OPENING AND INTRODUCTION

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Mr Chairman, ladies and gentlemen,
Welcome to the third international workshop Monitoring Tailor-Made. The subject of our workshop is information for sustainable water management. Sustainable water management is a matter of balancing. There are many stakeholders that share the same water resources and decision makers have to find a balance between all these interests. To support good decisions, that integrate between all interests, water management policy needs good, integrated information. Decisions have to balance ecological, economic and sociological factors that influence the quality and quantity of the water, and consequently the required information should integrate these disciplines. In this workshop we will be discussing what kind of information water management policy needs to be able to make decisions, and how such information can be obtained.

WATER CRISIS

Good information is becoming ever more important. There is a water crisis today. But the crisis is not about having too little water to satisfy our needs. It is a crisis of managing water so badly that billions of people, and the environment, suffer badly (Cosgrove and Rijssberman 2000). Managing water in a sustainable way requires good information.

The problem as described by the Prince of Orange is that we have too little, too much or too polluted water. At the second World Water Forum, this is extensively discussed, and it was concluded in the Ministerial Declaration of The Hague on Water Security in the 21st Century that an integrated water resources management is needed, accounting for social, economic and environmental factors and integrating surface water, groundwater and the ecosystems through which they flow. We will not discuss the water crisis here, but it underlines the importance of our work; without suitable and solid information about these social, economic and environmental factors and without knowledge of the status of surface waters, their linking to groundwaters and their relation to the surrounding ecosystems, sustainable water management is not possible.

In this workshop we will discuss how we can come to integrated information, how we should deal with transboundary issues, how we can make policy makers and scientists communicate and how we can make scientists from different disciplines communicate and come to an integration of techniques. I will discuss these items one by one and to the end, I will come to covering idea of this all: Tailor-Made Monitoring.

INTEGRATED WATER ASSESSMENT

There is not really a shortage of data. Of course there are regions in the world that can be described as data-poor. But it is not a matter of having too little data. Usually there is data on water quality and quantity, there is data on ecology, economics and sociology, but the issue is that we can not combine these data into useful information. Data are collected for a specific reason and are less suitable for use in an other context. Therefore, we must find an approach to collect data that can be integrated into suitable information for water management. This implies integration of different scientific disciplines in the domains of the natural sciences and socio-economic sciences. Can we link biological data to industrial production data and is it possible to combine data about recreation with chemical data? Such are the questions we are facing.

Integrated water assessment is also concerned with integrating information on groundwater and surface water systems. Both systems have different properties, but they are both part of the same hydrological cycle and influence each other. It was concluded from the workshop Monitoring Tailor-Made II that the same concepts in monitoring strategies are applicable for groundwater and surface water. The differences are mostly related to differences in physical characteristics (Van Luin and Ottens 1997). The same goes for the ecosystems surrounding water bodies. Water management cannot deal with these systems separately. This also requires an integrated approach; on a spatial scale.
TRANSBOUNDARY ISSUES

Then there is the matter that water bodies require an integrated approach crossing borders. Water bodies, being groundwaters or surface waters, do not stop at administrative borders. Management of a water body therefore requires a transboundary approach. Countries sharing a water body must tune their water management. This necessitates co-operation between different countries and harmonisation of different approaches towards the balancing of socio-economic and ecological values, as I discussed earlier. Such a process is full of pitfalls. It requires mutual trust, much communication and, not in the least, patience.

Water management is largely regulated through legal systems. These legal systems put obligations on the collection of information. But such obligations tend to be inflexible and do not account for specific local situations. And across the border, there is another legal system, having other requirements. We should ask ourselves if we can use information to influence developments in policy and legislation, and maybe use it to come to a harmonised system between countries.

POLICY MAKERS AND SCIENTISTS

One of the big problems we are facing in producing information is that information producers, like scientists, often do not speak the same ‘language’ as information users, like policy makers and the public (Helsel 1997). Information users tend to oversimplify and have unrealistic expectations, whereas information producers fail to address the management needs (MacDonald 1994). So one of the issues is to bring these professional groups together and have them approaching water management in an integrated way. The step of specification of information needs is a crucial one in this respect. And at the other end of the information cycle the step of information utilisation is just as essential. These are issues that need more and more fundamental discussion.

TECHNOLOGICAL PROGRESS

This is the age of information. Technology provides us with numerous possibilities to unlock information in a way that was never though possible before. But are we using these possibilities, and can we use technology to support water management? We use databases to store the data, we use sophisticated software to analyse data, we use models to make prediction and analyse water management, and we use GIS to make surveyable maps. But integration of all these techniques is seldom done. How can we integrate such techniques. And can we find examples of integration of techniques that really worked?

MONITORING TAILOR-MADE

This brings us to the workshops Monitoring Tailor-Made. The previous, succesful, workshops were in 1994 and 1996. In these workshops we discussed tailor-made monitoring. But after these two workshops do we know what tailor-made monitoring is? The theory says that tailor-made monitoring is providing the right information to support management decisions. Tailor-made information is effective: the information product is tailored to the questions, and efficient: the information is provided at a reasonable and affordable price (Adriaanse 1997). But that is theory. In practice, tailor-made monitoring is not fully developed. The 1994 workshop Monitoring Tailor-Made I concluded that a better fit should be found between the information produced and the water management needs (Adriaanse et al. 1995). A better fit yes, but how? In the 1996 workshop Monitoring Tailor-Made II this notion was developed into the conclusion that a lack of methodologies to deal with the upper ‘information’ parts of the monitoring cycle is hindering the linking of information to policy. A stronger participation of policy makers and public, and involvement of other than the traditional disciplines were recommended to improve monitoring information (Van Luin and Ottens 1997).

This is where we left four years ago. And now many of you are here again to discuss the progress we made since 1996. And many new people are here, also from disciplines that are new to this workshop, to add your experience. Monitoring Tailor-Made II made special effort to involve groundwater professionals. You seem to have settled, as you are well represented again.
Now Monitoring Tailor-Made III made an effort to involve professionals from the socio-economic sciences I hope this will prove to be lasting as well.

The workshop Monitoring Tailor-Made is intended to provide a stage for the exchange of knowledge and experiences in integrating across scientific disciplines, professional groups, and administrative borders. I hope you all will use this opportunity to discuss and learn. I wish you interesting, fruitful and, not in the least, very enjoyable days here in Nunspeet.

REFERENCES

Integrated assessment is an approach, which seeks to involve all disciplines in policy-relevant assessment. The process aims to encompass environmental science, technology and policy problems. The aim is to establish an overview of the environmental issue in question which attempts to avoid the mistakes of the past associated with narrow, one-sided or uni-dimensional approaches. There are a number of methods available for such assessments. However, they are also subject to a number of limitations, difficulties and dilemmas. Integrated methods are inherently complicated and the tradition is that only experts are involved. New more inclusionary procedures have to be devised in order to involve all stakeholders. They have to be involved in the framing of the issue and in the value judgements associated with the approach. The dilemmas cannot be solved by integrated approaches, but they can be mitigated via proper identification, analysis and evaluation of the gains and loses involved. In structuring the analysis the existence of ignorance has to be accounted for and communicated to the managers and the political decision makers. The ignorance/uncertainty aspects can be partially accommodated for via an intensification of feasible monitoring and research so as to minimise the risks of unpleasant surprises.

Key Words: uncertainty, risk, surprise, inclusionary decision making

INTRODUCTION

The need for integrated approaches is manifested by mistakes and surprises that have dogged the policy formulation and enablement process of the past. The regulatory agencies and the policy makers found to their dismay that the effects of and responses to policy implementation were not always as anticipated. There may have many reasons for this (e.g. ignorance in the first place) but it is beyond doubt that lack of integrated analysis has given rise to many potentially avoidable surprises. Such surprises may occur in cases where it turns out that the instrument chosen for implementation was insufficient, because aspects, other than those considered by the disciplines involved, far outweighed the cause-effect relationship. Similarly, integration was lacking in cases where the effects of the implementation of decisions were as anticipated, but undesirable and often cross-sectoral side effects were encountered as unpleasant surprises.

INTEGRATED ASSESSMENT (IA)

Integrated assessment, while still an emerging methodology, now has a standard terminology and enjoys a reasonable degree of consensus in terms of an understanding of what it is, (EFIEA 1998). It is generally agreed that there are two components i.e. that the assessment is integrated over a range of relevant disciplines; and that it serves to provide information suitable for decision making. It is equally implied that integrated assessment seeks to account for all aspects within the identified reference frame of the issue in question. In the short term, the assessment is based on the existing scientific knowledge. In the longer run, monitoring and research may become an important element in the process of providing adequate information for future assessment and decision making.

IA requires a toolkit that enables analysts to combine knowledge from a number of disciplines, but it also demands the interpretation and communication of this knowledge for a variety of audiences so that the maximum number of stakeholders are given the opportunity to better understand complex phenomena and policy response options. Value-laden assumptions underpinning analysis and policy need to be made as transparent as possible and not shrouded in technicalities. IA practitioners are agreed that incorporating policymakers and other stakeholders into the early design of assessments and testing protocols greatly facilitates this
requirement. Valuation of costs and benefits in this process is more than the assignment of monetary values and includes multi-criteria evaluation methods and techniques in order to identify meaningful and practicable trade-offs (Turner, 2000; Brown, Tompkins and Adger, 2001).

This more holistic assessment strategy will encompass core objectives such as:

- **risk reduction**, achieved through the use of expert-based analysis to better understand complexity, albeit with the caveat that zero risk/complete protection positions are rarely if ever feasible;
- **uncertainty mitigation**, achieved through analysis and precautionary measures to maintain and enhance the resilience of natural and social systems; this management paradigm explicitly recognises the connections between human choice and environmental functioning and tries to ensure that as diverse a set of ecological functions as possible is protected to retain flexibility and reduce vulnerability to surprise;
- **enhancement of social inclusion and fairness**, to manage the ambiguities present in socio-cultural attitudes, values and individual and group psychological effects. Thereby increasing accountability and trust.

Four broad overlapping procedural stages can be distinguished in an IA decision support system:

- scoping and problem auditing stage (utilising the "DPISR" framework (Turner et al. 1998))
- identification and selection of appropriate decision-making calculi-research methods and models
- data collection and monitoring, with the aid of an evolving set of indicators
- evaluation of project, policy or programme options.

![Figure 1: DP-S-I-R Framework: e.g. Continuous Feedback Process Catchment-Coastal Zone Continuum.](source: Adapted from Turner, Lorenzoni et al. (1998))
THE "DPSIR" APPROACH

In The European Environmental Agency the basic problem of the pollution of the environment is fitted to a formal structure called the DPSIR-approach to integrated environmental assessment, (European Environment Agency, 1998). The structure is a formalisation of the interplay (a co-evolutionary process) between the society and the environment with respect to Driving forces, by which Pressures are exerted on the environment. The Status of the environment is the basis for evaluating the Impacts due to the Pressures. That leads to Responses by society to the impacts, which requires identification of objectives and choice of tools by which to curb the pressures. From pressures to impacts is in fact the cause-effect relationship, based on physical, chemical and biological mechanisms describing the relationship between pressures and impacts. This environmental cause-effect relationship involves the natural sciences, while the driving forces' human welfare impacts and the responses involve the political, judicial and the social sciences (see Figure 1).

The role of social sciences is as important as the role of the natural sciences - a fact, frequently ignored, because there has been a general presumption in favour of natural sciences as exact, factual science (frequently an incorrect assumption), compared to the more ambiguous social sciences.

THE PLAYERS IN INTEGRATED ASSESSMENT

The players in integrated assessment are:

- **The scientists.** The fundamental role of the scientist is to know, describe and formulate the mechanisms of the cause-effect relationships, whether natural, technical or social.
- **The managers.** The role of the manager is to help identify and structure the issues in question, co-ordinate the whole process, communicate the approach and results, and implement the decisions made.
- **The policy makers.** The agents to which the assessment is communicated in order to inform decision making. If they are clever, they also demand a significant role in identification and framing of the issue.
- **The stakeholders.** Individuals and groups that have a vested interest in the whole issue.

The application of integrated assessments is most meaningful in relation to complicated environmental issues in society involving many institutions and many interests. Frequently, the roles of the players are confusing and intertwined. The interface between the stakeholders and integrated assessment (and similarly with the disciplines of technical and chemical risk assessment) can be especially controversial. The tradition is that assessments are made in quite an elitist forum of experts with fairly narrow communication lines to the managers and the policy makers. This approach has generated controversy, because it is prone to an actual or perceived lack of accountability; typically being overly influenced by industrial lobbying, and more recently NGO's counter-lobbying. The most important new requirement for integrated assessment as a discipline in its own right, is to institutionalise a much more open, transparent and accountable approach to assessment and decision making in the environmental field.

COMPONENTS OF INTEGRATED ASSESSMENT

Integrated assessment consists of a number of identifiable stages:

- **Identifying, scoping the issue.** This is a crucial component - frequently given too little attention. This is the process by which the issue is made clear for all to see and relate to. A stakeholder mapping exercise is a vital sub-component in this stage. There will be nothing but confusion further down the track if the issue is not identified properly and communication and consultation lines opened up with the relevant stakeholder set.
- **Structuring the issue.** While scoping determines the external boundaries of the issue - as compared to so many other issues, structuring creates the internal logic of the issue and the approach to its analysis.
- **Analysing the relationships.** Choice of tools for the analysis of the issue and the performance of the analysis. The purpose of the analysis is to establish the means by which to describe the inter-relationships between identified pressures and the ultimate impact on the environment, using the knowledge of natural sciences. Similarly, social sciences are used to describe the inter-relationships between the impacts (now defined in terms of gains/losses in
human welfare) and the responses to be anticipated in society, that will influence future performance of the identified system.

- **Data collection and monitoring via suitable indicator sets.** Data requirements are often formidable and the collection task problems are compounded by ‘scaling’ problems, both temporal and spatial. General policy objectives such as sustainable development have served to highlight the need for more comprehensive data sets, but also the lack of appropriate sustainability indicators to aid monitoring.

- **Assessment of results.** To make the results of the analysis fit for conclusions and for communication to decision makers, and ultimately the public, a comparative assessment is made. It may be a simple comparison of the results, but it might also be an evaluation of alternatives or scenarios, results of optimisation, cost-benefit analysis or multi-criteria assessments.

- **Feedback loops.** On the basis of the assessment the decision may be taken or, the whole process may be restarted with an alternative framing of the issue, a new structure and renewed analysis and assessment, before decision makers are satisfied and prepared to take action. After the decision has been taken, it has to be implemented via a range of possible policy instruments e.g. standards and regulations, economic incentive instruments, voluntary initiatives etc. The financing of the whole new management system is also a critically important factor (Bower and Turner, 1998). The performance of the management action, policy or programme is monitored over time and re-evaluated, on the basis of scientific and social feedback, in a cyclical process (see Figure 2).

Adapted from GESAMP Report No. 61, 1996 FAO, Rome.

**Figure 2: Integrated Management Programme Cycle: Catchment-Coastal Planning Example**

This process is illustrated as being non-linear and history shows it is unwise to make it linear. It is best done as a dynamic cyclical process designed to test the basis, renew the analysis, reassess the assessment and evaluate the options for implementation by good communication of alternatives. Many processes have been too narrow conceived in the past and have given rise to surprises, in the sense that the policies chosen were inadequate from the point of view of particular stakeholders and the wider public. Two issues are of particular concern:

- **Identification and framing the issue.** Here the tradition has been to forget to consult with the stakeholders or the public, who may look at the issue in question in a completely different way compared to that of the experts. It is also the case that value judgements can be hidden away behind technical analysis and in this sense the outcome can be steered to suit the purposes of the decision maker commissioning the research – a procedure know as “regulatory or institutional capture” (Bowers, 1988).
There is a need for institutional changes that can guarantee adequate inclusion of stakeholders at the earliest stage of the process, e.g. see New Zealand legislation on environment. Because trade-offs are often required, social processes need to be developed to mitigate or resolve conflict, so that decisions which confer differential entitlements of income, wealth and/or power, can be legitimised collectively.

