrated monitoring and assessment of the total environment including humans with special focus on the interests of the indigenous population. The first phase of the programme identified significant gaps in knowledge of the processes of transport, fate and effects of prioritive pollutants and discrepancies of the existing national monitoring systems to the needs of such a comprehensive assessment. The AMAP Assessment Report, which will be presented to the Ministerial Conference in June 1997, will give not only assessment of the environmental situation in the Arctic based on the current level of knowledge and recommendations for actions but formulate proposals for the second phase of AMAP to eliminate the existing gaps.

DETAILED CHLOROPHYLL-A SAMPLING ALONG THE STREAMS - A VALUABLE PROCEDURE FOR THE ASSESSMENT OF RUNNING WATERS

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ABSTRACT

The pattern of chlorophyll-a concentrations along the longitudinal profile of the transboundary River Elbe based on the results from campaigns of sampling in short site distances within a short time interval is presented. The aim of this paper is to demonstrate the applicability of the data on detailed chlorophyll-a sampling for assessment of running waters.

INTRODUCTION

Phytoplankton is an important biotic component which apparently affects surface water quality and consequently restricts the use of water for management purposes. Phytoplankton biomass is a sensitive parameter suitable especially for monitoring and assessing the impact of nutrient concentration changes in water bodies.
Overgrowth of planktonic algae, as a response to high nutrient levels, is a phenomenon mostly associated with stagnant water bodies. However, data on chlorophyll-a, reaching 300-350 /ug.l-1 in downstream parts of some Czech rivers (Desortová, 1994) indicate the occurrence of high phytoplankton biomass also in running waters (Tubbing *et al.*, 1994 and Garnier *et al.*, 1995)

CHLOROPHYLL-A CHANGES ALONG THE STREAMS

Generally, a gradual increase of chlorophyll-a concentration is observed from headwater to downstream part of water course. Simultaneously, the chlorophyll-a concentrations in longitudinal profiles of streams show a strong seasonal periodicity.
Three data - sets on chlorophyll-a distribution down the River Elbe (36 sampling sites along 1091 km, Fig. 1) were obtained by helicopter sampling flights in different parts of a growth season.
Chlorophyll-a concentrations down the transboundary River Elbe show a typical pattern, i.e. there is an obvious downstream increase in chlorophyll-a from the uppermost part of the river to the tidal stretch, where the values drop off as the Elbe enters the tidal part of the river (Fig. 2). Presented data-sets from various period of season demonstrate the seasonal variability of phytoplankton growth along the stream as well as the differences in inocula inputs from the tributaries.

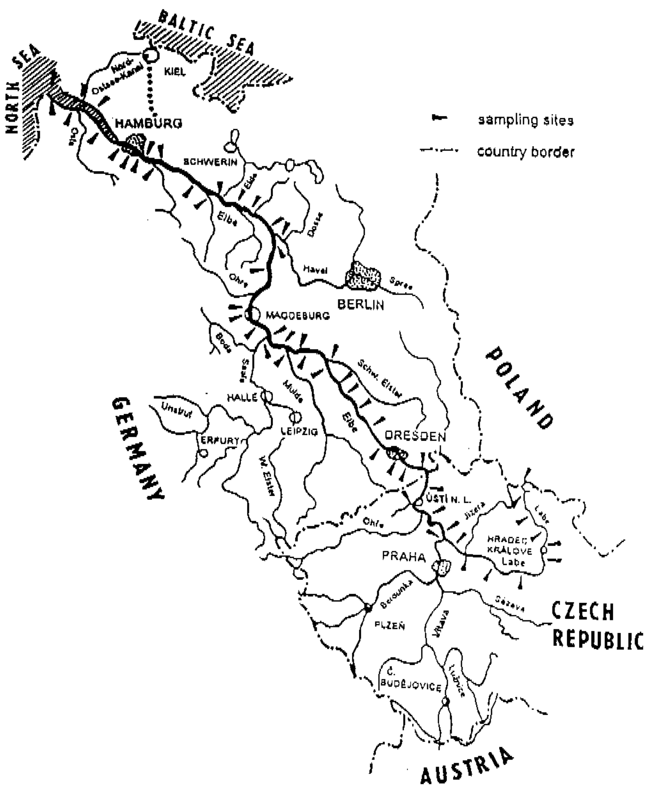

Figure 1: Map of the River Elbe system showing location of the sampling sites.

CONCLUSIONS

The evaluation of spatial and temporal changes of chlorophyll-a content along the longitudinal profiles of water courses based on detailed chlorophyll-a sampling permits:

- the stretches most influenced by eutrophication consequences to be determined

- the possibilities of using stream water to be assessed with regard to the impact extent and the seasonal influence

- the monitoring network design for a particular water course to be optimized.

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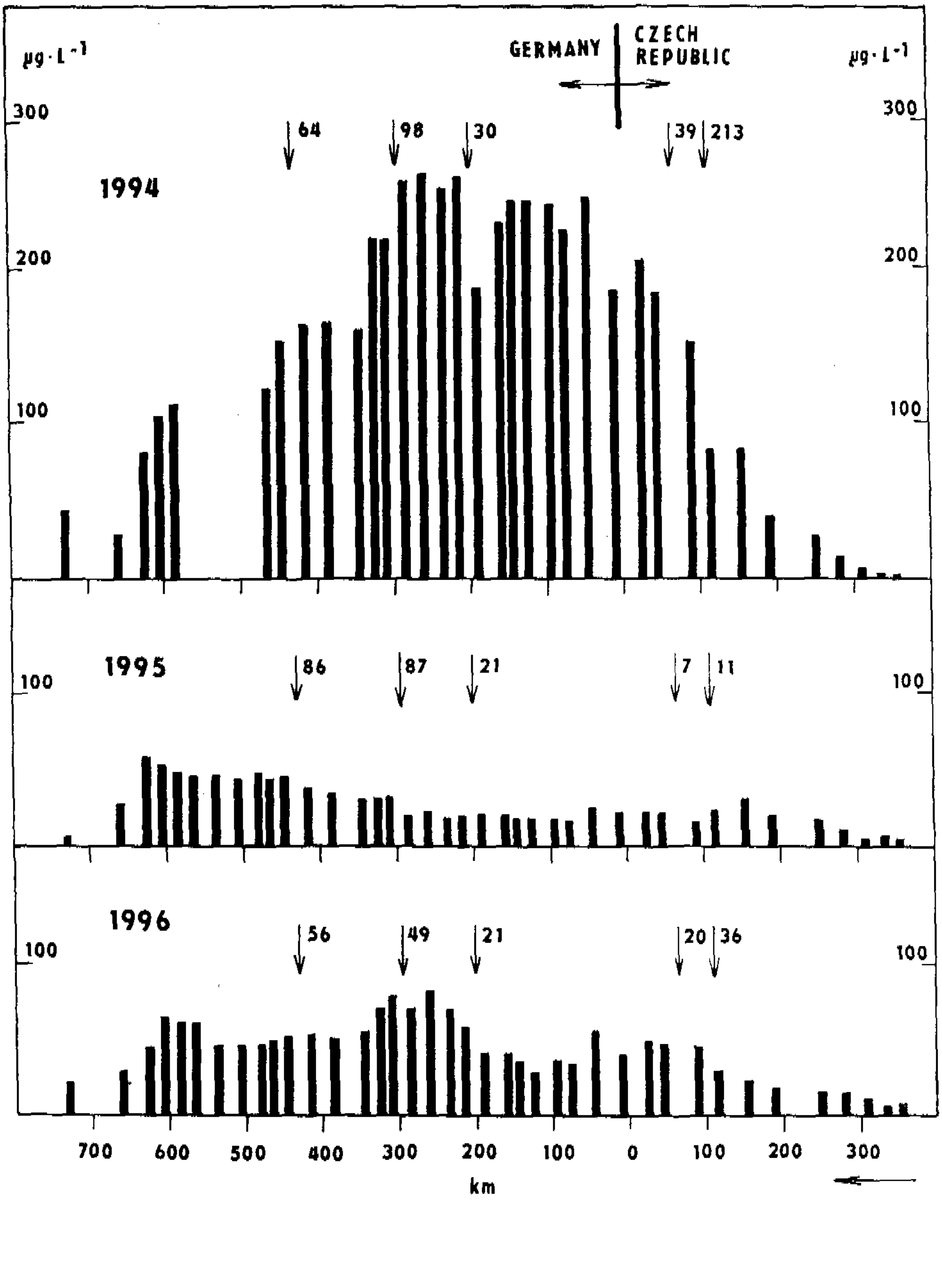