- **Diversity of Perceptions.** In the IA process there are a number of choices to be decided that can only be made on the basis of value judgements. These judgements are not always transparent to the outside (described diplomatically). There is a tendency in an elitist setting to disregard “irrational” arguments or perceptions disproportionate to scientific numbers. The example is traditional risk assessment, where very clear differences can be found between the laymen’s perception of risk and the probabilities associated with risk. The reason is that risk is equal to probability times consequence (harm). But there is a crucial element of value judgement in the determination of what is an acceptable risk. What right does the expert have to prejudge the stakeholders or the public’s acceptance of risks or potential harm (Langford, Georgiou, Bateman, Day and Turner, 2000)? Perceptions of risk are clouded by a range of ambiguities e.g. certain psychological effects on individuals such as dread; social group pressures which can serve to amplify the fear of harm; and spill over effects in terms of symbolic connections which magnify the harm potential as far as the public are concerned, most recently experienced in Europe with BSE and meat eating in general.

**CAUSE-EFFECT RELATIONSHIPS**

In the development of natural sciences since Galilæi, Kepler and Newton, the deterministic description of cause-effect relationships has been the core of development. It is axiomatically assumed that there is a unique relationship between the action taken and the effects (in the environment). However, during the last century it was realised that there are both inherited and practical uncertainties associated with this relationship and more recently questions have been raised as to whether there is such an identifiable, unique relationship which can be relied upon.

The cause-effect relationship can be written as follows:

$$\mathbf{e} = f(i_1, i_2, \ldots, i_n; p_1, p_2, \ldots, p_n) + \mathbf{\varepsilon}$$

Where:
- $\mathbf{e}$ is the effect
- $f$ is a functional relationship
- $i$ are input variables
- $p$ are parameters
- $\mathbf{\varepsilon}$ is physical, technical and social uncertainty

The functional relationship may be empirical in the form of correlations or theoretical in the form of generic relationships, based on a-priori knowledge of the phenomena involved. In relation to the environmental cause-effect relationships the phenomena are physical, chemical and biological. The input consists of the variables associated with natural phenomena and the anthropogenic pressures on the environment. The parameters characterise the functional relationship. In some cases such parameters are very well known from a-priori scientific knowledge, in many cases they have to be determined in each individual case by experiments and analysis of the phenomena involved. There is an increasing realisation that the uncertainty $\mathbf{\varepsilon}$ may in some cases be the dominant feature of the equation, (Harremoës and Madsen, 1999).

**UNCERTAINTY AND INDETERMINACY**

In a range of cause and effect contexts a degree of uncertainty complicates analysis and assessment in the sense that it has not been possible to describe the effect on the basis of a deterministic functional relationship. This includes the statistical uncertainty in cases where sufficient experimental data if it were available could provide information by which to estimate the statistical error; but it also includes the uncertainty associated with not knowing the essential phenomena and the lack of data for any estimation.

The literature contains a number of different terms which describe components or facets of the aggregate uncertainty problem: risk, uncertainty, ignorance and indeterminacy (Wynne, 1992).
Determinism is an ideal that is never achieved. However, history has demonstrated beyond any
doubt that determinism is worth striving for. However, even in the best case circumstances, the
input variables show statistical properties. But we have statistical instruments which can handle
variable input data; and risk expressed statistically is a rational approach to coping with the
variation. There is always statistical uncertainty involved, due to the mere fact that all
relationships have to be calibrated with data. Given a well known functional relationship and
an adequate combination of number and type of data, the uncertainty can be expressed
statistically and can be incorporated in a formal risk analysis.

Uncertainty beyond statistical uncertainty is experienced when we know the range of outcome,
but not the statistics of it. That is where we can use scenarios as an approach, because we can
describe and assess the consequences of a set of possible outcomes, but we cannot associate
probabilities to these future possible states.

Ignorance applies when we do not know essential functional relationships. The range of
outcome is simply unknown. The relationships may become known later as a result of new
research and development on the issue, but they may not be known at the time when far-
reaching decisions have to be made.

Practical indeterminacy arises in situations where the functional relationships are so
complicated and the number of parameters so large that neither determinism nor stochasticity is
within reach. The functions and the parameters become unidentifiable. Theoretical
indeterminacy is even more problematic because the relationships involved are inherently
unidentifiable, e.g. due to chaotic properties that make predictions impossible.

In the light of all these complexities, it is surprising that a general faith in the deterministic
approach still persists. This is even more surprising, because often the reference anchoring point
is the physical sciences, in spite of the fact that determinism was undermined from an
epistemological point of view with the development of quantum mechanics. In 1927
Heisenberg demonstrated the uncertainty-relationship that defies ultimate determinism.

The positivism of industrialism over the last two centuries was founded on a general belief in
progress, which - in spite of occasional set-backs - moved forward with the indisputable
improvement of the lot of human beings. This improvement can be documented by an array of
indicators related to the well-being of people in the industrialised world. However, in the last
decades of the last century the very foundation of positivism was put under scrutiny. The
industrialised world has responded to this challenge with a remarkable flexibility and
determination to improve performance; but the experience is still that fundamental dilemmas
are unresolved. There is an increasing realisation that the "no-know" component in decision
making is too large to be disregarded.

A HISTORY OF FAILURES/SURPRISES

The number of failures/surprises in the environmental field has been impressive. The European
Environment Agency has initiated an investigation into a number of failures/surprises related to
the environment, where decisions were taken with recognised undesirable results; (EEA, 2000).
The aim is to learn from these experiences and to modify approaches to analysis, assessment
and decision making. This involves an interpretation of the precautionary principle, which in
broad terms recommends precautionary measures to be taken without full scientific proof of
harmful effects (O’Riordan and Jordan, 1995). The Principle is not however, a panacea, it
carries an implicit cost-benefit trade off and often it is difficult to identify meaningful, generic
approaches to implementation in practice (Harremoës, 1998).

THE NO-KNOW SITUATION

The real importance of the precautionary principle is in situations where either ignorance or
indeterminacy prevails. Nevertheless, decision making is still inhibited in situations where no
knowledge of a cause-effect relationship exists at all. For action to be taken at the political level
there ought to be some basis on which to argue a reasonable suspicion, linked to an expected
cause-effect relationship - however uncertain. The policy debate is often a confusion of
arguments which range from those that are rational utilitarian in nature, to those based on
deontological principles or dogmas, or those supported by just superstition or non-structured
empiricism.
The use of the precautionary approach should be conditioned by the characterisation of the risk and policy responses under scrutiny. Thus in situations where contamination is likely to be persistent in the environment, where bio-accumulation possibilities are present and where the severity of the toxicity in terms of consequent harmful impacts of ecosystems and humans is high, a precautionary attitude is sensible. This is because the implications of taking, or not taking, action in these circumstances will typically be irreversible (either on grounds of practicability or because of biogeochemical realities). Irreversibility carries ethical dilemmas along with it, including intergenerational fairness and equity concerns (Page, 1982). The precautionary principle encourages the analyst to adopt an empirical and iterative strategy, with the aim of taking pre-emptive actions in a stepwise fashion. Parallel effort and resources should be invested in on-going research and monitoring to ensure that all feasible ignorance and/or uncertainty reduction opportunities are also taken.

INCLUSIONARY VERSUS TRADITIONAL MODELLING APPROACHES

The dominant approach over recent decades has been to try to rely on the existing knowledge of the cause-effect relationship as the means of predicting the consequences of policy options, while interalia ignoring the uncertainty/ignorance problem. The models have become more and more complex and less and less interpretable from the non-expert perspective. This in combination with a history of surprises/mistakes has increased mistrust on the part of the politicians and the public. The way forward is not to curtail development because of fear of new failures, but to involve all relevant stakeholders in the process of formulating, assessing and choosing policies (Stirling, 1999). During the Stockholm Water Prize laureate year 2000 ceremony, Kader Asmal, Minister of South Africa said "When moving forward we shall make mistakes, but it is wise to do so with open eyes and with a democratic consensus behind us" (SIWI, 2000). Do present elitist science/engineering procedures for integrated and risk assessment live up to such ideals?

INTEGRATED APPROACHES

Activity in one part of the aquatic system may have unexpected and unaccounted for effects in another part of the system, e.g. irrigation up-stream and its impacts on the Aral Sea; or nutrient flux changes across a large drainage basin and ecological impacts in the Baltic Sea (Gren, Turner and Wulff, 2000). Solving one pollution problem may give rise to other problems, e.g. water purification processes produce sludge contaminated with metals or organic chemicals so that the recycling of the sludge to agriculture may not be acceptable. The lesson must be that any economic activity, land use change or measure to abate pollution has to be analysed on a "life cycle basis" in order to account for the full range of imaginable effects on the environment and on humans, (Harremoës, 1997). In so doing, the role of potential ignorance and the surprise effects have to be given due consideration. While the individual tools for integrated analysis are sometimes reductionistic in character, the incorporation into the analysis of the potential role of ignorance must by definition involve a more holistic perspective and integrative toolbox.

Most managed ecosystems are complex and often poorly understood hierarchically organised systems. Coastal and related drainage basin systems and processes, for example, pose a particularly complex challenge because of the spatial scales and the degree of systems complexity and variability that are involved. At the marine interface persistent human intervention focused on gaining "control" i.e. reduce risk and uncertainty related to the coastal environment, has in many ways only resulted in a state of permanent disequilibrium. Such environmental conditions, driven by human reclamation and continued protection of intertidal land (for economic and/or nature conservation reasons) is arguably more risky to humans and not less so. On the terrestrial side, floodplains of rivers and estuaries are also of major socio-economic and ecological importance. Much of the historical utilisation of these resources has been piecemeal and undertaken without due regard for the underpinning natural processes and functioning, or for the long run consequences. In many areas, land use changes and their consequent increased rates of surface run-off, combined with a reduced accommodation space for water, have increased flooding.

Capturing and managing the range of relevant impacts on natural and human systems in the coevolving catchment-coastal continuum will be a formidable scientific and administrative task.
(Salamons et al. 1999). There is an urgent need to develop better methodologies for the improved understanding and detection of ecosystem change, as well as the evaluation of different ecological functions. Modelling work, monitoring and indicator work and scientific/administrative experimentation all need to be better integrated. Difficulties constraining the formulation and implementation of an acceptable IA are manifold: e.g. the diversity in phenotypes of coastal ecosystems and thus their functional value (Turner, Bateman and Adger, 2001); regional and national differences in socio-economic developmental stages; the pace of development and cultural constraints on social and environmental perceptions and attitudes. Another major analytical barrier to overcome is the different time and space scales on which these subsystems react and interact and thus the different required scales of prediction. But not every flood plain or coastal problem will require a fully integrated approach. Generic strategies focussed around a core of sound interdisciplinary science can be the basis for a future more flexible and inclusionary coastal management strategy.

**MONITORING AND RESEARCH**

Monitoring makes sense only in context. Monitoring of the status of the environment has become an important issue due to the two conflicting concerns: need and cost. Each monitoring operation has to have an identified purpose. In the European Environment Agency the following terms have become common terminology: BAI = Best Available Information and BNI = Best Needed Information. The two categories of information are not identical and data collected without referral to either concept can be called: WMI = Wasteful Monitoring Information. What is needed from a management point of view is an identification of what is interesting to the public and relevant to the political process. To define what is needed from a scientific point of view is even more difficult. In the past, water quality was looked at uni-dimensionally but this is no longer acceptable. Today, water availability and water quality as affected by use and abuse has to be analysed in an integrated fashion.

The key challenge is to identify monitoring needs given potential uncertainty and ignorance. That is no easy task due to the risk of generating overly comprehensive data sets a proportion of which falls into the WMI-category. The risk of making the wrong decision is not different from the risks involved in the actual decisions. However, the consequences of being wrong are different, because WMI is often far cheaper than failure in the decision itself. There is only one conclusion: move in the direction of intensified monitoring and research in cases of reasonable suspicion and doubt about irreversible pressures on the state of the environment and about the cause-effect relationships that connect pressures and potentially serious impacts (directly or indirectly damaging human welfare).

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**Figure 3: Research and Policy Agenda**

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Given the generic policy goal of sustainable development, management agencies seeking to sustainably utilise ecosystems should be giving a high priority to research which provides knowledge about system resilience and its maintenance. Such research would buttress a more integrated management process with a priority focus on systems functioning and linked outputs of economic and socio-cultural goods and services. Retaining as much systems functional diversity as is practicable is a key objective. Another dimension of sustainability is the equity/fairness principle both intragenerationally and intergenerationally. In this context a more "civic science" needs to be fostered, a paradigm in which scientists actively participate in the communication and use of science in the political process; and more inclusionary mechanisms to engage all relevant stakeholders and to place power and responsibility for planning and decision-making at the lowest feasible level of governance i.e. the subsidiarity principle (see Figure 3).

BASIC PRINCIPLES OF POLICY

The EU has introduced a number of principles as the basis for environmental policy in Europe (e.g. Principles such as Polluters Pays Full Cost Recovery, Pollution Prevention and waste minimisation, Best Available Technology Not Entailing Excessive Cost, Precautionary Approach, Subsidiarity, Proportionality etc.). Some are adopted from earlier policy traditions, but it is the combination that creates a somewhat confused overall picture. The political economy of regulation and policy instruments has become much more complex and contradictory in the context of these multiple principles which often conflict with each other (Harremoës, 1996). In the future it will be necessary to examine the deployment of instruments across different economic sectors and size of enterprises and governance; as well as the issues surrounding the transfer of instruments across countries with very different institutional contexts (European harmonisation and ‘greening government’ issues are relevant here). But IA can facilitate a process in which contradictory principles are highlighted and therefore there may be fewer policy dilemmas and surprises in the future than has been the case in the past.

CONCLUSION

Scientists/engineers have to acknowledge that the contribution they make to identification, framing, analysis and assessment of environmental issues is inevitably conditioned by significant uncertainty/ignorance. It is not an expression of incompetence, but the opposite, to associate any statement on cause-effect relationships with an accompanying statement on the uncertainties and indeterminacy associated with that statement. Likewise, managers and policy makers have to accept statements of uncertainty from experts without interpreting it to be the result of ineptitude on the part of the expert. The politician/manager should not expect the technical analysis to provide definitive answers sufficient to cover the back of the policy maker in any policy controversy. The politician can only cover his back by proper consultation with all stakeholders and the political spectrum of interests. New approaches can only be implemented by institutional changes that create systems designed explicitly to handle environmental issues in a fair, transparent, equitable and accountable fashion. There is still a long way to go before such governance goals can be achieved. The starting point is reformulation of the laws on environment, demanding new procedures for the greater involvement of stakeholders and the public from the very beginning of issue formulation.

New paradigms are on their way with respect to regulatory approaches to pollution of the environment, evaluation of risks and more social inclusion. There have been too many surprises in the past. Monitoring will play an increasing role in the attempt to identify potential surprises and learn lessons at an earlier stage of development than experienced in the past. There is an emerging interdisciplinary (IA) methodology, which needs to be built upon, that has begun to connect human choice and environmental functioning. The component theory, methods and techniques have begun to be deployed in areas such as environmental change management and environmental risk (including food safety) management. The contributing disciplines need to be continually in contact in a supportive and innovative research context.
REFERENCES


MONITORING, ENFORCEMENT, AND THE CHOICE OF ENVIRONMENTAL POLICY INSTRUMENTS

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How to choose among the dozen policy instruments available to environmental management agencies has been a matter of concern and debate among environmental economists for the entire life of the profession—nearly four decades. The ability, or lack of it, to measure the quantities or observe the actions made "enforceable" by particular policy instruments ought clearly to be central to this choice. But all too often the monitoring problem has been assumed away. When it is reintroduced in realistic forms, we find, not surprisingly, that some favorite policy instruments, such as pollution charges, are not applicable to some important problems, such as runoff pollution from farms; that marginal subsidies, by changing the burden of proof, may no longer be symmetric with charges; and that the apparent freedom from monitoring requirements of the newly fashionable instrument involving the public provision of information about firms or products is "paid for" by our inability to say anything about its performance on other dimensions that are also of interest.

Key Words: policy instruments, pollution charges, marginal subsidies, deposit-refund, informational regulation, enforcement.

INTRODUCTION

Environmental economists have been fascinated, since the effective beginnings of the sub-discipline, in the 1960s, by the problem we call in our jargon, "the choice of policy instruments." A "policy instrument" here may be a limitation on the quantity of a pollutant allowed to be discharged in a time period (often embodied in a permit or license); it may be a charge levied on each unit of pollutant discharged; or any of about a dozen alternatives listed in table 1 below. The origins of this fascination seem to lie in the seminal work of Allen Kneese (e.g., 1964), in which he praised the Germany system of pollution charges that was part of the Ruhr basin effort to manage water quality in the face of highly concentrated "dirty" industry and attendant population. Indeed, economists might be said to have been in the business of constructing arguments for such charges as opposed to other possibilities. The bias arises because, at least under certain assumptions, charges can be shown to lead to the "efficient" (socially cheapest) solution to a regional environmental quality management challenge.

For the purposes of this conference it is at least as interesting to note that a very large part of the instrument-choice literature effectively ignores another enormously important kind of "instrument"—the monitoring instrument. That is, monitoring has usually been assumed to be not just possible but costless and "perfect," and to be coupled with penalties for discovered violations that are sufficient to induce polluters to obey whatever rule, or pay a correct level of whatever charge, they face. That this is utterly unrealistic will be clear to those who make their living doing or studying monitoring. It is, by now, even clear to some economists. But, in the meantime (which is to say 35 years) a good deal of attention has been lavished on comparisons of charges and permits in fairly straightforward monitoring situations—such as pollution discharges from large point sources—while surprisingly little has been devoted to adapting more exotic policy instruments—such as ex post liability or deposit-refund systems—to the situations that present special monitoring challenges. This paper stresses the adaptation of policy instruments rather than the creation of new monitoring instruments because it is about economics, not physics, chemistry or engineering. The two efforts, adaptation and creation, will tend to go on simultaneously and to change the nature of the challenges even as we discuss them.

One final preliminary note: Essentially this entire paper is written using the pollution control problem as the general setting of interest, but analogous distinctions and arguments may be made for such fields as fisheries, forestry, mining, road building and, in general, any human activity that creates stresses for (degrades or injures) the natural environment.
DISTINCTIONS AND DEFINITIONS

The interest here is in what is called "continuing compliance monitoring," not in either ambient-quality or initial-compliance monitoring. The former will be familiar to readers of these proceedings, but "initial compliance monitoring" may not be familiar to Europeans. This exercise arises under U.S. laws and regulations when a new or modified pollution source is seeking permission to begin operation. It is generally required that the source satisfy the agency that its equipment is capable of meeting its permit terms. The incentives on both sides, source and agency, are to make this process effective. Surprise and randomness, two essential features of effective monitoring (auditing) for continuing compliance are not required here—indeed, might be said to be out of place.

Continuing compliance monitoring is done as part of a larger enforcement system in which some activities or levels of activities are defined as not permissible. In the simplest setting, this could mean having a banned substance in your possession. In a system based on permits to discharge up to certain quantities of a given pollutant per unit time, a violation exists when the permit terms are exceeded. Penalties usually follow when the source is discovered in violation. This is the enforcement part of monitoring and enforcement. Of course, because monitoring equipment is imperfect, even a source that was in perfect control and trying to comply might be found in violation. And vice versa for a violator. Add to this the possibility of imperfect source control, and the matter of identifying "true" violations becomes a serious statistical problem and, more important perhaps, the source of political difficulties for the responsible agency.

The monitoring problem, and thus the overall enforcement challenge, is defined by the policy instrument chosen. For example, if a substance, process, or product is banned, the essence of monitoring is to look for evidence of defiance of the ban. If an affirmative requirement to install a particular type of pollution reduction system is the policy choice, then checking for the required equipment and connections is necessary. If a limit is set on discharge of a pollutant per unit time, then amounts per unit must be measured. And so on. The essence of the link between monitoring and policy instrument choice, then, is the question: Does our ability to monitor match the requirements or restrictions of the chosen policy instrument? If the answer is no, at least not currently, then choosing a different instrument makes sense.

Table 1 Instruments of environmental pollution control policy

<table>
<thead>
<tr>
<th>Instrument Description</th>
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<tbody>
<tr>
<td>1. Prohibition (of inputs, processes or products)</td>
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<td>2. Technology specification (for production, recycling or waste treatment)</td>
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<td>3. Technological basis for discharge standard&lt;sup&gt;a&lt;/sup&gt;</td>
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<tr>
<td>4. Performance specification (discharge permits)&lt;sup&gt;b&lt;/sup&gt;</td>
</tr>
<tr>
<td>5. Tradable performance specification (tradable permits)</td>
</tr>
<tr>
<td>6. Pollution charges</td>
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<td>7. Subsidies</td>
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<td>8. Liability law provisions</td>
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<td>9. Provision of information</td>
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<td>10. Challenge regulation and voluntary agreements</td>
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</tbody>
</table>

Source: Russell and Powell, 1999

Notes to Table:

<sup>a</sup>In a technology-based standard setting, the amount of allowed pollution is determined via an engineering study in which a legally designated technology is applied on paper to a particular polluting operation with known uncontrolled pollution load (raw load). The result of this exercise is an achievable discharge amount.

<sup>b</sup>Performance specification can be based on any of a number of rules or methods, from uniform percentage reduction by all sources to modeling that determines the cheapest way to attain a given ambient quality standard.

<sup>c</sup>The deposit-refund system, for example for drinks containers, is a self-financed marginal subsidy for container return.
Three distinctions among regulatory challenges are relevant to our monitoring ability and thus form an important basis for the choice among policy instruments:

- Some events are, for practical purposes, "fugitive." Others might be called "labeled" or "persistent." Thus, SO2 discharged up a stack can be measured in the stack or just outside it. Once advection and diffusion have begun their work, the chance for measurement is gone. (In some situations it may be possible to work backwards from observed ambient concentrations, through an atmospheric model, to estimate discharges for a set of sources (e.g., Courtney et al 1981). In others, working forward from measured inputs, via materials balance in a well-understood process, may imply pollution discharges. Both are probably special cases.) But an oil spill into a water body persists for some time, the quantity spilled can be estimated and, often, the source identified for hours (at least) after the event.

- Some events are, in the current state of monitoring technology, "unobservable." Others are "observable." Point source pollution discharges are examples of the latter, at least for pollutants that are themselves measurable. Non-point source discharges to streams of nutrients, pesticides, and soil, from a particular farm are, in the normal, complex situation in which farms and streams do not come in perfectly matched and exclusive pairings, not observable. (Again, some special situations exist that change this. Most prominently, where fields are drained by installed systems, as they often are where irrigation is the only source of water, the non-point source is made (or can be made) into one or more point sources.)

- Some actions or events are, at a cost, concealable, others are not, for practical purposes. The prototypical concealable event is the midnight dumping in the woods, of a drum of some chemical, such as spent dry-cleaning solvent. The gases generated in a utility fossil fired boiler, on the other hand, must be discharged up the stack as they are generated and are difficult to conceal. The cost of providing and using storage, so that the gases could be stored and discharged when it was clear no monitoring was taking place are too high to make this realistic.

**IMPLICATIONS FOR INSTRUMENT CHOICE**

These distinctions help us think about the implications of monitoring capabilities and limitations for the choice of policy instruments. Some of these may seem blindingly obvious. Others will, I hope, strike you as worth thinking about.

- If something is unobservable, there is no point in choosing a policy instrument that requires its observation to be effective. Thus, it makes no sense to think about issuing pollution discharge permits to farms or charging them so many euros per kilo of nutrients that end up in streams. This is why countries try to affect farm pollution via such more or less imperfect instruments as fertilizer and pesticide taxes, specified ways to till, rules about how much manure to spread and when to spread it, and so forth (Abler and Shortle, 1991). There is even a suggestion in the literature to use a liability-like approach based on ambient water quality, which is observable (Segerson, 1988).

- Self-monitoring (called "self-reporting" in the environmental economics literature) can be useful, and does shift the financial burden, but if the events monitored are fugitive, auditing of the self-reports may be essentially impossible. Measuring discharges in week t + 1 does not tell us anything very definite about discharges in week t, or t - 1, or t + 2, for that matter.

- Actions or events that are either very difficult to observe or are concealable with only modest difficulty by someone determined to flout the law—or that combine these features—are excellent candidates for policy instruments that change the burden of proof from enforcer to "potential perpetrator." The most familiar such instrument is the deposit-refund (D-R) system, often used for glass, plastic, and aluminum drink containers. The concealable action that some of these systems are instituted to avoid is littering--tossing the container out of a car window. But, more broadly, society may decide it does not want such containers—or any of a number of other objects--mixed in with trash or dumped down sewers. Thus, there are, even in the U.S., D-R systems for auto batteries; and European countries have them on auto lubricating oil and the auto itself. They could be used much more widely for small-volume toxics such as dry cleaning fluid that are not consumed in use but require return to be recycled. Or for old TV sets and computers. (See Wall Street Journal, 14 July 2000.)
One beauty of the D-R approach is that the D finances the R so that general revenue needs are not affected. Another is that its incentive to proper return/disposal is "decentralized." Anyone who finds and retrieves an improperly disposed of item can collect the refund. But notice that, in many possible applications, monitoring will still be necessary. For example, if a liquid, such as motor oil or spent solvent is the product subject to the system, it will be necessary to check that the volume returned for the refund has not been "cut" by something cheaper. (The oil might be floating on water, for example.)

Another way of changing the burden of proof without going to a payment for desirable behavior is to adopt a "presumptive charge" on the undesirable (or at least costly) behavior (e.g., Eskelund 1996). Then, if the firm, farm or person subject to the charge wants to have it changed, she, he, or it must prove that a lower charge applies by supplying convincing monitoring data to that effect. This might be tried even for quite ordinary situations, such as conventional point-source pollution, where the monitoring resources of the responsible agency are scarce. Developing countries, for example, often seem to have trouble gathering themselves to do serious continuing compliance monitoring.

More generally, in situations in which compliance monitoring and compliance itself are not at all well established "habits," it has been suggested by me and others that the choice of policy instruments should be dominated by the concern to foster a "culture of compliance" (Bell, 1997; Russell and Powell, 1999). This could be approached via easy-to-monitor requirements, such as to install particular technologies. Economists know that such instruments cannot, except in the rarest circumstances, lead to "efficient" outcomes in the short run; may postpone the date of adoption of better technology; and, most important, say nothing definite about the skill or consistency of operation and thus of environmental results. Yet the scarcity of currently available "institutional resources" may make such a choice desirable. This, I might add, in the interests of full disclosure, is a suggestion that scandalizes many of my colleagues, who are in the business of pushing "economic instruments" such as charges on pollution, on developing countries (e.g., World Bank, 2000).

There is currently much excitement in policy circles, and even in environmental economics, about the possibility of using the public provision of information as a policy instrument. Eco-labeling and the U.S. experiment called the Toxics Release Inventory (TRI) are both examples. The former is aimed at consumers; the latter at investors. Enthusiasm stems in good part from the seemingly non-confrontational and non-directive nature of the techniques. (E.g., For the TRI no one is required to do anything, except report their toxics releases.) There have even been attempts to adopt the magic (to economists) mantle of pollution charges by referring to information provision as a "market-based instrument," the label sometimes used for charges and marketable permits. After all, if information does have an effect, it happens through reactions in markets. But it is important that we be clear about the monitoring link. For example, the TRI currently is based entirely on self-reports with no auditing. These, in turn, seem likely to be based on calculation from materials balances and process specifications rather than actual measurement. This should give us some pause about apparently observed effects of the instrument's use. (E.g., Konar and Cohen, 1997, find a statistically significant effect of TRI publication on firms' stock prices. This, in turn, seems to lead to reductions in toxics releases in later years.) More broadly, there is never going to be a policy instrument for which monitoring is not a concern.

Finally, it is ironic but true that the great instrument-choice debate for economists—that between fixed limits on discharges (permits) and charges or taxes on those discharges—is not illuminated by monitoring considerations. Under either policy choice, discharges per unit time must be established in a believable way (subject to errors of inference, of course). The economics literature began by trying to make a distinction where none exists and then went in for clever but artificial symmetry-breaking tricks, such as assuming charges would be based on self-monitoring, while fixed permits would be enforced on the basis of agency monitoring (e.g., Harford, 1978). This gave rise to some interesting theory but does not introduce a real distinction between the instruments.

**CONCLUDING COMMENTS**

As a general rule, monitoring capability functions as a necessary condition in the instrument choice decision. If monitoring the quantities or actions that are constrained, required, charged, or rewarded by a policy instrument is impossible, or even very difficult or expensive, it is
almost certainly better not to use that instrument. This relationship is very likely to vary across nations because of differences in institutional "capability" generally, so that the appropriate instrument for any particular problem, such as point source discharge of some conventional pollutant, for example, might best be dealt with by a different instrument in Uganda than in Portugal.

In general there will by this test be several possible instruments for any given challenge. Choosing among them will involve such considerations as total cost of achieving a socially chosen goal, where "total" includes the cost of monitoring when the monitoring system is designed to be least-cost for the desired results. (E.g., Malik, 1992, explores this in a special but interesting situation.) Other considerations will include implications for government revenues, relative incentives for seeking and adopting new control technologies, and the flexibility of the instrument in the face of inevitable change in the underlying situation (e.g., the birth and death of products and firms or the experience of inflation).

Finally, as clever people invent new policy instruments and new monitoring techniques, the choice landscape changes. The latter seems inevitably to be in the direction of enhancing the regulator's information gathering ability, which will tend to broaden the choices available for pursing policy goals. Thus, for example, when a technique is perfected for estimating farm runoff using satellite imagery, it will be possible to treat farms like factories for pollution control permitting purposes. (See Florens and Foucher, 1999, for a discussion of the use of satellites in oil spill detection.)

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Found at dsharman@WorldBank.org.
PERSPECTIVES AND LIMITATIONS OF INDICATORS IN WATER MANAGEMENT

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This paper addresses the issue of indicators for water management by considering why we need them, what the characteristics of a good indicator should be, how they can add value to policy and decision-making, and what are their limitations. There is a vast quantity of data available on water in Europe from which a range of basic trend indicators has been derived. Most of these indicators address "what is happening?" type questions. Some indicators go further and address "does it matter?" questions by assessing trends against prescribed standards and targets. There have been some interesting developments in indicators which help to understand the demand side of water management and to assess the relative efficiencies of resource usage in different socio-economic sectors. More effort is needed to bring together relevant social, economic and environmental information interactively to define indicators which address questions about the sustainability of water use and the contribution of the water environment to our overall quality of life. The development of indicators to date has concentrated on making the best use of available information. But we now need to move from a position of "best available information" towards "best needed information". This will require better communication between information users and providers. It will also require a re-thinking of exactly what are our priority knowledge needs to support policy-making and environmental management. This is essential if we are to obtain the best value from limited monitoring resources by eliminating the current redundancy in reporting requirements and re-focusing programmes to deliver priority information needs.

INTRODUCTION

The quest for producing indicator sets currently taking place in different countries and international organisations has already resulted in a huge variety of different graphical presentations and texts which attempt to answer different kinds of questions. Many of these relate to variations of the Pressure-State-Response (P-S-R) framework proposed many years ago by the Organisation for Economic Co-operation and Development (OECD). Some organisations have built on this basic framework to provide more definition to the different parts of the cycle, such as the Driving Forces, Pressures, State, Impact and Response (DPSIR) model being used by the European Environment Agency (EEA) as the basis for its environmental analysis and reporting (EEA, 2000). Others have used different approaches linked to sustainable development themes which bring together social, economic and environmental objectives. The UK Government, for example, has used this approach in developing indicator sets which will be used to chart progress with the UK's sustainable development strategy, "A Better Quality of Life" (DETR, 1999).