Figure 2: The distribution of chlorophyll-a along the River Elbe (based on helicopter sampling flights on 11-12 May 1994, 11-13 September 1995, 22-24 April 1996).

MONITORING STRATEGY TO SAFEGUARD THE WATER SUPPLY IN THE CASE OF ACCIDENT - RELATED RIVER POLLUTION

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ABSTRACT

The Sandoz accident provided fresh impetus to improve pollution control on the Rhine. As a result, the riparian states of the Rhine initiated the „Rhine Action Program". The waterworks developed tools for monitoring and assessing the effects of peak load on bank filtration and water supply. The additional tools of the waterworks are based on the International Quality Monitoring Program and the Rhine Warning and Alarm Service.

COMPONENTS OF THE PEAK LOAD MONITORING SYSTEM

After Sandoz accident in 1986, an efficient system for monitoring and assessing the possible effects of peak load on bank filtration has been developed . The essential components of this system will be described in the following.

QUALITY MONITORING PROGRAM

The network along the Rhine for monitoring water quality, continues to be of prime importance. This comprises the monitoring stations set up by the competent authorities in the relevant states, and is supplemented by joint monitoring systems of the larger Rhine waterworks. The scope of the monitoring and intervals at which it is carried out are coordinated in such a way that almost continuous monitoring is achieved. Daily reference samples make it possible to pinpoint the site of discharge of pollutants.

RHINE WARNING AND ALARM SERVICE

Six central reporting stations on the Rhine and two on the Mosel, are responsible for transmitting information on the sudden pollution of waters to the component authorities and to the waterworks per fax. The international warning and alarm service is supplemented by regional warning and alarm services, which are integrated in the reporting system.

RHINE ALARM MODEL

Essential factors are the time of arrival of a pollutant wave at any observation point, the maximum pollutant concentration curve along the river and the concentration curve of the pollutant wave. For this reason the Rhine Alarm model has been developed to predict the course of a pollutant wave in the Rhine. This model was experimental tested and improved and exists in the form of an operator-friendly PC-program.

BANKFILTRATION MASS TRANSPORT MODEL

An important objective within the framework of a waterworks joint research project was the preparation of groundwater models to simulate mass transport processes between the river and the wells. As the model is rather complex, in order not to have to activate calculations each time an accident happens on the Rhine, a series of one to three-day peak loads was simulated, taking as an example a relatively extreme case. The simulation results cover a wide range of possible cases which can be supplemented and extended as required.

METHODS OF ANALYZING SUBSTANCE BEHAVIOUR

To allow the waterworks to take any measures which may be neccessary to limit the effects of accident-related water pollution, it is important to have knowledge of the degradation behaviour of the substances during passage through the soil. The substance data sheets do not, as a rule, provide adequate information in this regard. Therefore, test filters, with which degradation processes during passage through the soil can be simulated in quick motion, were also tested in the course of the joint waterworks research project. Since then, test filters have always been held in readiness at the larger waterworks and are an essential component of the monitoring system.

ADDITIONAL TESTING FACILITIES

Should the procedures for balancing out concentrations and the degradation processes during passage through the soil not provide adequate protection against impairment of the quality of well water, among other things, the question as to the behaviour of the substances in the existing treatment plants arises. Conventional laboratory tests, e. g. to determine the adsorption behaviour or the behaviour under oxidising conditions, can be employed for this purpose.

MEASURING POINTS BETWEEN THE RIVER AND THE WELLS

As a rule, the flow time through the soil in the case of bank filtration plants on the Rhine is several weeks. This means that the time necessary for making use of the monitoring system described is available. It is also possible to monitor substance dispersion via special measuring points between the Rhine and the wells.

CONCLUSIONS

The system for monitoring peak loads has improved safety as far as the potable water supply from bank filtrate is concerned. The components described have been tailored to the situation in the Rhine catchment area. In case of future accidents, it will be easier for the waterworks and authorities concerned to reach agreement on any preventive measures which may be necessary.

THE IMPLEMENTATION OF ECOTOXICOLOGICAL METHODS TO THE SYSTEM OF ROUTINE MEASUREMENTS OF THE SURFACE WATERS IN THE SLOVAK REPUBLIC

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ABSTRACT

Nowadays, in the Slovak Republic only traditional chemical monitoring is performed, along with saprobic index and coliform bacteria as biological determinands. It has been recognized that such approach does not cover the information needs for the aquatic environments in the decision making processes. Therefore, the key points of the project will be to implement the eco-toxicological methods and biological monitoring to national systematic monitoring programme of surface water quality for both Váh and Ipel River Basin. As the important part of the project, will be the contribution of strategy for water quality monitoring of transboundary rivers (Ipel River basin).

THE OBJECTIVES OF THE PROJECT

The Project sutdy will fulfil the objectives as follows:

- Implementation of the biological and ecotoxicological methods to support the routine surface water quality monitoring programme in the Slovak Republic,
- To ensure the compatibility of the Slovak surface water quality monitoring programme system to the systems used in the EU countries,
- To contribute to a strategy for water quality monitoring of transboundary rivers - Case study in the Ipel River Basin.

NEEDS FOR ECOTOXICOLOGICAL METHODS

The Slovak Republic could be characterized as country with intensive development of industry. There are large facilities focused on production of basic chemicals, agrochemicals and petro-chemistry.
In the Slovak Inventory of Produced and Imported Chemical Substances, 250 chemicals above 1000 t/year of production/import (170 in amount higher then 10.000 t/year) have been identified. But nearly no one of them are regulary monitored in main Slovak rivers. Data on chemicals with respect to their long-term ecotoxicity and environmental are also scarce, if even existing. It is therefore understandable that a keen interest in developing both chemical monitoring techniques and biological monitoring with ecotoxicological studies is needed.

STUDY AREAS

The project will be realised in two parallel case studies:

A. Váh River Basin
   Four Hot Spots have been selected to represent relevant types of industries/waste water effluents, which are located in the river basin:

- LiptovskýMikuláš - tannery/municipal WWTP
- Liptovský Hrádok - electro-telecommunication industry
- Ružomberok - pulp and paper industry
- Šal'a - chemnical and agrochemical industry

B. Ipel River Basin
In the basin wide range of different pollution has been identified (glass industry, heavry machinery industry, food production and also untreated municipal waste waters). The case study is concentraded on complex evaluation of water quality using integrated approach (ambient water quality and effluents). Biological monitoring will by used for more comprehensive knowledge in selected sites.