This paper looks specifically at indicators for the water environment, but does so in the wider context of the different conceptual frameworks being developed elsewhere. The paper concentrates on what priority questions we are attempting to answer through the use of indicators and how effectively we are currently doing this in Europe. The paper draws upon examples taken from a recent exercise carried out by the Environment Agency to develop environmental indicators for England and Wales.

WHY DO WE NEED INDICATORS?

The concept of indicators is not a new one. Indicators are just a way of packaging information in a simple and straightforward way that helps to deliver clear and meaningful messages to target audiences. The development of indicators started to become popular as a tool for assessing progress with sustainable development objectives and have been used in this context at international, national, regional and local levels. Water resources are fundamental to social and economic well-being, but relatively few of the indicators in common usage relate to the water environment. Because of the inherent complexity and variability in water quantity and
water quality it is essential that scientific information is presented in a way which has meaning to the policy makers who are making decisions about management and to all those, including the general public, who have to live with the consequences of those decisions. Indicators have a very important role in this context.

**WHAT ARE THE CHARACTERISTICS OF A GOOD INDICATOR?**

Although the principal aim of indicators is to simplify complex information and crystallise key messages, they must satisfy certain scientific and technical criteria if they are to have credibility.

An indicator should therefore:
- be representative;
- be scientifically valid;
- be simple and easy to interpret;
- show trends over time;
- give early warning about irreversible trends where possible;
- be sensitive to the changes it is meant to indicate;
- be based on readily available data or be available at reasonable cost;
- be based on data adequately documented and of known quality; and
- be capable of being updated at regular intervals.

Some of the indicators in common usage for water fall short against some of these criteria. One of the main constraints is the availability of relevant data in relation to the priority questions being asked. The indicators proposed to date have therefore tended to concentrate on making best use of available data rather than starting with the questions first.

**WHAT ARE THE PRIORITY QUESTIONS?**

Water management relies on all kinds of information to support decisions on policy and legislation, river basin management planning and investments, and a plethora of local operational needs. It is not possible, or sensible, to attempt to capture all of these information needs in sets of indicators. This paper therefore focuses on a limited number of priority questions related to whether policies for water management are working and what more needs to be done. Some priority questions are:

(I) To what extent are existing policies and measures serving to reduce critical pressures on the water environment?

(II) To what extent is the general state of the water environment improving as a consequence?

(III) Are human and ecological health standards being met?

(IV) What is the result of actions to address specific impacts on aquatic ecosystems?

(V) What is happening to the water environment in response to global climate change?

(VI) How are different socio-economic sectors contributing to the sustainable use of water?

(VII) How far are these changes contributing to our overall quality of life?

Table 1 shows some examples of indicators that have been proposed for the water environment which help us address these questions.

**HOW WELL ARE WE MEETING OUR INFORMATION NEEDS?**

Looking broadly across the information base for water in Europe, there is a long history of collecting certain kinds of data related to different water uses. This has resulted in the accumulation of large quantities of data on discharges to and abstractions from the water
environment, much of which is archived and not used. We are able to assess trends in the levels of pollutants released through point-source discharges from industrial and domestic wastewater treatment plants and also abstractions for water supply for different uses. Figure 1, for example, shows the quantities of certain contaminants entering the sea from land-based sources in England and Wales. This indicator shows the result of changing practices in industry, improved regulation, investment in pollution abatement, and pollution prevention initiatives. It also shows the impact of natural forces where reduced flows following the 1995 droughts led to reduced contaminant fluxes. The overall loadings increased again when river flows returned to normal in 1998. This kind of information is useful in so far as it allows us to address general “what is happening” questions. But it is quite difficult to discern the specific outcomes of individual pieces of policy and legislation. This is becoming increasingly important as we need to gauge the success of major items of legislation such as the Urban Waste Water Treatment (UWWT), Nitrates, and Integrated Pollution Prevention and Control (IPPC) Directives which are driving significant expenditure across Europe. This will require better targeting of monitoring and assessment programmes in relation to this legislation.

Table 1. Priority questions and examples of related indicators for the water environment

<table>
<thead>
<tr>
<th>1. To what extent are existing policies and measures serving to reduce critical pressures on the water environment?</th>
<th>Examples:</th>
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</thead>
<tbody>
<tr>
<td>• Loadings of organic pollution from wastewater treatment plants.</td>
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<tr>
<td>• Inputs of contaminants from land to sea.</td>
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<tr>
<td>• Water abstracted relative to total renewable resources.</td>
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<th>2. To what extent is the general state of the environment improving as a consequence?</th>
<th>Examples:</th>
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<tbody>
<tr>
<td>• General chemical and biological quality of rivers.</td>
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<tr>
<td>• Status of salmonid fisheries</td>
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<td>• Litter on beaches.</td>
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<th>3. Are basic human and ecological health standards being met?</th>
<th>Examples:</th>
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<td>• Bathing water quality.</td>
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<tr>
<td>• Drinking water quality.</td>
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<tr>
<td>• Dangerous substances in surface and ground-water.</td>
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<th>4. What is the result of actions to address specific impacts on aquatic ecosystems?</th>
<th>Examples:</th>
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<tr>
<td>• Nutrient (N&amp;P) concentrations in rivers, lakes and coastal water.</td>
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<tr>
<td>• Exceedence of critical loads of acidity to freshwaters.</td>
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<tr>
<td>• Extent of river habitat modification.</td>
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<th>5. What is happening to the water environment in response to climate change?</th>
<th>Examples:</th>
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<tr>
<td>• Sea level rise.</td>
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<tr>
<td>• Numbers and severity of flooding incidents.</td>
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<tr>
<td>• Frequency and intensity of storms.</td>
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<th>6. How are different socio-economic sectors contributing to the sustainable use of water?</th>
<th>Examples:</th>
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<tbody>
<tr>
<td>• Emissions and water consumption from industry relative to economic output.</td>
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<tr>
<td>• Water consumption per household.</td>
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<tr>
<td>• Water use for irrigation per unit of agricultural production.</td>
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<th>7. How far are these changes contributing to our overall quality of life?</th>
<th>Examples:</th>
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<tr>
<td>• Water affordability in relation to water quality improvements.</td>
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<tr>
<td>• Expenditure on water infrastructure as proportion of GDP.</td>
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</table>
There is also a vast amount of data available on certain aspects of the state of surface and groundwaters. Much of this derives from monitoring networks set up to classify waters according to their general chemical, biological, and hydrological status. Figure 2, for example, presents an indicator showing recent trends in the chemical and biological quality of rivers in England and Wales in relation to a general classification scheme. This reflects a general pattern of improving water quality taking place in many European countries as a result of increasing investment in sewerage and wastewater treatment, and better regulation. The indicator conveys a very clear message of overall improvement to which policy-makers and the general public can relate. For this reason it is one of the limited set of "headline" indicators chosen by the UK government. However, because it reflects the sum total of effort and expenditure, again, it only provides us with a feel for what is happening and it is not possible to assess the relative results of different measures from this kind of information.

Much of the data collected through these general quality assessment programmes only provide proxy information on the real state of the environment. Ultimately, we are really interested in the protection, restoration and improvement of the ecological quality of water ecosystems. This overall objective has now been formalised in the EC Water Framework Directive which requires that good ecological quality is achieved for all waters unless there are particular circumstances for which derogations can be made. Very few indicators have been developed which allow us to measure progress in achieving ecological outcomes, and those that are available are often difficult to interpret. Figure 3, for example, shows trends in populations of Atlantic salmon in England and Wales in terms of both rod and net catches. The indicator shows a clear declining trend in salmon numbers for which we do not yet have adequate scientific explanation. It is thought that this could be linked to changing conditions at sea which are influencing the numbers of migratory fish returning to their rivers of origin. This declining trend masks another very important positive observation which is that because of recent river quality improvements, salmon populations are becoming re-established in many rivers where they have been absent since the Industrial Revolution. The return of salmon (and other species for which similar trends have been seen, such as the otter) is a very resonant indicator of success with river users and the general public but needs to be properly understood.
One of the main concerns of the public is that the water environment is safe and that the different uses to which it is put do not present a risk to their health. Health-related standards are applied in some aspects of water management such as those for drinking water and bathing water quality.

Figure 4, for example, shows trends in compliance with the standards of the EC Bathing Water Directive at beaches in England and Wales. This indicator is very important from a policy perspective as it is crucial to know the outcome of investment in coastal sewerage schemes and progress in delivering the necessary standards. The public, however, are more interested in finding out about the quality of bathing waters while on their holidays and not the results of overall compliance judged at the end of the bathing season. The indicator therefore needs to be underpinned by up-to-date information made available locally for each site. The World Wide Web has proved to be an invaluable tool for disseminating this kind of information quickly to a wide audience using mapping interfaces and postcode access.
Sustainable water management requires that we take a long-term view. This is particularly important in the context of the changes that are predicted in global climate and their potential consequences for the water environment. Much of the research effort to date has been focused on the prediction of possible future changes, but relatively little attention has been given to the monitoring and measurement of actual environmental change. This will require more systematic long-term monitoring to answer the priority questions related to the future management of water resources. Indicators have a very important role in presenting this information to support future policy development and operational planning. The indicator presented in figure 5 shows the increasing sea level around the UK coast over recent decades derived from valuable long-term monitoring programmes. This increase is predicted to accelerate with increasing thermal expansion of the oceans. It has very significant implications for policy on future flood defence planning.

Until recently, very little attention has been given to indicators which help us gain an insight into the socio-economic driving forces which are giving rise to the changes observed in the environment. This has been a major gap. Environment policy cannot be divorced from economic and social considerations. We need a better understanding of how changes in different sectors are resulting in environmental change and the relative eco-efficiencies in resource management within these sectors. Figure 6 presents some simple comparisons of water usage in different sectors with other measures of socio-economic progress in the UK. Although this kind of analysis is quite rudimentary it does begin to give an idea of performance with respect to efficiency of resource usage. The industrial sector has made some significant progress in decoupling water consumption from economic output. This contrasts with performance in the domestic and agricultural sectors which have not achieved similar improvements in efficiency of resource usage. This is an area of indicator development that requires more work to bring together the relevant environmental and socio-economic data sets to see how they can be used more interactively.

Finally, what really matters for most people is how these changes contribute to their overall quality of life. This concept is quite difficult to capture in individual indicators and it is therefore not surprising that hardly any of the indicators proposed to date embrace this quality of life dimension. Work is currently being carried out in the research community to try to derive indices of "green GDP" which take account of social and economic well-being as well as environmental impacts. There is no doubt that the water environment makes an important contribution to social well-being and economic prosperity. Improvements to the water environment have certainly been achieved, but at significant cost, which ultimately has to be met by water users and the general public. The affordability of water charges is therefore an important consideration in addressing quality of life issues. Figure 7 shows an indicator which gives a measure of water affordability expressed as the percentage of households spending more than 3 per cent of their income on water charges. It indicates that despite the significant investments made by the water industry, there has been a slight fall in the proportion of income
spent by households on water charges. This is over a period where the public have enjoyed significant improvements in the quality of rivers and coastal waters which must, in different ways, contribute to our general quality of life. This is an important area for the development of new indicators which should help move the environmental debate into mainstream social and economic policy thinking.

**BRIDGING THE GAP**

The value of indicators is dependent upon the ability of our monitoring and data acquisition systems to generate the right kinds of information relevant to the priority needs of today (and second - guessing those of tomorrow). This is a significant issue for water information in Europe
where resources for environmental monitoring are already heavily committed to the reporting of compliance with a wide range of legal instruments, some of which are now quite old. The development of indicators has tried to make the best use of available information, but we now need to move from a position of "best available information" towards "best needed information". This will require better communication to bridge the gap between information users and providers. It will also require a re-thinking of what exactly are our priority knowledge needs to support policy-making and environmental management. This is essential if we are to obtain the best value from limited monitoring resources by eliminating redundancy and re-focusing programmes to deliver priority information needs.

Figure 6. Water consumption in different sectors
There is now a developing consensus in Europe that we must modernise our monitoring and reporting systems. This was reflected in the conclusions of an international conference "Bridging the Gap" - new needs and perspective for environmental information, London, June 1998 which stated that:

"At present some of the systems for gathering information about the environment in European countries are inefficient and wasteful. They generate excessive amounts of data on subjects which do not need it, and they fail to provide timely and relevant information on other subjects where there is an urgent policy need for better focused, and for consistent environmental assessment and reporting".

This statement is just as relevant for water as it is for other media. Indicators have a crucial role in helping to focus the programmes of information providers according to the priority needs of information users.

![Figure 7. The percentage of households spending more than three per cent of income on water charges in Great Britain](image)

**CONCLUSIONS**

- For practical reasons indicators have tended to concentrate on making best use of the information already available, rather than framing the priority questions first and then defining what information is needed to answer them. More emphasis needs to be placed on achieving a consensus on what are the priority questions to be asked to support water management in Europe. This will then provide the basis for targeting and modernising the monitoring systems which generate the data streams.

- There is already a huge amount of data available on the water environment, most of which relates to basic pressures (discharges/abstractions) and states (quantity/quality). Indicators have an important role in obtaining best value from this information by making it available in a way which is meaningful to a range of target audiences.

- The majority of the indicators proposed for water relate to "what is happening?" questions by presenting basic information on trends. These kinds of indicators are often difficult to interpret because they do not allow value judgements to be made. Some indicators in common usage do show trends with respect to certain standards and targets, but many of these derive from old legislation and their relevance and scientific validity need to be re-examined.

- Very little attention has been given to the demand side of water management in indicator development. It is becoming increasingly important that we understand the relative efficiencies of resource usage in the different socio-economic sectors to support and to promote the sustainable use of water resources. This will require more interactive use of social, economic and environmental information.
• There are some outstanding gaps in the information base needed to support current and future water management needs. Some priority information needs are:

- diffuse sources of pollution;
- emerging issues on human and ecological health (such as endocrine-disrupting chemicals);
- patterns in the consumption of water and the efficiency of resource usage;
- the relationship between socio-economic driving forces in different sectors and environmental impacts;
- indicators of the contribution of water to the overall quality of life;
- the ability to assess future outlooks and to assess long-term environmental change, particularly in the context of global climate change; and
- the effectiveness of policy and legislation concerning the water environment.

• The resources required to fill these gaps will have to be found through re-balancing and modernising water monitoring and information systems in Europe. It is important that the current redundancy in reporting requirements is removed so that resources can be released and targeted upon meeting new priority information needs.

REFERENCES


U.S. WATER QUALITY CRITERIA AND STANDARDS: EVOLVING MANAGEMENT TOOLS

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Water quality criteria, as a means of distinguishing ‘good’ water from ‘bad’ water, have been employed for over 150 years. During the 1940s and 1950s, the formulation of water quality criteria and their role in water quality management, were hotly debated in the U.S. as each state formulated its own approach to water quality pollution control. In 1965, when the Federal government assumed control of water quality management in the U.S., a single, national approach to the formulation and use of criteria had to be selected and implemented nationwide.

The purpose of this paper is to briefly review the history of criteria as a basis for water quality management in the U.S.; note the different water quality management philosophies used by states prior to 1965; and summarize the relationship of criteria to monitoring over the past 30 years.

Standards, as legal instruments, are developed for a specific water body by designating a use of the water (society value) and the criteria necessary to protect the use (a scientific finding). Standard violation, as a concept, tends to remain as a single-sample exercise (ie, the sample measurement either is above or below the standard). Increasingly, however, there is a demand to classify large bodies of water (eg, lake, river segment, or aquifer), over a set time period, such as a day, year, or decade, as either in compliance or in violation of water quality standards. No peer reviewed, standardized method for computing standard ‘violations’ has been widely accepted for this purpose.

The paper ends with a call to develop such methods, if water quality managers must produce standard violation information (infer sample results) over time and space.

Key Words: Water quality criteria, standards, monitoring, standard violation, history.

INTRODUCTION

Definitions of water quality criteria, as a ‘tool’ for managing water quality, have evolved over the years. The concept of quantifying acceptability of water for a given use was formulated in Europe over 150 years ago in response to the unsanitary conditions resulting from the industrial revolution. During the late 1800s and early 1900s, ‘water quality management’ involved investigating health and nuisance problems, formulating a solution, and, through implementation of the solution, solving the problem.

During 1920’s and 1930’s in the United States, it became apparent that more waste was being discharged, on a rather regular basis, into the environment than the environment could assimilate in a safe manner. Thus, the focus of water quality management at this time shifted from solving specific problems, as they arose, toward controlling pollution on a regular basis. ‘Pollution control’, as a term, began to appear in the names of government agencies charged with managing water quality. At this time, criteria and standards focused on the limits of waste permitted in an effluent. The goal was to tie the effluent standard to protecting a beneficial use of the water in a river or lake.

Arguments developed as to what constituted a water quality criterion and water quality standard, what form they should take, and how they should be used within a water quality management program. At the same time, considerable discussion regarding water quality management strategies to be employed, at both the federal and state levels, was taking place in the U.S. Wolman (1940) presents a picture of this dialogue and notes that the standards being requested at the time, in his opinion, would create a water environment of 1500 surrounded by a land environment of 1940.
In the 1960s, pollution control alone did not seem sufficient to address the growing environmental contamination problems. At this time the concept of managing water quality in a more integrated manner emerged in legal form. Criteria and standards became more targeted to routine efforts to 'control' the quality of water in a stream or lake. Focus of management, therefore, expanded to embrace ambient water quality conditions.

Until 1965, the United States federal government did not 'direct' water quality management efforts – individual states had the right to pursue state-based water quality management strategies. States, as would be expected, opted for a number of different management strategies. New York, for example, chose a uniform stream standard approach to managing water quality. Pennsylvania chose a discharge permit approach. California, because of its diverse environments (from desert to redwood forests) chose a flexible criteria and standard approach. In California, it became obvious that for a flexible approach to work effectively, more information about the effects of pollution on different uses of water was needed.

The California State Water Pollution Control Board appointed, in 1950, a Committee on Water Quality Criteria to examine how water quality managers were to get information to make judgments on levels of constituents in water that affected specific uses of the water. The Committee's progress report (California Water Pollution Control Board, 1950) was published in the Journal of the American Water Works Association. The purpose behind publication was to obtain a 'peer' review by the larger water management community. The report described the:

1. Water quality management concept employed in California;
2. Method of analysis for solving water quality problems; and
3. Water quality requirements needed to support various uses of water.

The 'requirements' received a negative response from reviewers. Many reviewers felt there was a need for a more thorough review of the literature. The result, after a number of iterations and updates, was the publication by McKee and Wolf (1963). The preface to the first edition of McKee and Wolf's "Water Quality Criteria" stated the following:

"A prominent feature of the new legislation for water-pollution control in California is the provision for a "case-by-case" study of each problem of pollution, with abatement requirements to be based upon the economics of waste treatment was well as upon the injury to potential beneficial uses of polluted waters. Of necessity, this "case-by-case" stipulation precludes a broad-brush treatment on a state-wide, or even regional, basis, and it rules out rigid stream standards and the arbitrary zoning of the streams or underground basins. In effect, then, it depends on astute judgment by members of the water-pollution-control boards and such judgment must be founded on the most recent, complete, and reliable compendium of data pertaining to the limiting and threshold concentration of each potential polluting substance, for each possible beneficial use of the water."

The information assembled by McKee and Wolf (1963) has been heavily utilized by water quality agencies all over the U.S. and the world. It remains today an impressive attempt to assemble the existing knowledge regarding the impact upon various uses of water stemming from various levels of constituents known to exist in water.

McKee and Wolf's (1963) contains an extensive review of the literature on many water quality constituents, presenting effects on use of various concentrations of each constituent. No judgments are made as to the best or one criterion to be used in all situations – this choice is left to judgment of the professional solving the particular problem, on a case-by-case basis.

The various editions of McKee and Wolf's criteria documents, developed during the 1950s and early 1960s, were the basis for considerable discussions of water quality criteria and standard formulation during this time in the United States. The literature of this period has a number of arguments for and against the:

1. Types of standards used in water quality management; and
2. Criteria used within a particular type of standard.

The idea of zoning, or classifying water, in order to protect specific uses of the water was gaining momentum during this period of time (1950-1965), particularly among those who felt that formalized criteria were needed as a necessary basis for any water quality management
strategy. It was proposed that separate standards be set for each stream or zone of the stream. A water quality standard, as a legal management ‘tool’, was considered, by the late 1960s, to consist of a designation of the use of the water to be protected, a criterion (concentration of constituent) required to protect the use, and an implementation plan. Definitions of criteria and standards, however, were debated well into the 1960s and early 1970s, as the following sections of the paper will show.

There were those (McKee and Wolf, in particular) who felt that the case-by-case approach to water quality management was best. It permitted a professional to assess the situation and develop appropriate criteria needed to protect the designated use. Such an approach may result in different levels of water quality existing in different streams where use of the water is the same. Extenuating circumstances, such as natural conditions, could be factored into the criteria determination by the water quality management professional.

Until 1965, Federal water quality laws avoided requiring criteria and standards. Within the emerging Federal-State cooperation, however, the Federal government was suggesting use of criteria and standards. The following quote from McKee and Wolf (1963) describes the pre 1965 relationship between the states and federal government.

"Congress recognizes that primary responsibility in the field of water-pollution control rests with the states. The federal role in water pollution is to provide technical services and financial aid to states, interstate agencies, and municipalities. A 'Suggested State Water Pollution Control Act' has been issued by the Public Health Service, and as of July 1961 its principles had been used by 40 states. An explanatory statement with respect to classification and standards is as follows:

'Some agencies in the administration of water pollution control programs have classified the waters of the State according to their use and have established standards of quality for such waters in accordance with their respective uses. Proponents of this method have urged that classification and setting of standards is an essential element of any comprehensive program and also that no enforcement action can be undertaken without determination of the use to which a particular body of water should be put and the degree of the quality which the water must have in order to be suitable for such use. This approach, however, has been severely criticized by others who maintain that the process is administratively difficult and time consuming, that classifications, once made are hard to change and tend to create vested interests, and that the tendency will be to reduce waters to the level of mere carries of wastes because of the pressure of special interests.

'The Suggested State Water Pollution Control Act authorizes the agency to classify waters and set up standards of quality for water falling within particular classifications but does not make it mandatory to do so. Classifications and standards once promulgated have a definite legal effect and their violation is made unlawful; conversely, discharges which comply with such classifications and standards are not pollution within the meaning of this Act. In view of the number of persons affected by classifications and standards, it is required that their adoption be preceded by a public hearing open to all residents of the area affected, and that adequate notice thereof be given. The Act provides that in classifying waters and setting standards the agency will be guided by the principle of constantly seeking to improve water quality and upgrading streams for progressively higher uses to the maximum extent practicable.'

The Suggested Act itself deals with the above subject matter in Section 6 as follows:

'Section 6. Classification of Waters; Standards of Water Quality.
(a) In order to effectuate a comprehensive program for the prevention, control and abatement of pollution of the waters of the State, The Board is authorized to group such waters into classes according to their present and future best uses for the purpose of progressively improving the quality of such waters and upgrading them from time to time by reclassifying them, to the maximum extent that is practical and in the public interest. Standards of quality for each such classification consistent with best present and future use of such waters may be adopted by the Board and from time to time modified or changed. (States which adhere to the 'prior acquisition doctrine' should take cognizance of the effect that classification will have on water rights established under other laws.)
(b) Prior to classifying waters or setting standards or modifying or repealing such classifications or standards the Board shall conduct public hearings in connection therewith. Notice of public hearing for the consideration, adoption or amendment of the classification of waters and standards of purity and quality thereof shall specify the waters concerning which a classification is sought to be made or for which standards are sought to be adopted and the time, date and place for such hearing. Such notice shall be published at least twice in a newspaper of general circulation in the area affected and shall be mailed at least twenty days before such public hearing to the chief executive of each political subdivision of the area affected and may be mailed to such other persons as the Board has reason to believe may be affected by such classification and the setting of such standards.

(c) The adoption of standards of quality of the waters of the State and classification of such waters or any modification or change thereof shall be effectuated by any order of the Board which shall be published in a newspaper of general circulation in the area affected. In classifying waters and setting standards of water quality or making any modifications or change thereof, the Board shall announce a reasonable time for persons discharging wastes into the waters of the state to comply with such classification or standards, unless such discharges create an actual or potential hazard to public health.

Any discharge in accord with such classification or standards shall not be deemed to be pollution for the purposes of this Act.

"Whether or not this new and suggested legislation indicates eventual application of federal standards to interstate and navigable waters remains to be seen."

Before discussing the evolution of criteria and standards after 1965, let us first look at some definitions that were rather fluid in the 1960's, and continue to evolve even to this day.

DEFINITIONS OF CRITERIA AND STANDARDS

The difference between a water quality standard, requirement, objective, criteria, guideline, etc., has been the basis of considerable discussion over the years. Streeter (1949) defines the term "standard" as follows:

"The term 'standard' as here used does not necessarily mean a fixed and immutable legal requirement, or established regulatory criterion, but often merely an operational goal, or objective, towards which concerted efforts to maintain or restore desirable stream conditions may be directed."

McKee and Wolf (1963) present criteria that can be used as means of developing a standard. 'Criteria', in McKee and Wolf (1963), are interpreted to be information that portrays various levels of water quality and the resulting impacts upon a use of that water. With these criteria, a judgment can be rendered as to the quality needed to meet a use and a standard can be determined. The standard is then used to judge (a second judgment) whether water is fit for a given use.

The criteria published since McKee and Wolf (1963) are interpreted to be the "standard" as defined by McKee and Wolf. In other words, the criteria contain the judgment as to which quality level is acceptable. The standard is then defined as consisting of a classification of the water with respect to its intended use, criteria necessary to protect that use, and a plan for implementation of the standard.

Criteria published by McKee and Wolf (1963) would result in standards being developed with possible variations for the same uses. This would be due to the differences in judgments as to what quality levels are acceptable for different uses. Later criteria avoided this situation by including that initial judgment in the development of criteria. Thus, generally, all waters designated for a given use would have the same standard.

"Criteria," as defined by McKee and Wolf (1963), relate to the accumulation of scientific data to serve as yardsticks of water quality. The criterion is a fair and truthful description of the effects, on a use, of various levels of concentration of a particular constituent. Criteria are not goals or standards or requirements (guidelines). They are not legal nor do they describe an ideal condition -- they simply state, in an unbiased and scientifically sound manner, what are the effects of pollution on a particular use.
McKee and Wolf (1963) define a standard as "any definite rule, principle, or measure established by authority," and note that since a standard is definite and established by authority it is rigid, official or quasi-legal. Since a standard is established by a legal authority, there is no requirement that a standard be fair or scientifically based. This situation may arise when standards are established in the absence of sound technical information. In this case, standards are often set somewhat arbitrarily with a large safety factor incorporated. This last point is being extensively discussed in the current Total Maximum Daily Load (TMDL) debates in the United States. In other words, is the reason the U.S. has so many TMDL assessments to conduct because the standards were not properly established?

**FEDERAL CRITERIA PUBLICATION DEFINITION**

The Federal Water Quality Act of 1965 (PL 89-234) ended the states' rights to establish their own form of water quality management. Instead a uniform management strategy was described and minimum requirements had to be met by each state. PL 89-234 mandated that all state water quality management efforts had to establish stream standards for all state waters. The criteria to be used in establishing standards would be established and provided by the Federal Government. The argument, over the form and role of criteria and standards, was 'won' by those who advocated a uniform criteria and standard setting process employed across the country.

A U.S. water quality criteria book was published in response to PL 89-234. The National Technical Advisory Committee (NTAC, 1968) presented criteria that States could use to meet the requirements of PL 89-234. This report defined a standard as follows: "a plan that is established by a governmental authority as a program for water pollution prevention and abatement."

A criterion was defined as: "a scientific requirement on which a decision of judgment may be based concerning the suitability of water quality to support a designated use."

The NTAC (1968) report further elaborated on the definition of a standard by noting: "The standards adopted by the States include water use classifications, criteria necessary to support these uses, and a plan for implementation and enforcement."

The NTAC (1968) report is commonly referred to as the "Green Book" due to its cover color. The NTAC (1968) criteria, as noted in EPA (1976), are a major departure from the criteria presented by McKee and Wolf (1963). As noted earlier, McKee and Wolf (1963) presented considerable information on quality levels and associated effects on use. NTAC (1968) presented a given value of a constituent above (or below) which a use would be harmed. Thus, the NTAC (1968) was more than a presentation of the state-of-the-art of water quality effects on uses -- it included a judgment as to which was the one level below (or above) which a use was protected. The ability to adjust criteria, based on specific watershed needs, had been greatly reduced.

NAS/NAE (1973) and EPA (1976) are revisions of the NTAC (1968) criteria and in each case the latest finding and judgments are utilized to develop the criteria. The NAS/NAE (1973) report is often referred to as the "Blue Book" and the EPA (1976) report is commonly referred to as the "Red Book," again due to cover colors.

The NAS/NAE (1973) report defines criteria as "the scientific data evaluated to derive recommendations for characteristics of water for specific uses," and a standard as the "definition of acceptable quality related to unique local situation involving political, economic and social factors and including plans of implementation and questions of water use and management."

The NAS/NAE (1973) report further notes that the words "criteria" and "standards" are not interchangeable nor are they synonyms for such commonly used terms as "objectives or goals." Thus, between the NTAC (1968) report and the NAS/NAE (1973) report, the idea of standards as objectives in water quality management was completely removed.

The EPA (1976) criteria report defines a "criterion" as "a constituent concentration or level associated with a degrees of environmental effect upon which scientific judgment may be based," and notes that with respect to the water environment, criteria have come to mean "a designated concentration of a constituent that when not exceeded, will protect an organism, and organism community, or a prescribed water use or quality with an adequate degree of safety." A standard refers to "a legal entity for a particular reach of waterway or for an effluent."
A standard may contain a criterion, an extreme quality limit, as a basis of regulation or enforcement. As noted in the NAS/NAE (1973) report, however, a standard may differ from a criterion due to local conditions, either natural or economic, or due to the desire to have different safety factors.

**FORM OF CRITERIA TODAY**

In general, water quality criteria in the U.S. today are ‘based solely on data and scientific judgments on the relationship between pollutant concentrations and environmental and human health effects.’ Water quality criteria do not consider economic impacts or technological feasibility in achieving chemical concentrations in ambient waters. The criteria issued by the U.S. Environmental Protection Agency are to be used by states and tribes in establishing water quality standards that ultimately provide a basis for controlling discharges of pollutants (U.S. Environmental Protection Agency, 1999).

Water quality criteria are issued to protect both aquatic life and human health. The purpose of human health criteria, for example, is to estimate the ambient water concentration of a pollutant that does not represent a significant risk to the public. Ambient water quality criteria for human health are primarily based on two types of biological endpoints: (1) carcinogenicity and (2) toxicity (i.e., all adverse effects other than cancer). In some cases, criteria may be based on organoleptic effects (thresholds for taste or odor). There are essentially two procedures for assessing health effects - one that addresses carcinogens and one that addresses non-carcinogens. Two methods are deemed necessary for the purpose of deriving ambient water quality criteria because carcinogenicity is regarded as a non-threshold phenomenon, whereas toxicity is regarded as having a threshold below which there will not be an effect (U.S. Environmental Protection Agency, 1990).

Following the Red Book, a "Gold Book" and "Silver Book" were produced. Today, of course, the criteria and standards used in water quality management are located on the internet. The national recommended criteria are summarized at the following home page: www.epa.gov/ost/pc/revcom.pdf. Colorado standards, as an example of how the national criteria are converted into state water quality standards, can be found at: www.cdphe.state.co.us/regulate.asp.