CONCLUSIONS

In the project complex evaluation of water quality using integrated approach will be done on two selected rivers. Furthermore, project results should support development of legislative tools on three levels:
- single chemicals discharge control
- toxicity limits for waste waters
- classification of surface waters based on toxicity and biological determinands.

A specific result will be improvement of monitoring knowledge of local experts by training in RIVM Bilthoven (The Netherlands).

MONITORING FOR POLICY EVALUATION ON A REGIONAL SCALE

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ABSTRACT

A method is developed for the monitoring and evaluation of the water management on a regional scale. Provinces and district waterboards work together in this project. It concerns all respects of water management, surface water and groundwater, quantity and quality of water, morphopoly and maintenance of watercourses, biological assessment. In 1996 the methods are developed, in 1997 the proposed methods will be tested. From 1998 on the implementation will start routine monitoring will be introduced step by step.

DEFINITION

Monitoring for policy evaluation is defined as the activity of gathering and processing data and reporting them in such a way, that relevant information is generated to review and, if necessary, adjust the policy itself as well as the policy implementation.

WATER MANAGEMENT PLANNING IN THE NETHERLANDS

The provinces formulate the policy for the management of regional surface waters and groundwater in The Netherlands, based on the National policy plan. The district waterboards carry out this policy for surface water, the provinces themselves do the same for groundwater. Every four years the policy plans are reviewed and updated. A policy evaluation, based on monitoring, precedes this review.

THE PROJECT

A method is developed for the monitoring of the progress that is made in the implementation of water management policy for regional waters and groundwater. The aim is to develop a basic set of indicators, that all provinces will use in the same way. Each province is free to complete this basic set according to her own needs or specific situation.
The project concerns all aspects of water management, surface water and groundwater, quantity and quality of water, morphology and maintenance of watercourses, biological assessment. For all these aspects, indicators and yardsticks are developed to describe the situation and determine the ranking of the indicators on a scale that is deduced from the policy objectives. The method focuses on source indicators that describe the load on the watersystems, and on effect indicators that characterize the physical, chemical and biological condition of the watersystems in comparison with the objectives. The source indicators are related to the effect indicators in dose-effect chains or networks, and can be influenced by the policy. The presentation of the results must be simple and clear, easily understandable for administrators.
As much as possible, methods are applied that are already developed or in use, e.g. for the national monitoring programme. The increases the enthusiasm to join in.

PHASING OF THE PROJECT

In the first phase of the project, lasting until the end of 1996, the method is developed. This is done in five subprojects. The first deals with the monitoring scale, the definition of the basic areas and categories for measurement, the density and location of measuring points, and aggregation methods. The second concerns the quality of surface water and submerged soils, emissions and biological assessment. The third deals with morphology and maintenance of watercourses, and surface water quantity aspects, the fourth with groundwater quantity and quality, as well as the item desiccation of nature reserves. Last but nog least, the fifth subproject develops the information system and instruments needed.
In the next phase, that will start in 1997, the method will be tested in pilot projects and promoted. From 1998 on the implementation will start and routine monitoring will be introduced step by step.

ORGANIZATION

The project includes organizational as well as technical aspects. Provinces and district waterboards gather and process data on watersystems. Data have to be exchanged among several authorities, information has to be generated in comparable ways, the same definitions and codes used.
Therefore representatives of the national water management authorities, provinces and district waterboards work togester in this project. In addition to this, every province has established a working group in which all authorities dealing with water management participate. They verify the feasibility of the draft methods in their area and give feed back.
Also, in this project much attention is paid to communication, with the people who will carry out the monitoring, as well as the administrators who will use the results.

ECOLOGICAL EFFECTS OF CHANGES IN THE WATER QUALITY OF THE RHINE

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INTRODUCTION

Plants and animals make demands upon the conditions of their surroundings. These species can only grow and develop competitively if those requirements are met. An important precondition for the existence of plants and animals is the availability of a natural habitat (for example sand beds or silt beds, hard objects, or places where erosion or sedimentation occurs).
Besides habitat factors, a number of water quality parameters are particularly important; the main ones being:

- limpidity: the availability of light.
- oxygen: the oxygen content and its fluctuations.
- salinity: content of salts (as a permillage).
- nutrients: content of the nutrients phosphate, nitrogen and silicate.
- warmth: the water temperature.

During the past 50 to 100 years, a number of these physicochemical factors have undergone great changes, with consequent modifications to the composition of species. At first, this concerned the great deterioration of the water quality in the 1950 to 1975 period, which involved an enormous setback in the composition of species. Since 1975 however, a clearly positive turn has been observable in quality and the composition of species. The current water quality is no longer by definition the restrictive factor for the development of flora and/or fauna.
Nevertheless, a number of factors, especially the temperature and the salt content, have changed so drastically and permanently that the original situation has not yet been restored. Because of this, the original occupants of the Rhine trail behind compared to a number of immigrants or exotic organisms from warmer and more saline areas.

DEVELOPMENTS

The development of the water quality has great influence on the developments in the variety of species of invertebrates, among other things.
This variety of species is currently increasing again, largely as a result of the return of original occupants (larvae and caddis worms). For example, the mayfly (*Ephoron virgo*) has been spotted again after many years of absence.

The variety of species is also being enlarged by immigrants. This is due to the permanently changed factors mentioned above, for example temperature and salinity. Since the average chloride content has increased by factor 15 and the average water temperature has increased by approximately 2-3° Celsius, species which tolerate salt and thrive on heat and salt have a stronger competitive position. Many of these species come from a brackish environment and southern areas (immigrants or organisms from exotic areas). In addition, the continuously increasing integration of water systems which used to be separated, plays a part in the distribution of flora and fauna. The number of immigrants is small in theory, though some of these are present in large numbers and dominate the biocoenosis.

MACROFAUNA DEVELOPMENTS

Between 1970 and 1980, the freshwater wood louse (*Asellus aquaticus*) was the most important invertebrate on the stones. From 1980 to 1985, this position was taken over by the dance fly larvae with the genera *Cricotopus* and *Dicrotendipes* as the most important representatives. In 1987 and 1988 an immigrant dominated for the first time. This was the Tiger gammaridae (*Gammarus tigrinus*).
Since 1988, two sorts of basket mussels have been found in the river bed of the Dutch branches of the Rhine. These are the twisted form *Corbicula c.f. fluminalis* (upstream) and the coarse ribbed *Corbicula c.f. fluminea* (downstream; tolerates brackish water). Both species are immigrants.
In 1990, the Caspian mudshrimp (*Corophium curvispinum*), also an immigrant, was still a (sub)-dominant species. This mudshrimp is a formidable competitor to other organisms which live on stones. This mudshrimp is very dominantly present nowadays, while the populations of other organisms which live on stones, like the zebra mussel, have greatly decreased.

After 1990, the number of species increased even further. The mayfly *Ephoron virgo*, which has been mentioned before, returned in 1991. In 1992, the Gammaridae *Echinogammarus ischnus* (immigrant) was spotted in the Netherlands for the first time. It is striking that in the two years after the accident near Sandoz (1987, 1988), the density of invertebrates was significantly lower than before and in later years. This may well have accelerated the dominance of immigrants.