The current push toward a more integrated approach to watershed management seeks more flexibility to establish standards that reflect actual watershed function. In many ways, efforts to better match water quality standards to local environmental conditions is a return to the approach of McKee and Wolf. The standard setting process, however, involves much more scientific input on a broader range topics than occurred in the 1950s. For example, ecosystem health, as defined by the needs of endangered species, is often establishing the criteria and standards based on local ecosystem needs rather than a national criterion. Available science and data permit this approach to be seriously considered as an option to the ‘one size fits all’ approach to standard setting used in the 1960s and 1970s. However, the costs involved in conducting a scientifically sound assessment for each river reach, lake, wetland, and aquifer is of concern.

**MONITORING COMPLIANCE WITH WATER QUALITY STANDARDS**

If water quality standards, carefully crafted from scientifically sound criteria, are to be an effective foundation for modern water quality management, they must be enforceable. Should every sample result, exceeding a particular standard, be cause for management action? Clearly there are practical considerations, such as naturally occurring ‘contamination’, spatial/temporal variability in constituents, and measurement error, that could make such an interpretation of monitoring results an enforcement nightmare.

The means of interpreting monitoring results, for purposes of judging whether a water body (in total) is in compliance with its standards, is not generally included in the wording associated with a standard. Rather a standard assumes that the sample is the water body – the need to infer beyond the sample, both in time and space, is generally not considered in preparation of criteria and standards. The long-term exception to this is the microbial drinking water standards that traditionally required at least 5 samples over a 30-day period with the average being judged against a limit.
Prior to 1965 in the U.S., there was no federal legal requirement that states ‘monitor’ water quality. With passage of PL 89-234, States not only had to establish water quality standards, they also had to establish a means for the standards to be ‘implemented and enforced’ (i.e. a plan for implementation). Monitoring was deemed a key element in knowing a stream’s quality to support implementation and enforcement of water quality standards. States had to develop monitoring plans as part of complying with PL 89-234.

In general, monitoring programs were developed to ‘check’ standard compliance by taking samples regularly from streams and lakes, checking the sample results with the applicable standard, and deciding, based on the sample’s findings, that the water body is either in compliance or violation of the standard. To develop a conclusion regarding the compliance of a stream or lake with applicable standards over a given time period, say a year, involves inferring the samples’ findings over potentially large spatial and temporal scales. There have been a number of efforts since the early 1970s to develop methods to make such inferences. An example of guidelines to prepare reports describing general compliance with water quality standards is presented in U.S. Environmental Protection Agency (1997).

The listing of waters that violate standards in the United States is required under Section 303(d) of the Federal Water Pollution Control Act of 1972 (this act, as amended, is commonly referred to as the Clean Water Act). Other sections of the act also request, or imply, that information about the Nation’s water quality conditions is accurately known (Sections 106(e), 204(a), 305(b) and 314(a)).

Section 303(d) requires that:

"(A) Each State shall identify those waters within its boundaries for which the effluent limitations required by section 301(b)(1)(A) and section 301(b)(1)(B) of this title are not stringent enough to implement any water quality standard applicable to such waters. The State shall establish a priority ranking for such waters, taking into account the severity of the pollution and the uses to be made of such waters."

The method to be employed to create lists of waters (e.g. stream segments and lakes) not meeting standards is not spelled out in the law. Currently, the U.S. Environmental Protection Agency recommends that States use the following interpretation of monitoring data for purposes of judging whether waters are supporting the uses designated in the standards:

- ‘Fully Supporting’ - waters that have less than 10% of samples exceeding the standard;
- ‘Partially Supporting’ - waters with between 11 and 25% of samples exceeding the standard;
- ‘Not Supporting’ – waters with greater than 25% exceeding the applicable standard.

The exact means for making such assessments varies from State to State, according to Martin (2000).

Once a stream or lake appears on the 303(d) list, the U.S. Environmental Protection Agency must approve the means by which a State proposes to bring the stream or lake into compliance with its standards. Section 303(d) (c) states:

"(C) Each State shall establish for the waters identified in paragraph (1)(A) of this subsection, and in accordance with the priority ranking, the total maximum daily load, for those pollutants which the Administrator identifies under section 304(a)(2) of this title as suitable for such calculation. Such load shall be established at a level necessary to implement the applicable water quality standards with seasonal variations and a margin of safety which takes into account any lack of knowledge concerning the relationship between effluent limitations and water quality."

Thus, when a stream appears on the 303(d) list an assessment of how much load the stream can tolerate, and still meet its standard, must be conducted. The result of the assessment is the computation of the ‘total maximum daily load’. This TMDL is generally defined to consist of the following additive components: (1) a wasteload allocation amount of the offending constituent (to be allocated to point sources of pollution via permit limits); (2) a load allocation (to be allocated to non-point sources of pollution); (3) an amount of load to be allocated to future growth (if desired); and (4) a load to be assigned as a margin of safety.
The legal requirement to ‘implement’ the applicable standards, even when the source of the pollution may be from non-point sources (for which there is no regulatory mechanism to control), has spawned a number of ‘TMDL’ lawsuits in the United States. These lawsuits are often successful. It does not appear that the lack of a legal enforcement mechanism (e.g. permits) for non-point sources of pollution is acceptable as a reason for not complying with the applicable standards.

The reporting of water quality conditions in the United States is required under Section 305(b) of the Clean Water Act. In implementing the provisions of Section 305(b), the U.S. Environmental Protection Agency (1997) requires each State develop a program to monitor the quality of its surface and ground waters and prepare a report describing the status of its water quality. EPA then compiles the data from the State reports, summarizes them, and transmits the summaries to Congress along with an analysis of the status of water quality nationwide. This 305(b) process is the principal means by which EPA, Congress and the public evaluate whether U.S. waters meet water quality standards, the progress made in maintaining and restoring water quality, and the extent of remaining problems. In 1996, 56 States, territories, Interstate Commissions, and Indian Tribes prepared 305(b) reports (U.S. Environmental Protection Agency, 1997).

Recent 305(b) reports state that 40% of the nation’s waters are not supporting the uses stated in their standards primarily due to non-point sources of pollution. Thus, many States have a large number of stream segments and lakes for which they must allocate waste loads in a manner that results in the water complying with its standard. The magnitude and cost of the management activities needed to perform TMDLs is causing a re-examination of the methods used to establish standards in the first place and, then, to create both the 305(b) report and 303(d) lists. This re-examination has focused considerable attention on the relationship between management decision-making and the way information is obtained from monitoring to make management decisions. It appears that water quality management has not developed a sufficiently strong connection – a connection that will stand up to a court challenge (Public Employees for Environmental Responsibility, 1999; and General Accounting Office, 2000).

Defining and measuring standard compliance, in a transparent and auditable manner, is not a new problem. During Monitoring Tailor-made I, Ellis and Newman (1994), for example, examined four ‘compliance rules’ to determine comparability of compliance conclusions. They concluded that unqualified ‘comparisons of river quality between Member States should be made only where the respective classification systems have a common basis of definition and interpretation’.

Public Employees for Environmental Responsibility (1999) and General Accounting Office (2000) both indicate monitoring problems with the 305(b) reporting process, especially given the important role such information plays in making major water quality policy decisions. The problems, in a fundamental manner, stem from the desire, on the part of Congress and the EPA, to permit States as much latitude as possible in meeting water quality standards, even to the point of designing individual monitoring and reporting systems. As a result, EPA is often compiling different types of data and information into a national, comparative assessment.

Martin (2000) examined the various methods being employed today to measure compliance with water quality standards. She found that a variety of strategies are used to convert water quality data into standard compliance assessments. To increase the comparability of the information from monitoring, relative to standard compliance monitoring, interpretation and reporting, will require much more coordination in monitoring system design and operation than has occurred in the past.

As presented in the paper by J.M. Klein (during this MTM-III workshop), the U.S. is attempting to develop more ‘comparability’ in water quality data and information via the National Water Quality Monitoring Council. Given the absence of a legal requirement at the federal level to perform monitoring using ‘mandated’ common methods, the Council is attempting to formulate ranges of methods, which employed in a voluntary manner, will result in comparable data and information.
SUMMARY AND CONCLUSIONS

Water quality criteria and standards have evolved over the years without establishing a strong connection to the manner in which compliance or violation is measured and interpreted across time and space. As protection of water quality becomes increasingly important to the general public, water quality managers are being asked to increase the sophistication of their strategies and greatly enhance the consistency and thoroughness of the information employed within management decision-making.

Conferences, such as the Monitoring Tailor-made, play a large role in developing better connections between management decision-making and the water quality information produced by monitoring. Hopefully, this paper has provided a brief overview of the history of water quality criteria and standard development and the emerging need to better relate monitoring standard compliance and violations over time and space in the U.S. Much more attention will need to be focused on this connection if the water quality information needs of the public are to be met and connected, in a transparent and auditable manner, to the actions of water quality managers.

REFERENCES

THE CONTEXT OF ASSESSMENT

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Substantial diversity exists of assessment methods and assessment criteria used over Europe. The character of pollution principally differs depending on the degree in which the pressures are under control. Rather natural rivers can be found in Central and Eastern Europe, where heavy pollution may occur locally and temporary. Western European rivers are often heavily modified because of high demands for ship traffic and energy production. An adequate assessment of a water system takes into account the socio-economic context and the water management concerns. Political priorities are expressed in assessment criteria.

INTRODUCTION

For assessments used for the management of a water system, information on all aspects in a decision making process is required. These should take into account the whole causality chain of how the water system is related to the processes in society. The scheme in figure 1 as an example (EEA, 1998) presents an assessment which does not restrict to the state of the water. From our past monitoring and assessment practices we are used to monitor the state, sometimes the impact, and we assess the state and report the state, we are somewhat addicted to the state. We want to manage, however, the environmental pressures and its socio-economic drivers, and the response of the society. There is no interest of the social community in the state, if there is no impact on human welfare (Harremoës and Turner 2001).

Figure 1. DPSIR-framework for environmental assessment (EEA, 1998).

It should go without saying that no monitoring starts without the goal of the assessment has been defined and the questions to be answered has been specified. These goals can be very specific, and very restricted. Not all assessments intend to be integrated assessments, and have to follow a holistic approach. But when monitoring and assessment has to support the whole decision making process, for example the management of a river basin, than the assessments have to take into account the specific context and developments of the water system, its specific problems and characteristics.
I would like to touch on five points concerning the context of assessment:
Firstly, knowledge of economic and social developments in past and present make that results of measurements are interpreted in a different way.
Secondly, functions of a water system may conflict, a function can even structurally damage other functions of the water system, they are subject of political priority setting and we use these political priorities as a basis for assessment criteria.
Thirdly, the character of water pollution may be fundamentally different for different socio-economic contexts and we should be aware of it when designing a monitoring and assessment system.
Fourthly, concerning assessment criteria and assessment results we face enormous differences in defining what is ‘good’ and what is ‘bad’.
Fifthly, the lack of knowledge remains ‘the ugly’? Only a restricted number of countries in the world measure what they need to know.
Finally, the former urges for renewed encouraging words for tailor-made assessments.

ECONOMIC DEVELOPMENTS

Concerning European rivers, the differences between east and west are meaningful. Last thirty years changed the face of Europe. We came from terribly bad conditions, with characteristics like excessive SO2 emissions, acid rain, untreated waste waters, rivers as open sewers, etc.

Intensive legislative efforts were made. Especially in a number of western countries action plans, combined with big investments initiated the rehabilitation of some river basins. For example, approximately 70 billion USD were invested in the rehabilitation of the river Rhine (Adriaanse and Broseliske, 1997).

East European rivers dropped far behind with their old-fashioned, dingy industries, refuse dump sites, devastated mine tailing areas, failing waste water treatment. The water-quality of these rivers was told to be extremely bad and their ecology deteriorated. This picture is changing: the quality of Central and East European rivers is not that very bad and discouraging as the general opinion makes us believe. Besides, the prospects for rehabilitation are rather good.

What happened in Eastern Europe? The transition of central and east European economies and the economic decline during the 90's caused a bankruptcy for an significant part of the industrial sector. Also agricultural practices changed significantly: the use of pesticides and fertilisers was more and more restricted by economic factors. These factors in turn caused a significant improvement of the environment. It is true that on the other side budgets for maintenance of installations for industrial production and waste water treatment are under pressure.

Since several years there are signs of economic stabilisation. But what will happen when the economic situation structurally starts to improve? This could easily lead to a substantial new conflict of environment and economy and a decreasing quality of surface and groundwater’s. However, it is challenging that new investments often have the possibility to use clean production processes based on high technology solutions.

In this respect the role of legislation is of utmost importance. The integration of interests of environment and economy have to be encouraged and strict standards are to be set in the new legislation concerning permits for new industrial activities (and also for the use of fertilisers and pesticides in agriculture).

CONFLICTING FUNCTIONS

The specific functions of European rivers make them very different. The same achievements can not be met for all rivers. Improving navigability can easily be conflicting with the ecological functioning.

Once again the river Rhine can be used as an example. The river flows through one of the worlds most densely populated area's, is the source of drinking water for more than 20 million people and has 20% of the world production of chemicals along its borders. At the same time it has the worlds highest ship traffic density between Duisburg as the world's biggest inland
harbour and Rotterdam as the world's biggest sea harbour. This river has turned in a 30 years period from an extremely polluted river in the 60's to a river where these years the salmon is making its first appearance.

But the high economic importance of this river and the immense investments made in the last century in building of dams, dikes, weirs and canalisation works, for navigability and for the generation of electricity, caused structural damage to the hydromorphology of this river and made that the ecological function of the river Rhine can never be restored in the same order as for rivers like Vistula or Oder. It is true that the latter rivers may suffer from high pressures by lack of sufficient waste water treatment from municipalities and industries. But the damaged ecological functioning of these rivers may be restored in a rather short time if specific threats are solved, because of the excellent hydromorphological conditions. Special care should be given to safeguarding these conditions and avoiding developments which structurally damage them.

THE CHARACTER OF POLLUTION

The character of pollution of rivers may be essentially different. It is a unique characteristic of the river Rhine, that accidental pollution has been reduced to a low level. The river is principally under control: production processes have been optimised, emission permits are well elaborated, emissions were safeguarded, storage reservoirs were built to safeguard the river from accidental pollution, important threats are known and under control, emergency warning in case of accidents is well established, regular surveys for screening of potential pollutants are made. Permanently the river is under "intensive care". Ten years ago countries still had to face about 70 messages of the international accident emergency warning system per year, while these years only about 15 are left, mainly due to oil spills by ship traffic.

This is a principle difference with most rivers in other parts of the world. The average pollution concentrations of 'well controlled' West-European rivers are not so very low and the composition of the pollution is very complicated. The bad image of Central and East European rivers is often caused by its occasionally occurring high accidental pollution, while average water-quality concentrations may be rather good (Figure 2). Accidental pollution's are mainly caused by a number of local hot spots, locations at high risk for accidental pollution.
GOOD AND BAD

What is good, and what is bad? A huge diversity exists in criteria against which water-quality is assessed. National water-quality criteria for nutrients easily vary with factor 10 over European surface waters and with a factor 100 for heavy metals (IWAC, 2001). Toxicological as well as political arguments can be the basis for water policy requirements. The role of EU legislation in harmonising the assessment criteria and legislation is important, but a broader scope is needed than the European Union and its accession countries only. Diversity in strategies on a global scale have to be considered as well.

Harmonisation and integration can not be forced, but differences can be made visible and transparent by comparisons. Such information plays an important role as it makes things ready for communication. The next step includes the raising of understanding and consensus on differing assessment systems and the development of a more common approach.

THE UNKNOWN

Good knowledge about the quality of a river is only available for a restricted number of rivers in the world. For most rivers only basic data are available about oxygen content, and few more basic parameters; information about nutrients and heavy metals is scarce. Throughout no information is available about toxicity of the water and sediments, organic micro pollutants (pesticides), PAH’s and PCB’s.

The terms BAI (best available information) and BNI (best needed information) were defined before (Harremoës and Turner 2001). The major part of the world however does not know what is the best needed information, because the quality of its river is a mystery. In between BAI and BNI is a third kind of information: it is the insight obtained from specific surveys, solving the gaps in information, to make a problem analysis possible (Figure 3).

Figure 3. Need and availability of information

Earlier this year was an important meeting of stakeholders of the Kura river in the Caucasus region. It was found that the best available information was very restricted, and the best needed information was not known. It was concluded that there is an urgent need to survey and detect the pollution, so that the problem could be characterised of this river, which is the major source of drinking water for the city of Baku.
TAILOR-MADE ASSESSMENTS

Tailor-made assessments are needed because rivers are different and so are the pollution characteristics. It makes no sense to copy the assessment approach made for such a complex system as river Rhine. Often priority information for a river can be obtained by much easier (but specific!) assessments.

Under the UN/ECE Convention on Protection and Use of Transboundary Waters, it is recommended to countries to find for the implementation of their monitoring and assessment strategy explicitly a link between information and policy making (ECE TFMA, 2000). Pilot projects in eight European river basins under the work programme of this convention learned last two years that sometimes it is a big job to find out what is the real context of assessment, because essential information is missing which is needed to specify what is the problems.

CONCLUSIONS

a. Going beyond the point of structural damage of a river should be avoided, because it will not be able anymore or only with excessive costs to rehabilitate. Assessments should indicate these critical points.
b. The pollution character of rivers may be essentially different. A controlled river flowing through densely populated area with highly developed economic potential, like river Rhine, is essentially different from that of rather natural, uncontrolled rivers which are dominated by frequently occurring accidental pollution's.
c. A huge diversity exist in applied assessment criteria and assessment methods. The first step to harmonisation implies making differences visible by comparison.
d. An overflow of data and figures exists, but most countries have to cope with a striking lack of relevant information about the quality of water resources.
e. Integrated assessments should take into account the specific character and the developments in the socio-economic context of the water body.

REFERENCES

MONITORING APPROACHES IN THE CASE OF ACCIDENTAL POLLUTION: CHEMICAL SPILLS AND MILITARY ACTIVITIES

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The major environmental pollution incidents resulted in the release of different chemicals from breaking pipelines, oil taker accidents, industrial accidents at factories or mining, sabotage, and military activities during peace or war, etc. Most of these pollution incidents affected water resources and required emergency actions, significant mitigation and rehabilitation efforts. The success of these efforts and minimization of the adverse effects depend on the timely obtained, reliable information from immediate and follow-up surveys and assessments. Emergency monitoring and rapid assessment of the hazard and likely exposures are needed for the immediate actions and should be followed by proper monitoring and assessment for long-term rehabilitation.

The monitoring data should allow mass-balance calculations, description of the pathways and the fate of pollutants, differentiation between the incident related pollution and the background or pre-incident pollution levels. On the basis of these data, risk assessment by experts should be made available for decision makers, politicians. The role of the media is important for dissemination of information and the resulted public awareness should be based on valid, reliable data. Each pollution incident has its own specific character and the lessons learned from the earlier incidents should be used in contingency planning ensuring preparedness in coping with future accidents.

INTRODUCTION

A great deal of environmental pollution accidents resulted in the release of crude oil or oil products, such as 20,000 tons from the tanker "Anne Mildred Brovig" in 1966; 120,000 tons from the "Torrey Canyon" in 1967; 95,000 tons from the "Urquiola" in 1976; 230,000 tons from the "Amoco Cadiz" in 1978; 170,000 tons from an oil platform in the Gulf of Mexico in 1979; 155,000 tons from the tanker "Castillo de Bellver" in 1983; 40,000 tons from the "Exxon Valdez", 70,000 tons from the "Khark 5" and 25,000 from the "Aragon" in 1989; 143,000 tons from the "Haven" in 1991; 80,000 tons from the "Aegean Sea" in 1992; 85,000 tons from the "Braer" in 1993; 70,000 tons from the "Sea Empress" in 1996 and 17,000 tons from the tanker "Nachodka" in 1997. As a consequence of military activities, significant amount of crude oil and refined products were released in 1983 from the Nowruz (Iran) off-shore oil wells; during the 1991 Gulf War in Kuwait and the 1999 bombings in Yugoslavia. Concerning other type of chemicals, the largest dioxin accident occurred at the Italian town Seveso in 1976; methyl isocyanate escaped from the Union Carbide factory at Bophal in 1984; pesticides from the Ciba-Geigy and the Sandoz factories at Basel in 1986 resulted in fish kills in the Rheine, and in the same year the largest nuclear plant accident happened at Chernobyl. Around 1,000 tons of dichloroethane were released during the 1999 bombing in Yugoslavia at Pancevo, and in lately, in 2000, the cyanide and heavy metal spills at mining facilities in Romania affected the water resources (http://www.umweltbundesamt.de/uba-info-daten-e/environmental-disasters.htm; Literathy, 1992 and 1993; UNEP/Habitat BTF, 1999).

All of the world’s "famous" pollution incidents contributed to the understanding of the immediate, short- and long-term affects of the particular pollutant(s) in the environment. The lessons learned from such incidents are used by environmental scientists and institutions, as well as by the responsible government agencies for appropriate response and for setting up contingency and monitoring plans ensuring preparedness to cope with pollution accidents in the future.
CHARACTERISTICS OF POLLUTION INCIDENTS

Pollution incidents may happen during peace and war, industrial accidents, sabotage, transportation accidents, and can be initiated by natural disasters such as flushing out hazardous materials by floods, etc. The sources of environmental impacts and the potential for pollution incidents creating emergency situation are summarized in Table 1.

Table 1. Sources and creation of chronic impact and emergency situation

<table>
<thead>
<tr>
<th>Chronic Impact By</th>
<th>Emergency Situation By</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wastewaters:</td>
<td></td>
</tr>
<tr>
<td>• Effluents of municipal treatment plants</td>
<td>• Failures at treatment plants</td>
</tr>
<tr>
<td>• Industrial effluents</td>
<td></td>
</tr>
<tr>
<td>Liquid/solid/hazardous wastes:</td>
<td></td>
</tr>
<tr>
<td>• Waste treatment effluents</td>
<td>• Unsecured disposal of waste</td>
</tr>
<tr>
<td>• Leaching from waste disposals</td>
<td>• Illegal disposals</td>
</tr>
<tr>
<td>Agricultural wastes:</td>
<td></td>
</tr>
<tr>
<td>• Sustainable use of agrochemicals (fertilizers, pesticides)</td>
<td>• Industrial accidents</td>
</tr>
<tr>
<td>• &quot;Safe&quot; disposal of manure</td>
<td></td>
</tr>
</tbody>
</table>

The success of coping with emergency situation to minimize the environmental damage and the possible affects on public health depends on: (a) the existence of contingency plans and preparedness for rapid monitoring and combating, (b) the knowledge on the type and size of the released pollutants, (c) the temporal and spatial distribution characteristics of the pollutants and their access to the water resources, etc. All these conditions are needed for the first assessment of the pollution incident, which can be very different if the accident happens during peace or war, as it is compared in Table 2.

Table 2. Characteristics of chemical spills in peace and war

<table>
<thead>
<tr>
<th>Knowledge - Action</th>
<th>In Peace</th>
<th>In War</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pollutant, hazard</td>
<td>usually known</td>
<td>usually unknown</td>
</tr>
<tr>
<td>Extend of pollution</td>
<td>can be estimated</td>
<td>after the war only</td>
</tr>
<tr>
<td>Affected area</td>
<td>can be located</td>
<td>not accessible</td>
</tr>
<tr>
<td>Start of monitoring</td>
<td>immediate</td>
<td>after the war only</td>
</tr>
<tr>
<td>Immediate action</td>
<td>possible</td>
<td>almost impossible</td>
</tr>
</tbody>
</table>

Hazard associated with pollution accidents

Assessment of the hazard associated with the accidental pollution will primarily depend on the properties of the polluting substance, which
• can be toxic to the aquatic and wild life (to all organisms, or specific groups only);
• can accumulate in the food-chain;
• can accumulate in the particulate matter, sediment;
• an be detrimental to human health through consumption of contaminated drinking water or contaminated food (aquatic products, or plant irrigated with contaminated water).

After recognition of the accidental pollution quick response is needed
• to estimate the type and level of contamination, and
• to identify the likely exposed area (directly or potentially affected).
Risk associated with the pollution accidents, risk management

The environmental and public health risk should be assessed individually in each pollution accident. Although the polluting materials could be the same, e.g., most frequently crude oil or its refined products, the fate and pathways and the exposure of the living organisms to the pollutant should be evaluated on in each particular case. Figure 1. shows the chain of events in the case of pollution accidents, which should be followed to manage and minimize the risks.

![Figure 1. Chain of events in the case of pollution accidents](image)

The success of the management of the risk primarily depends on the appropriate, timely carried out monitoring. This should provide all necessary data and information needed for prediction of the hazards and exposures, which ensures proper, effective decisions and actions. The monitoring should be extended to monitor the affects of the actions and will provide the information for corrective measures if needed.

**MONITORING AND ASSESSMENT**

In the case of accidental pollution the protection of the environmental and public health requires immediate action to start with the assessment of the associated risk in short- and long-term. This assessment requires field surveys and appropriate monitoring, which usually should be implemented in three time-frame as follows:

- Immediate surveys: emergency monitoring;
- Short-term monitoring, and
- Long-term monitoring.

Although the survey and monitoring methodologies could be different during these monitoring approaches, appropriate quality assurance and quality control measures are needed to ensure the reliability of the monitoring results. The reported results can be as informative and accurate as expected from the applied analytical method, however data validation is particularly important in the short- and long-term monitoring. In the case of transboundary pollution, cooperation between laboratories in the neighbouring countries is of vital importance. This cooperation should also include agreed sampling and analytical methodologies, quality control measures, which are prerequisites of obtaining comparable monitoring results.
Determinands to be monitored, methodologies

After recognition of the pollution accident, the pollutant and the supporting variables should be identified, and the analytical method, appropriate for obtaining the necessary information in the required time, should be selected. The design of the representative sample collection requires knowledge on the type and appearance of the pollutant, such as:

- inorganic - organic
- toxic - non-toxic
- synthetic - naturally occurring
- water soluble - non-soluble
- reactive - conservative
- floating - sediment-bound
- degradable - persistent
- bioaccumulable?

Based on this knowledge, the relevant matrix, i.e., water, sediment, biota, shall be sampled. Selection of the analytical method should be done on the “fit for the purpose” principle. Less selective and sensitive but rapid methods, i.e., in-situ probes or test kits, are usually used for field measurements during the emergency monitoring, screening for the pollutants, whereas selective, sophisticated laboratory methods for verification and detailed assessment.

Field kits are available for rapid detection and rough quantification of several pollutants, e.g., heavy metals, sulphides, ammonia, however detection and identification of low concentration of pesticides requires laboratory, chromatographic measurement. In the case of the most frequent oil pollution accidents, the fluorescence fingerprints, particularly the 3D total fluorescence spectra, which require laboratory instrument but the sample preparation and the measurement is very simple and fast, are very effective for both qualitative and quantitative characterization of the oil pollution and pollution caused by polar aromatic compounds (Literathy, 2000).

Automatic water quality monitoring (AWQM)

Since the establishment of the first network of AWQM stations on the Ohio river in 1955, there are pros and cons regarding their roles in the detection of pollution accidents. Several AWQM stations are in operation in major river system, such as on the Rhine river, and their application is described in several publications (AQUALARM, 1990; Boesmans and de Werger, 1987; Philipot et al., 1989). The usefulness of the AWQM station in the detection of accidental spill is questionable because most of the spills have been recognised by information from other sources, e.g., report on transportation accidents, report on industrial accidents, visual observations. It is also short-coming of the AWQM stations that the instruments usually capable to detect a relatively small group of target polluting compounds. However, AWQM stations can be useful to collect supporting data and information during the spill and the follow-up short- and long-term monitoring for recording temporal variation in the water quality.

Immediate surveys, emergency monitoring

Immediate surveys and the emergency monitoring aims to provide fast results on the level of the pollutant(s), expected immediate fate and pathways for rapid assessment of the situation. The information obtained should be transferred to the authorities responsible for combating the pollution and to the downstream water users. After recognition, detection of the pollution incident the immediate survey should be designed on the basis of all available information on the spill, particularly on the nature, type of the polluting compound(s) and the size of the spill. The emergency monitoring should provide fast information to decision makers allowing actions for prevention and reduction of the adverse effects, and for controlling, combating and mitigating the pollution. Because the monitoring should provide fast information, usually there is no time to carry out sophisticated analytical measurement in laboratory. Field methods, usually screening techniques and use of portable kits can provide the necessary information, however, samples should be taken for more time-consuming confirmation tests in laboratories. The target matrix usually the water column, but depending on the type of the pollutant, suspended solids and sediment, as well as indicator aquatic organisms should be also sampled and analysed. The emergency monitoring should last until the contamination will be reduced to the warning level, the permissible concentrations for the aquatic life and the water users.
Short-term monitoring

The main objective of the short-term monitoring is to assess and confirm the consequences of the pollutant spill: (a) effects on the ecosystem functioning (aquatic life, biodiversity), and (b) accumulation of the pollutant(s) in the bottom sediment.

Short-term monitoring should be designed on the basis of the results from the immediate surveys and the emergency monitoring. Determinands to be monitored include both the released pollutant during the accident and the related water quality characteristics, which may play a role in controlling the fate and pathways of the pollutants, may affect the toxic effect being synergetic or antagonistic. The target matrices are the water column (dissolved and particulate matter), the bottom sediment and the biota at all levels (bacteria, phyto- and zoo-plankton, and fish) living in the water and in sediment.

Long-term monitoring

The main objective of the long-term monitoring is to monitor the mitigation of the adverse effects and the progress of rehabilitation of the contaminated area until the accident-related contamination will be reduced to the warning level, or to the pre-accident concentrations, and the aquatic life is revitalized to the pre-accident level. The long-term monitoring programme should be designed on the basis of the immediate surveys and the short-term monitoring results.

The chemical characteristics, determinands in the long-term monitoring usually includes the persistent compounds, which either accumulate in the bottom sediment or in aquatic organisms. Therefore, in most cases the target matrices are the sediment and biota. Long-term monitoring should always concern biological monitoring, which will provide information on the damaged ecosystem functioning, the revitalization of the aquatic life and the rehabilitation of the aquatic environment.

LESSONS TO BE LEARNED: CASE STUDIES

Although each pollution accident has its own special character the evaluation of the situation provides knowledge on: (a) the behaviour of the spilled pollutant, (b) its fate and pathways, spatial and temporal distribution in the environment, and (c) the ways of differentiation between the spilled chemicals and its background, pre-accident concentration levels in the ecosystem. The effects of the pollutant(s) in the environment could be direct or in-direct, in the later case the pollutant could mobilise "inactive" pollutants, such as heavy metals buried in the sediment. Without discussing each of the pollution incidents in details highlighting typical characteristics of selected pollution accidents could be used in preparation of contingency and action plans to cope with future pollution accidents.

Case study 1: 1991 Gulf War in Kuwait

The largest environmental pollution incident by petroleum occurred in 1991 as a result of sabotage and military activities in Kuwait (Literathy, 1992 and 1993). More than one million tons crude oil spilled into the marine environment and an estimated 150 million tons of crude oil or its combustion products were released from the burning and/or gushing oil wells. The monitoring efforts, which started few months later after the war, considered heavy metals, particularly vanadium and nickel having the highest concentrations. Because heavy metals are natural components of the soil/sediment the monitoring approach had to be designed to differentiate the pollution input from the background concentration. Table 3. shows the concentration of five selected heavy metals in the crude oil and their background concentration in marine sediment and the desert soil.
Table 3. Characteristic concentration of some heavy metals in the crude oil and sediment/soil samples in the impacted marine and desert areas.