OTHER DEVELOPMENTS

The composition of species of the other flora and fauna of the Rhine is also influenced by changes in the water quality, but especially by developments in various habitat factors.
The phytoplankton mainly consists of large amounts of diatoms and green algae, caused by the fact that contents of nitrogen and phosphate are not limiting to growth. Water plants and riparian plants also have a negative correlation with salinity, alkalinity, phosphate, nitrogen and turbidity. However, physical factors such as water level fluctuations, flow and decreased limpidity also play an important role for fish, birds and amphibians.

Further recovery of the river fauna depends on continued repression of the pollution, but also on the development of more natural habitats and an increase in morphological dynamics. Improvement of the design of the river, involving creating waters rich in plants, gravel banks, slow streaming water, secondary channels, flood plain forests and other biotopes, would offer new chances for many flora and fauna species.

PHYSICO-CHEMICAL WATER QUALITY INDICES

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ABSTRACT

A study was carried out on the composition and use of physico-chemical water quality indices. Information on 28 indices was collected. Most indices are composed in three steps: parameter selection, standardized scoring of parameters (sub-indices) and aggregation of the sub-indices. The parameters most frequently used are: O_2, BOD, COD, NH_4^+-N, PO_4^{3-}, NO_2^-, NO_3^- pH, suspended matter and Coli bacteria. The sub-indices can be expressed as a nominal value, a dimensionless numeric value or a numeric value with dimension. The aggregation of the sub-indices can be carried out according to several (mathematical) formula. Finally the index is presented as e.g. a number, a class, a verbal description, a unique symbol or a colour.

INTRODUCTION

For the assessment of water quality, various organisations of different nationalities involved in water quality control, use or have previously used (physico-)chemical water quality indices (WQ Indices). Information on 28 WQ Indices was collected from literature (see list of references). A description is presented of the compostion of WQ Indices in general and of the WQ Indices studied in particular.

A WQ Index is defined as:
A simple expression of a more or less complex combination of a number of water quality parameters which serves as a measure for water quality. The index is presented as a number, a class, a verbal description, a unique symbol or a colour.

The use of the WQ Index lies in its ability to reduce and simplify a large amount of information. This makes the information easier to understand and use both for the water quality manager and for persons less directly involved in quality control. The users should realize, however, that underlying information is lost when using WQ Indices.

RESULTS

GENERAL COMPOSITION OF WQ INDEX

A WQ Index is composed in three consecutive steps:

1 selection of the parameters;
2 determination of the quality scores per parameter: the sub-indices;
3 determination of the WQ Index by aggregation of the sub-indices.

SELECTION OF THE PARAMETERS

Theoretically, a WQ Index can comprise between two and an infinite number of water quality parameters. The choice is made on the basis of circumstances, standards and criteria characteristics of time and location, and of expert opinion.

DETERMINATION OF SUB-INDICES

The sub-indices (step 2) can be determined using various methods:

1 After comparison of a parameter value with a standard or criterion, the parameter is given a nominal or numeric valuation.
2 The parameters are converted into a dimensionless number by means of calibration diagrams. In this case for each parameter a separate diagram has been developed which indicates the correlation between the parameter value and quality score. The diagram scales are equal for all parameters. The quality score usually is between 0 and 100.
3 An alternative to the calibration diagram is the calibration table. In these tables, the parameter values are also related to a quality score.
4 For each parameter a mathematical formula has been developed, which converts the parameter values according to various scales, whereby the parameter values retrain their original units.

DETERMINATION OF THE WQ INDEX

Aggregation of the sub-indices to form an overall WQ Index can be made by means of the aggregation formulas listed in table 1.

NO.	METHOD	FORMULA
1	unweighted average	$I = \frac{1}{n} \sum_{i=1}^{n} q_i$
2	weighted average	$I = \sum_{i=1}^{n} q_i w_i$
3	unweighted geometrical average	$I = (\prod_{i=1}^{n} q_i)^{1/n}$
4	weighted geometrical average	$I = (\prod_{i=1}^{n} q_i)^{w_i}$
5	minimum sub-index	$I = \min(q_1, q_2, \ldots q_n)$
6	maximum sub-index	$I = \max(q_1, q_2, \ldots q_n)$
7	modified unweighted average	$I = \frac{1}{100} (\frac{1}{n} \sum_{i=1}^{n} q_i)^2$
8	modified weighted average	$I = \frac{1}{100} (\frac{1}{n} \sum_{i=1}^{n} q_i w_i)^2$

Table 1 Aggregation formulas for the calculation of a WQ Index from sub-indices

I = WQ Index; n = number of parameters; q_i = quality score (sub-index) of parameter *i*; w_i = weighting factor of parameter *i*.

COMPOSITION OF THE WQ INDICES STUDIED

The 28 WQ Indices studied are composed of a different number of parameters, varying from 3 to 72. In some cases the choice and number of parameters is to be determined by the user.

Practically all WQ Indices use three or more of the following parameters: O2, biochemical oxygen demand (BOD), chemical oxygen demand (COD), NH4+-N, PO43-, NO2- and/or NO3-, pH and suspended material (and Coli bacteria). The three parameters combined most frequently are O2, NH4+-N, BOD and/or COD. Less frequent use is made of parameters such as metal concentrations, Cl- and pesticides, temperature, electric conductivity (EC) and turbidity.
Table 2 gives a summary of the parameters used per WQ Index.

O2, BOD and/or CZV en NH4+-N	Practically all WQ Indices
One or more of the parameters PO_4^{3-} and NO_2^- and/or NO_3^-, pH and suspended material (and Coli bacteria)	Practically all WQ Indices
Metals	Prati-Index, POLIN, RIVM/CPCB-Index, CUWVO-Index
Pesticides	RIVM/CPCB-Index, CUWVO-Index
Cl^-	Prati-Index, POLIN, Schalekamp-Index
Oil	RIVM/CPCB-Index, CUWVO-Index
PAK	RIVM/CPCB-Index, CUWVO-Index
DOC	Schalekamp-Index, "Wing-diagram"
Temperature	Bach-Index, NSF-Index, Smith-Index, Weighted Solway-Index
Turbidity	RIVM/CPCB-Index, NSF-Index, EPA Region VIII-Index, Smith-Index, Weighted Solway-Index
EC	RIVM/CPCB-Index, Bach-Index
Other parameters	Prati-Index, POLIN, EPA Region X-Index, RIVM/CPCB-Index, GREMU-Index, CUWVO-Index

Table 2 Summary of parameters in the WQ Indices studied

Table 3 gives an overview of the number of parameters used, the use of calibration diagrams and/or tables or mathematical forumulas and the aggregation formula used per WQ Index.

WQ Index	Number of parameters	Calibration diagrams, tables or mathematical formulas	Aggregation formula
IMP-Index	3	-	**
Gem. Benelux-Index	3	-	**
Lisec-Index	4	-	**
VVM-Index	5	-	**
Saubain-Index	7	-	**
Pollution-Index	5	diagrams	1
Prati-Index	8/13	formulas	1
POLIN	6	formulas	1
EPA Region X-Index	*	diagrams	**
Dunnette-Index	6	diagrams	2
Oregon's Index	6	diagrams	2
Ross-Index	5	table	2
Radar-plot	*	diagrams	1/2
GREMU-Index	*	diagrams	2
Provencher-Index	72	diagrams	2/4
KWF-Index	4	formulas	3
Bach-Index	8	diagrams	4

NSF-Index	9	diagrams	4
EPA Region VIII-Index	5	*	4
TM-Index	8	diagrams	4
Smith-Index	9/11	diagrams	5
GQA-Index	3	-	5
FC-Klasse-indeling	3	-	6
Schalekamp-Index	5	formulas	6
(Ongewogen Solway-Index)$^{\alpha}$		(-)	(7)
Gewogen Solway-Index	10	diagrams/table	8
CUWVO-Index	8/9	-	-
LWA-Index	3β	–	-
"Wing-diagram"	4	-	-

* = to be determined by user/not specified in reference

** = determined by means of weighted summation, without the divison by n (= number of parameters), contrary to the unweighted average (formula 1)

α = the Unweighted Solway Index data is listed in parentheses, because no evidence of its use was found in the references studied

ß = in addition to the 3 chemical parameters, the LWA Index is also determined by saprobity

Table 3 *Overview of the number of parameters used, the use of calibration diagrams, tables or mathematical formulas and the aggregation formula used per WQ Index*

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FROM DATA TO INFORMATION: THE WATER DIALOGUE

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ABSTRACT

The Aquatic Outlook information system, the WATER DIALOGUE, is a presentation module for the Aquatic Outlook project. The application provides processing facilities and a clear, reproducible presentation of data relating to the condition and use of the water systems in the Netherlands. In addition to information regarding the physical, chemical and biological quality of the waters, the WATER DIALOGUE also provides emission indicators, the costs involved in water policy and information with respect to the functional aspects.
The WATER DIALOGUE is implemented as an interactive PC application, which enables the user to define policy queries by means of selecting groups of variables, standards and even future developments. The results, for both the present and the future, are presented using a number of methods. The WATER DIALOGUE not only offers starting points for the analysis of bottlenecks in water management, but also for policy evaluation and preparation.

AQUATIC OUTLOOKS AND THE INFORMATION CYCLE

Policy makers who ask themselves "What's the present water quality in the Rhine?", can of course be supplied with as much information as they require with respect to all the substances measured in the Rhine annually (approximately 250). Each individual location alone provides approximately 5000 water quality measurements. Whether or not these are of any use is not even the question, but an illusion. Staring at long tables is not a healthy basis for new policy. Now that integral third water management has become communal property, the demand for information has become greater than ever. The National Policy Document on Water Management focused on the water system approach and formulated target values for long-term water management. As a result there is now a requirement for information that quantifies and defines those target values. The Aquatic Outlook project provides the initiative for this. The information strategy followed in its implementation was shaped by descriptions of water systems on the basis of a selected set of so-called *target variables*. Numeric values reflecting the present situation, values from previous years, prognosis values and standards or calibrated reference points being represented in relation to one another, the so-called yardstick approach.
The enormous quantity of data collected in this manner has in the course of time been recorded in numerous reports, work documents and the WATER DIALOGUE. Using the WATER DIALOGUE, the presentation module for the Aquatic Outlook project, the data collected can be processed and converted into unambiguous information. The WATER DIALOGUE plays an important role in the information cycle by processing data and providing information.
In this article the background of the WATER DIALOGUE is explained.

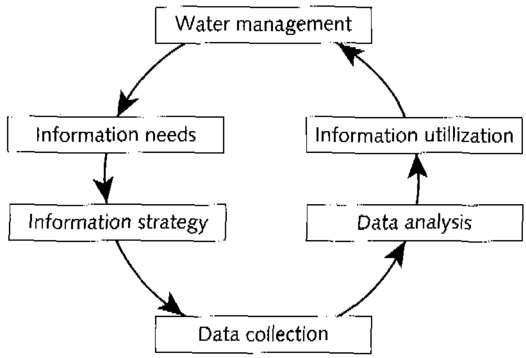

Figure 1: The information cycle

FROM DATA TO INFORMATION

The Aquatic Outlook information system, the WATER DIALOGUE, was developed by the Directorate General for Water Management in order to retrieve the great diversification of data required for integral water management in the Netherlands in an uncomplicated manner.
The WATER DIALOGUE consists of a database containing all the relevant data for the target variables, and software for presenting that information. It is an interactive PC application with which the user can indicate the information that is required. To achieve this, users can make selections from the database for assessment, clustering, aggregation and presentation. In this manner the user is able to answer policy queries by means of selecting a single target variable or a group of related target variables, by selecting a set of standards or calibrated reference points, and by indicating the water system for which information is required. A choice can also be made from the results of various possible future developments that have already been analyzed within the Aquatic Outlook project.
The selected data is assessed against the selected standard. The result of this assessment can then be presented in a variety of manners. One eye-catching method of presentation is the so-called "Water Mondrian" (© 1995 ABC/Mondrian Estate/Holzman Trust. Licensed by ILP), see figure 2. The Water Mondrian is a highly abstract map of the Netherlands, which in terms of design is very similar to the well-known paintings by the artist Piet Mondriaan. The water systems are represented as blocks; the result of assessments are expressed as colours. Using this method of presentation a vast quantity of data can be converted into conveniently arranged and aggregated information in a relatively simple manner. The WATER DIALOGUE generates information with which bottlenecks in water management can be signalled, subsequently providing starting points for policy evaluation and preparation.

The Water Mondrian displays information at a highly abstract level. The WATER DIALOGUE however also offers an opportunity for exploring the underlying information. Accordingly a "score list" can be called up from within the Water Mondrian. This indicates in a matrix the extent to which clustered target variables and individual or aggregated water systems deviate on average from the standards or calibrated reference points. In the "radial diagram", a figure based on the Amoeba, the deviation from the standard or calibrated reference point for individual target variables can be viewed in a circular pattern. In the "water index graph" the chronological development of a cluster of target variables is displayed on a scale of 0 to 100. Finally, the "time graph" displays the information at the lowest level of the WATER DIALOGUE; the chronological development of one variable and one water system in the original units.

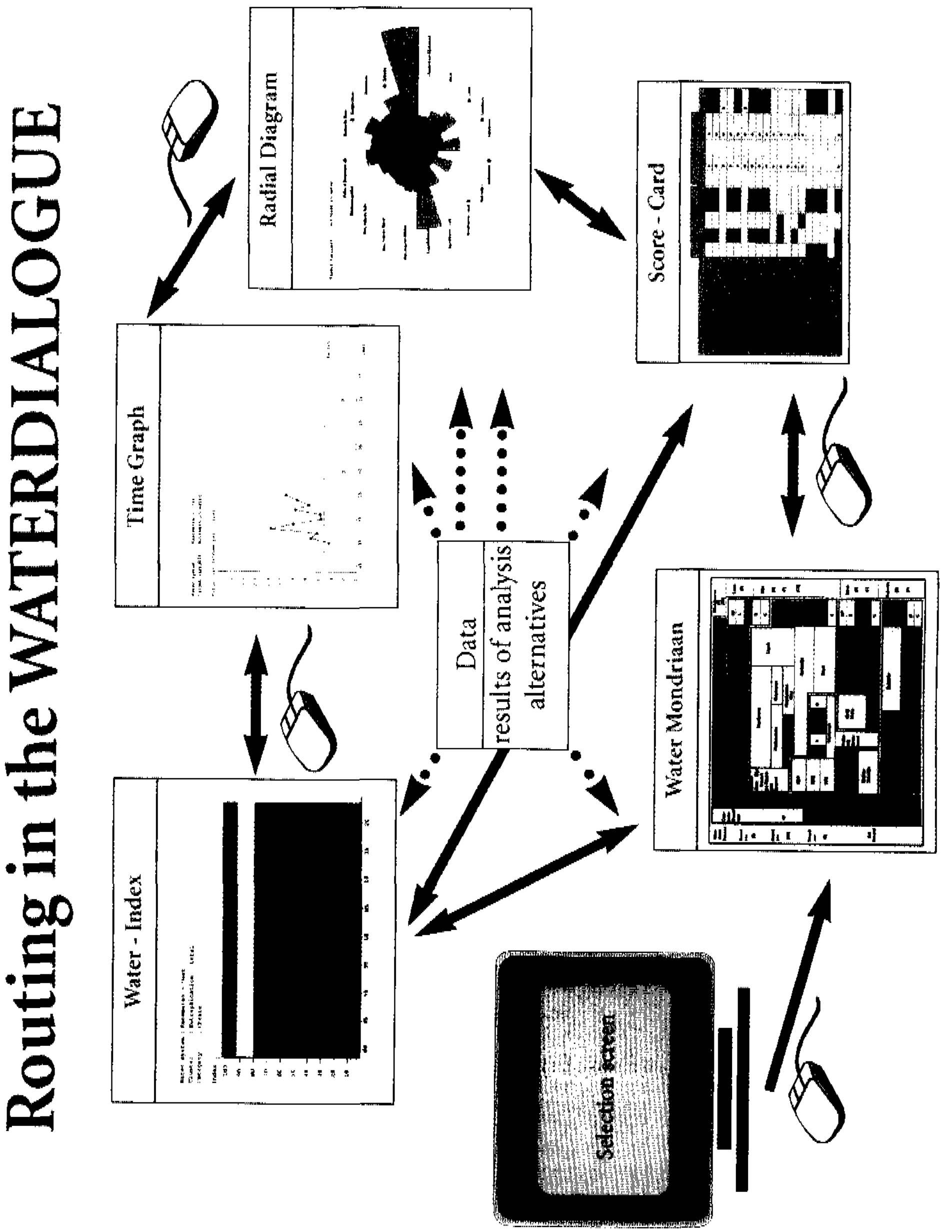

Figure 2: Presentation options in the WATER DIALOGUE

RECORDING INFORMATION

The quantitative information obtained from water system indicators is collected in the WATER DIALOGUE. In addition to information about the physical, chemical and biological quality of the waters, the database also includes information regarding inputs, economic aspects of water management and the use of water. For each target variable, values are recorded with respect to the previous situation, for the present situation (mostly 1994) and for prognoses for the future based on the Aquatic Outlook analysis alternatives for the years 2000, 2015 and in some cases also for 2045.
The information regarding the previous situation and the present situation is based on monitoring and other sources from various institutions. As assessment against standards is the basis for the WATER DIALOGUE, only one value to be assessed (test value) is recorded every year. The prognosis values used to provide an indication of the future are derived from mathematical models or "expert judgement".

A mutual influence exists between monitoring and the choice of target variables. To a great extent monitoring provides the basic information, and this can be evaluated and analyzed with the WATER DIALOGUE by assessing it against standards or calibrated reference points and analyzing trends. Recent information is supplemented on an annual basis. For many target variables systematic monitoring is already performed. For chemicals, for example, the concentrations of a growing number of substances have been regularly measured for some time. For inputs the figures provided by the Central Bureau of Statistics (water purification plants) and the Integral Water Management Committee-CIW (specific industrial sources) are of importance. Diffuse sources are quantified using estimation methods. Systematic monitoring of the biology in national and regional waters was commenced several years ago. A great deal of shape still has to be given to the monitoring of the physical aspects. Initiatives in this respect have already been taken within the CIW framework and the Regional Water System Reports, amongst others. Information regarding the economic aspects of water management have been collected on a project basis.
On the one hand, the developments in the points of attention for water management or the findings of studies such as that performed by Aquatic Outlook can give cause to changes in the parameters being measured, and on the other hand, the evaluation of monitoring data can give cause to the changing of target variables.

The assessment of standards or calibrated reference points is taken as the starting point for assessing target variables. In doing so use is made of both policy established objectives (such as chemical target values and limit values), and the calibrated reference points defined within the framework of the Aquatic Outlook project but which do not possess policy status. Examples are the basis year 1985 as the calibrated reference point for inputs and the natural target situation for biological indicators. The numerical values of these standards and calibrated reference points are also recorded in the WATER DIALOGUE.

CALCULATION METHOD AND INTERPRETATION

Combining a great quantity of data in the WATER DIALOGUE can lead to one overall assessment regarding the functioning of the water systems. Calculation methods have been designed for clustering target variables and aggregating the water systems into a higher level (see insert) (Stutterheim, E.; Duijts, H., in prep). The extend of exceeding or failing to meet standards, i.e. the typical ratio, is calculated for each individual target variable. The average standard exceedance rate is then calculated from these typical ratios in a number of intermediary stages, and for the benefit of presentation expressed as one colour. A blue colour indicates that the target variables meet the standards against which they were assessed. The colours change from blue to green, yellow, orange and finally to red in relation to the extent of the exceedance rate from the standard.

To convert the test values and prognosis values of individual parameters into a value for group parameters, the following steps are followed:

Step 0 Calculation of target value (the value to be assessed) from measurement values. This calculation stage is target variable dependent. For example, for phosphate the summer six-monthly average is calculated. The calculation step is performed outside the WATER DIALOGUE and the result subsequently entered into the Aquatic Outlook database (one figure per target variable/water system combination per year).

Step 1 Determining the distance from the test or prognosis value to the standard or calibrated reference point.
The result is the number of times the standard has not been met or exceeded, the typical ratio (TTX of times to objective).

Step 2 Limiting TTX between 1 and 11.
A TTX smaller than 1 is assumed to be 1 (to prevent compensating for exceeding standards by undermining); values greater than 11 are made equal to 11 (scale limitation for the benefit of step 3).

Step 3 Scaling between 0 and 100.
Conversion of the TTX to a water index figure between 0 (great deviation from standard) and 100 (everything good): water index figure = 110 - (10 * TTX).

Step 4 Aggregation step.
The 50 percentile value is calculated from the individual water index values of the individual target variables; if there are less than 6 target variables the average is taken.

Step 5 Conversion of the calculated water index value (of aggregated information) into a colour for the benefit of presentation.

water Index	**colour**	**average exceeding rate from standard value reference point**
< 53	red	typical ratio > 5.7
53 - 79	*orange*	*5.7 = typical ratio < 3.0*
80 - 92	yellow	3.0 = typical ratio < 1.7
93 - 100	green	1.7 = typical ratio < 1
100	blue	typical ratio = 1

The calculation procedures for aggregation are established in the WATER DIALOGUE. In principle, on the basis of the calculation method described, every form of aggregation is possible. A condition however is that standards or calibrated reference points are available for the target variables to be combined.

In case the Water Mondrian illustrates a deviation from the standard, it is possible to determine the cause of the deviation by zooming in on the radial diagram. A slight average standard deviation might be caused by a significant deviation from the standard by one parameter, or by a very slight deviation from the standard by all parameters. Furthermore, by zooming in it is also possible to reveal that the colour might be based on a varying number of relevant target variables.

Interpretation of the deviation from standard values as presented in the WATER DIALOGUE is relatively simple where the standards used can be assumed to be relatively well-known, such as *limitation and target values for pollution. This is somewhat different when use is made of the* calibrated reference points established specifically for the Aquatic Outlook project, such as biological references for example. Knowledge of the background that led to the definition of these values is essential in order to be able to rate an assessment result at its true value. This background information is established in Aquatic Outlook reports and in the Aquatic Outlook meta information system.

Integral water management demands information that provides an integral impression of the condition and use of the water systems. The existence of various types of standards also means that target variables can only be clustered in case standards or calibrated reference points are applicable that are mutually comparable. If for example the standards for ecology and economy are not consistent it is not worthwhile providing an integrated illustration of those elements. The same applies if water systems are aggregated. If for example salt and freshwater water systems are aggregated, it would only be ultimately worthwhile if the standards applicable to them remained consistent.

A wide variety of assessment parameters exists for the indicators used in the Aquatic Outlook project for depicting water systems. These too are only relevant to specific elements of integral water management, without there being any question of a relationship to the standards or calibrated reference points used for other components. Thus there is no relationship between the present water quality standards and the objectives of emission reductions. This results in a bottleneck when clustering target variables into integral information. In order to nevertheless meet the demand for integral information, several so-called combination sets of standards or calibrated reference points have been included in the WATER DIALOGUE, thereby making the aggregation and clustering of data possible.

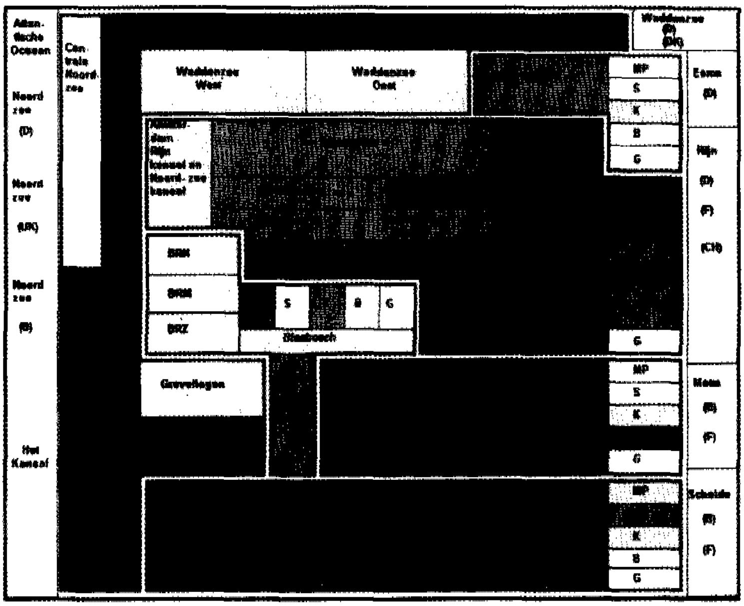

Figure 3: The Water Mondrian for all system target variables for the years 1993-1994, assessed against target values and biological references. LP = Lakes and pools; D = Ditches; C = Canals; B = Brooks; G = Groundwater; NTR, CTR and STR = Northern tidel rivers, Central tidel rivers and Southern tidel rivers.

EXAMPLES

In the Water Mondrian the condition and use of the water systems in the Netherlands is visualized by the use of colours. This can be performed for each parameter or for groups of parameters. Figure 3 displays a Water Mondrian for all target variables indicative of the condition of the water systems. The parameters used for the national waters relate to the chemical condition (from nutrients to organic micro-contaminants), biological target variables (obtained from various

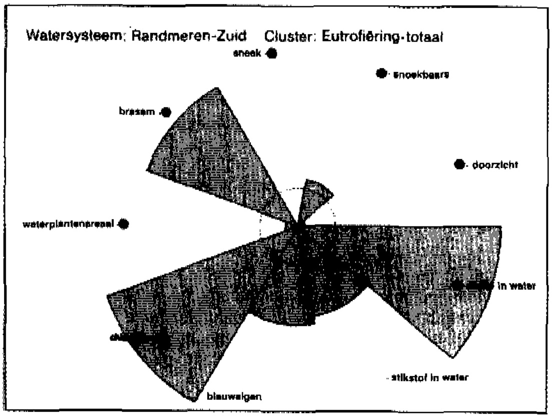

Figure 4: Radial diagram of the Randmeren-East area for the problem field eutrophication.

Amoeba studies), and physical variables (turbidity and a number of indicator for habitat type). For the regional water types only the information available for chemical parameters was used. In the WATER DIALOGUE regional waters are in the first instance depicted at the level of water types. Via the score lists it is possible to zoom in to the level of an individual regional water. The colour in the Water Mondrian illustrates a clustering of underlying information. At a glance it is then possible to determine which water systems meet the standards or calibrated reference values that have been established (in this case the target values and reference values). With the aid of the WATER DIALOGUE a further investigation can be made of which variables cause deviations in standards or calibrated reference values. To illustrate this, figure 4 displays a radial diagram which indicates that for the issue of eutrophication in the area *Randmeren-Zuid (Gooimeer, Eemmeer and Nijkerkernauw)*, the virtual absence of pike (snoek) and poor turbidity (doorzicht) is a particular bottleneck. The concentrations of nutrients are also too high. Furthermore, the water index graph can also be used to display what the progress of this subject, i.e. eutrophication, has been over the course of time, and what the prognosis is for a specific analysis variant.

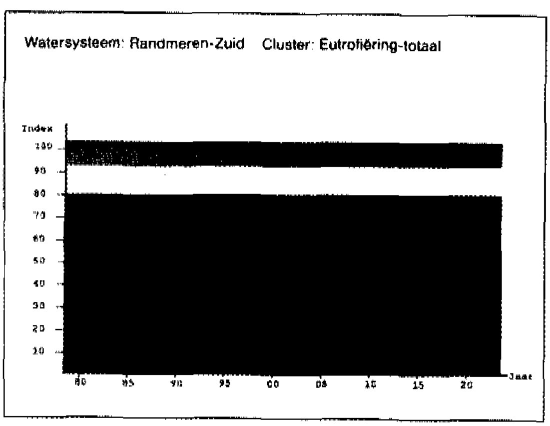

Figure 5: Water index graph for the Randmeren-East area for the problem field eutrophication. The prognoses were derived from the Aquatic Outlook analysis alternatives "current policy".. On the x-axis: year.

CUSTOMISED EXPANSION OF THE WATER DIALOGUE FOR OTHER SURVEYS

The methods that serve the Aquatic Outlook and the WATER DIALOGUE are increasingly seen at other scale levels. River basin approaches and "distance-to-target" assessments are also becoming more applicable in the countries surrounding us (COM, 1996; EEA, 1996). Distance-to-target is directly comparable with the typical ratio. In addition, within the framework of the Regional Water System Reports the condition and use of regional water systems are quantified in a corresponding manner and assessed against policy objectives. The measurement data is collected at a level of basic geographical units, the water systems, and clustered into information per geographic unit or category (Interprovincial Consultations, 1996).
The present WATER DIALOGUE was implemented in order to present data from national and regional waters on a national basis. National borders and choices for the scale level of water systems used in the Aquatic Outlook were also factors that determined the appearance of the WATER DIALOGUE and the Water Mondrian. The WATER DIALOGUE was however set up in such a manner that the lay-out can be adapted to the level of application. A customised expansion, whether entailing international river basins at European level or provincial water systems, has been included amongst the options.

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AUTOMATIC TREND ANALYSIS OF RIVER WATER QUALITY

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ABSTRACT

An important objective of the RIWA (Association of Rhine and Meuse Water Supply Companies) is the detection and estimation of changes in water quality of the Rhine and Meuse. Data on water quality at many points along these rivers are provided by a monitoring network. An objective way to detect and estimate changes in water quality is statistical trend analysis.
Given the large amount of data in the database, the RIWA has expressed the need for a procedure for automnatic trend analysis.

WHAT IS A TREND?

We define a trend as a (semi-)permanent change in the location (mean or median) of a process (such as the concentration or load of a certain compound) over at least some years. It does not comprise changes related to seasonal cycles, or sudden and short lived changes, caused by calamities.

TREND DETECTION AND ESTIMATION

Trend analysis has two different sides, being trend detection and trend estimation. Trend detection gives us an objective answer to the question whether a trend exists. Trend estimation gives us an estimate of the size of the trend, accompanied by a confidence interval.

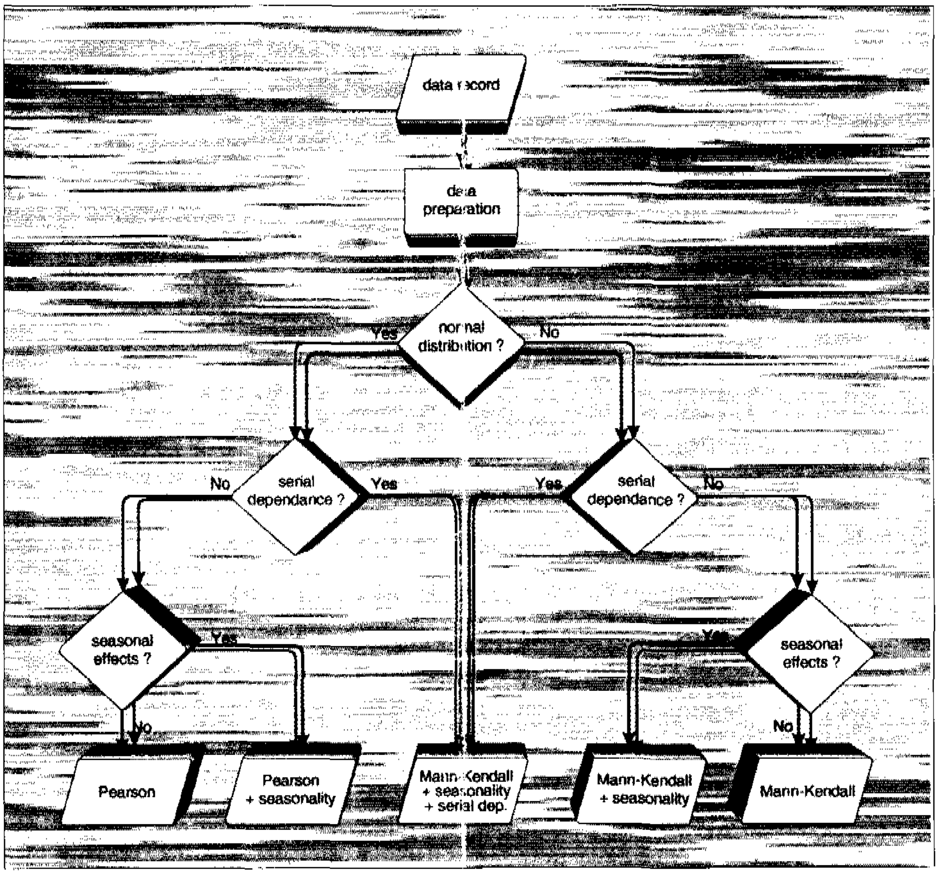
WATER QUALITY DATA ARE OFTEN LESS IDEAL

Data records on water quality are rarely suited for classical statistical analysis, because of underlying non-normal probability distributions, seasonal effects and long-term memory effects (serial dependence). Furthermore, data may be missing or censored (reported as 'less than detection limit'). In recent years many new methods for trend analysis have become available, that can deal with the less ideal characteristics of water quality data. Given the variety of test methods nowadays available to analyze a water quality data record, each with its own capacities and underlying assumptions, considerable judg-ment is required to select the appropriate method.

OUR STUDY

Using Monte-Carlo simulation we studied the application areas and capacities of various tests for trend detection, to establish which test would be optimal for each kind of data record. From this we derived a procedure, enabling automatic trend analysis. It entails specific decision rules leading to the selection of the optimal test for each particular data record. The basic tests in the procedure to detect a monotonic trend are the parametric Pearson test (based on linear

regression) and the non-parametric Mann-Kendall test, as well as extensions to these tests to deal with seasonality and/or serial dependence (see the flow chart).
Concurrent estimates of the magnitude of trend can be obtained from accompanying slope estimators. In case of a monotonic trend this is either the regression slope estimator, Theil's slope estimator, or Kendall's seasonal slope estimator. The decisions in the procedure (the diamond shapes in the flowchart) are based on the outcomes of statistical tests. These are the Shapiro-Wilk test for normality, the runs test for serial dependence and the Kruskal-Wallis test for seasonality. For each particular data record the procedure warrants trend analysis that is statistically sound, with maximum power (the probability to detect an existing trend) and unbiased trend estimates (i.e. without systematic error) that have maximum efficiency (i.e. with lowest error variance).



SOFTWARE

We developed software for automatic trend analysis, with an option to detect and estimate either a monotonic trend or a step trend. The latter is only recommended when the data record contains a long time gap, or when there is a known event which is likely to have resulted in a change in water quality. The decision to use step trend methods should not be based on examination of the data.
The software also has an option to correct for natural influences, like river discharge, water temperature or concentration of suspended matter, before performing trend analysis. This correction will enable a better focus on the antropogeneous influences on river water quality.

RECOMMENDED LEVELS OF SIGNIFICANCE

It is common practice to test on trends with a significance level of 5%, meaning there is only 5% probability that a trend is detected that actually does not exist.
If we are not interested in the direction of the trend, being upwards on downwards, we should perform a two-sided test. We call this exploratory trend analysis. And if we want to evaluate the effects of measures, we are interested in a specific direction of the trend (being downward in most cases), in which case we should perform a one-sided test. We call this confirmatory trend analysis. If we are interested in a downward trend, only the upper limit of the confidence interval is of concern. An estimated downward trend is statistically significant if this upper limit lies below zero.

RECOMMENDED TIME SCALE

Trend analysis of surface water quality should be performed on at least monthly values. Data records on a lower time scale (such as weeks or even days) may suffer from strong serial dependence and a diffuse seasonality, which are hard to deal with in trend analysis.

CONSISTENT DATA PREPARATION

Raw data records rarely permit trend analysis, because of outliers, missing values (or even gaps), changing observation frequencies or censored values.
Therefore, almost any trend analysis should be preceded by a preparation of the data. To warrant consistent results the data preparation process forms an integral part of the procedure for trend analysis.

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