<table>
<thead>
<tr>
<th>Metals</th>
<th>Kuwait Crude Oil, mg/kg</th>
<th>Average Background Concentration, mg/kg</th>
<th>Desert Soil, in Area/Oil Fields</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Gulf Sediment in Kuwait, &lt; 63 µm fraction</td>
<td>Wafra</td>
</tr>
<tr>
<td>Vanadium</td>
<td>34.7</td>
<td>49.3 ± 3.8</td>
<td>10.3</td>
</tr>
<tr>
<td>Nickel</td>
<td>8.44</td>
<td>116 ± 9.9</td>
<td>13.0</td>
</tr>
<tr>
<td>Mercury</td>
<td>0.125</td>
<td>0.04 ± 0.01</td>
<td>0.016</td>
</tr>
<tr>
<td>Lead</td>
<td>0.090</td>
<td>4.7 ± 1.3</td>
<td>2.90</td>
</tr>
<tr>
<td>Cadmium</td>
<td>0.004</td>
<td>0.21 ± 0.10</td>
<td>0.027</td>
</tr>
</tbody>
</table>

The pollution potential of the metals depend on level of the relative contribution from the polluting material to the background concentration of the same metal in the sediment/soil matrix. This could be expressed with concentration ratio of the metals in the crude oil versus their concentration in the sediment/soil. As shown in Table 4., the mercury has the highest pollution potential in both marine sediment and the desert soil followed by the vanadium and nickel.

Table 4. Concentration ratio of metals in the crude oil and sediment/soil samples.

<table>
<thead>
<tr>
<th>Metals</th>
<th>Concentration ratio of metals</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Crude Oil, Gulf Sediment</td>
</tr>
<tr>
<td></td>
<td>Wafra</td>
</tr>
<tr>
<td>Vanadium</td>
<td>0.70</td>
</tr>
<tr>
<td>Nickel</td>
<td>0.073</td>
</tr>
<tr>
<td>Mercury</td>
<td>3.12</td>
</tr>
<tr>
<td>Lead</td>
<td>0.019</td>
</tr>
<tr>
<td>Cadmium</td>
<td>0.019</td>
</tr>
</tbody>
</table>

From this lesson one can learn: if the pollution incident involves complex mixture of different pollutants, such as the petroleum, the pollution potential of an individual compound in the polluting material will depend on its relative concentration to the background concentration of the same compound in the affected environmental matrix, rather than its absolute concentration in the polluting material.

Case study 2: Pesticide spill in Hungary (1998)

In May 1998, accidental pesticide (β-cypermethrin) spill from a chemical factory downstream of Budapest resulted in localised fish-kills in the Danube. The immediate sampling and analysis indicated the need for sending warning message through the Danube Accident Early Warning System to the downstream countries. At the same time, running of the "DUNAWARN" predictive mathematical model and the subsequent sampling provided appropriate information on the fate of the pollution and the progressing dilution of the pollutant. The assessment of the possible transboundary impact revealed that the pollutant would be diluted to below the toxic concentration within the Hungarian Danube section, so timely information, the "end-of-alert" message was sent to the downstream countries.
This pollution accident demonstrated the usefulness of the early warning systems, particularly in the case when transboundary impact is expected.

Case study 3: Military conflict in Yugoslavia (1999)

As the consequences of the Kosovo conflict and the bombing in Yugoslavia (April-May, 1999) resulted in the release of different petroleum products and chemicals, particularly in the vicinity of Novi Sad and Pancevo, affecting the Danube river. Because of the war situation, monitoring efforts started a few months after the military conflict. The UNEP Balkan Task Force was assigned to make the survey to assess the damage caused by the released pollutants (UNEP/Habitat BTF, 1999).

It was inevitable that the water soluble pollutants, such as ammonium-hydroxide, nitric acid, were diluted and traveled downstream with the water column, and only those pollutants can be monitored, which are either sediment-bound or not water soluble but heavier than the water and may settle to the bottom. Such released pollutants included petroleum products at both Novi Sad and Pancevo, mercury and 1,2-Dichloroethane (EDC) at Pancevo. The sampling campaign was designed to attempt differentiation between pre-bombing pollution from the pollutants released during the bombing.

In the case of the petroleum products, due to the chronic oil pollution along the Danube and particularly at Novi Sad and in the Pancevo wastewater canal, it was impossible to differentiate pollution levels before and after the bombing. The sediment core sample collected from the Pancevo wastewater canal indicated significant petroleum and mercury contamination in the past, i.e., decades before the bombing. The results of the EDC analysis in the water of the Pancevo wastewater canal, as well as the Danube water samples downstream of the confluence of the canal indicated continuous release of the EDC form the Pancevo industrial area. This is demonstrated in figure 2.

The estimated amount of the released EDC was around 1000 ton, the major part of which most likely retarded in the bottom of the 1.8 km long wastewater canal. The need for recovery of the EDC and minimisation of the adverse effects on the Danube, mapping of the disposal of the EDC in the wastewater canal was decided. In February 2000, collection and analysis of water and sediment samples along and in cross-sections in the canal revealed the deposition sites of the EDC and its release into the water column, as demonstrated in figure 3. At two sites, the concentration of EDC exceeded 15% in the sediment. The continuation of the release of the EDC, particularly from the deposition at the middle of the canal, into the Danube is demonstrated with the concentrations measured in the water column even after 10 months of the bombing. This monitoring effort provided the basic information for the feasibility study of mitigation and rehabilitation of the polluted wastewater canal.

![Figure 2. 1,2-Dichloroethane (EDC) in the water (µg/l): Danube, upstream and downstream of the Pancevo wastewater canal, and the canal, four months after bombing in Yugoslavia.](image-url)
One can learn the lessons from this pollution incidents as follows: (a) it is very difficult, or impossible, to monitor the fate of the water soluble pollutants release in war zone, particularly into large water bodies, where the dilution rate is significant, (b) if the released pollutant(s) from the incident is(are) identical to historical, pre-incident pollution, it is difficult and questionable to distinguish between the two sources, and (c) after detection and identification of the accidental release of pollutants, the monitoring should be extended in space and time, as appropriate, to obtain adequate information on distribution, location of the polluting material, which is needed for the effective combating and mitigation.

Case study 4: Cyanide and heavy metal spills in the Tisza river basin (2000)

Major chemical spills, discharging cyanides and heavy metals into the Tisza river basin in Romania resulted significant damage on river’s ecosystem. Details of the monitoring results and assessment of the consequences of the spills are discussed in another paper in this workshop (Laszlo et al., 2000). Two lessons could be learned, however, as follows:

(1) The appropriate monitoring provide all necessary data to characterise the movement of the pollution plum, the decrease of the peak concentration of the cyanide as a result of dilution. The prediction modelling allowed early warning, providing expected peak concentrations in the downstream river reaches. According to the forecasting samples were collected from the Danube at Pancevo and from the Iron Gate reservoir and near the dam (N.B. It was fortunate that the mapping of the EDC in the Pancevo wastewater canal was at the same time when the cyanide spill reached this Danube reach). Despite the expectation of the retention of the water in the reservoir, the cyanide peak concentration was measured in the overflowing water at the dam at the time calculated with the normal flow velocity. This result demonstrated that such conservative accidental pollutants could be a tracer for calibrating mixing-, prediction models, and proved that the polluted water body was “floating” on the top of the water in the reservoir.

(2) Dissemination of uncontrolled “expert” opinion through the media created panic among the public, misleading information had long-term negative effects in utilisation of the water resources, damaging fisheries and tourist enterprises. Incorrect, unrealistic control measures,
such as using iron-sulphate, sodium hypochlorite, sodium thiosulphate and ozonization for elimination of the cyanides in the water were recommended and publicised. Warning for the accumulation of the cyanide in the fish meat and its long-term effects and existence ("We can not eat fish from the Tisza river during the next 3-4 years due to accumulation of the cyanides") mislead the public. From that lesson it is obvious that the public information and awareness should be based on scientifically sound evaluation of the pollution incidents.

DISSEMINATION OF INFORMATION ON POLLUTION INCIDENTS

It is acceptable that even experts can make pessimistic judgement of the expected consequences of pollution accidents. It is fortunate that in most cases the expected damage were less than first estimated mainly due to the driving forces in the ecosystems to cope with the pollution. When accidental water pollution occurs its is important to make an expert judgement on the pollution potential, the likely affected water uses, and to make early warning for those who may use the affected water resource.

It is important the both the decision makers and the public are provided with validated data and sound information. When the pollution accident happens in international river basin, the downstream countries should warned in time to allow starting with the needed monitoring and to make the necessary arrangements for minimising the adverse effects.

As demonstrated with the fourth case study, dissemination of information and warning to the public should be carefully quality controlled. Misleading, invalid information can create confusion and panic in the public and should be avoided. Relevant, quality controlled data and information, the possible environmental and public health consequences of the pollution incident should be disseminated to the public as part of the public awareness elaborated in the contingency plan.

CONCLUSIONS

Pollution incidents may have local/national or regional/transboundary environmental consequences. It is important for each country or countries in international river basin to develop contingency plans for emergency situations created by pollution accidents. Although each pollution accident has its own characteristics, lessons could be learned and utilised for updating contingency plans, which among others, should include preparedness for appropriate emergency, short- and long-term monitoring and early warning communication systems.

The decision makers should have timely information on the characteristics of the spills, prediction of the concentration and distribution patterns of the pollutants allowing the preparation of the action plan for minimising and mitigating the adverse effects and rehabilitation of the environment. This could be achieved on the basis of appropriate monitoring results.

The monitoring approach in the case of pollution incidents should include:

- representative sample collection from the relevant matrix, i.e., water, suspended solids, bottom sediment, at affected sites and unaffected control sites,
- fit for the purpose analysis of the samples according to the information needs providing all necessary data/information on time and allowing: (a) comparison of pollution levels between affected and control sites, (b) assessment of spatial and temporal variation in the pollution levels in the water resources, and (c) comparison of the pollution levels resulted from the incident with natural background, baseline and pre-accident pollutant concentrations,
- co-ordinated quality assurance, analytical quality control measures in the laboratories, i.e., national and international, to ensure reliable, comparable monitoring results.

Public awareness, dissemination of the pollution incident related information and warning should be scientifically sound, avoiding political motivations and media manipulations.
REFERENCES


An accidental industrial spill of high cyanide concentration, originating - due to dike failure – from a storage pond of a mining company in Baia Mare (Romania) caused disastrous pollution on 30 January 2000. The total volume of the accidental spill was approximately 100 000 m³ containing around 100 tons of cyanide. The pollution plume entered the Tisza river which is the main tributary of the Danube river. The cyanide caused acute toxicity for different aquatic organisms (fish, phytoplankton, zooplankton, macrozoobenthos).

On the 10th of March 2000 another serious accident occurred in the upper Tisza region in Romania. Bursting the dike of a storage pond caused the discharge of 20 000 tons of ore slurry containing high concentration of lead, copper and zinc. The heavy metals were transported mainly in the suspended sediments along the Tisza river. In different sedimentation zones the settled polluted sediment increased the lead, copper and zinc concentration of the bottom sediment up to 1000 mg/kg order of magnitude.

The above noted accidents revealed the lack of information on the potential pollution sources in the Tisza river basin. A relevant inventory of the hazardous industrial activities and potential pollution sources is a basic requirement. The multicountry character of the basin press for the practical implementation of the guidelines on water quality monitoring and assessment of transboundary rivers.

INTRODUCTION

The Tisza river is the longest (977 km) tributary of the River Danube, having the largest drainage basin as well. It drains water from the eastern Carpathian Basin and discharges into the Danube. The drainage basin area is 157 200 km² amounting to 20 % of the total catchment area of the Danube. The Tisza basin is shared by Ukraine, Romania, Slovakia, Hungary and Yugoslavia.

Two serious accidental spills (a cyanide spill and a heavy metal spill) occurred in the upstream region of the Tisza river basin in January and March of the year 2000. Both accidents had the similarities that enclosure dams of industrial impoundments collapsed in extreme weather conditions.

EVALUATION OF THE CYANIDE POLLUTION

The Australian-Romanian company AURUL at Baia Mare (Romania) extracts gold from old mine tailings by cyanide process. The used cyanide solution has been recycled through a sedimentation impoundment. The dike of the storage pond breached on 30 January owing to heavy rainfall and rapid snow melting. At least 100 000 m³ impounded cyanide solution was spilled via the small river Lapus into the river Somes/Szamos which is the tributary of the Tisza river in Hungarian territory. Repair of the dam of the process water pond was completed on the 31st of January, thus eliminating the continuation of the spill.

On 31 January the Romanian authorities sent warning messages about the incident to Hungary. Upon the warning of the Romanian party, the Hungarian environmental and water management organisations made preparations for "receiving" the pollution wave. Third (the highest) level water pollution control action was officially ordered, which included frequent (two-hourly) sampling, in-situ and laboratory analyses, aimed firstly at the determination of the cyanide content of the water. A defence action plan was prepared on the basis of the analytical and discharge measurement data, aimed at restricting and minimising the impact of the pollution wave with appropriate governing of flows (closing sluices of outflows, filling the reservoir...
Kisköre – Lake-Tisza – with clean river water, etc.). Piped drinking water supply was terminated for the period of the passing of the pollution wave in the town of Szolnok.

The pollutant discharge consisted mostly of cyanide complexes bound to readily water soluble metals. In addition to analysing for the cyanide content of the water dissolved heavy metals (of complex forms) were also analysed. Measurement results indicated that copper was the dominant heavy metal in the river water, although zinc concentrations also exceeded the natural background levels and lead and silver were also detected. Regarding concentrations the maximum cyanide concentration was in the range of 20-30 mg/l in the Hungarian part of the River Szamos and of 10-15 mg/l in the Tisza river, downstream of the confluence with the River Szamos. Further downstream in the Tisza river the concentration became gradually lower upon the diluting effect of the tributary rivers and also due to the release of clean water from the reservoir Kisköre (Lake Tisza), which had been filled with clean water before the pollution wave arrived. The maximum cyanide concentration of the Tisza water, when leaving the country, was 1.49 mg/l. Figure 1 shows the maximum cyanide concentrations in the set of samples and the related times along the affected Hungarian rivers.

The typical concentration patterns of the pollutants cyanide and copper at the rkm 430 section of the Tisza river show that the pollution wave passed the cross section 24-26 hours, and the maximum concentrations were 5 mg/l cyanide and 3.5 mg/l copper in the set of samples (Figure 2).

The cyanide and the copper remained dissolved in the pollution plume, therefore the transport of the pollutants in the river was basically determined by the longitudinal dispersion and by the dilution of water discharge from the tributaries of the Tisza river. Figure 3 shows the change of the cyanide concentration profile along the Tisza river. The peak concentrations decreased, the length of the polluted plume increased during the transport, while the load of cyanide was rather stable (around 105-110 t) at different sections.

The cyanide caused acute toxicity for different aquatic organisms (fish, phytoplankton, zooplankton, macrozoobenthos). In the polluted section of the Tisza river the fish kill amounted to more than 1000 tons, around 30 % of the fish population.

On the basis of the results of biological microscope investigations it can be stated, that the cyanide pollution resulted in the devastation of the larger part of planktonic organisms of the rivers Tisza and Szamos. Reestablishment of the planktonic populations has started a few days after the passage of the pollution wave and planktonic communities characteristic to the season were formed afterwards.

Figure 1. Maximum concentrations of cyanide (mg/l) in the Hungarian reach of the Szamos and Tisza rivers
The cyanide pollution had considerable influence on the macroinvertebrate fauna of the rivers. The most sensitively reacting groups to the cyanide pollution are the **Crustacea**, where in case of some species (*Corophium curvispinum*) some 50-60% devastation; while in case of the larvae of **Chironimidae** some 50% of death were identified in the given sections. Besides these, the destruction of **Oligochaeta** taxa and among the aquatic insects that of the **Trichoptera** taxa was observed, although in a smaller scale. In the case of some taxa (*Ephemeroptera, Trichoptera, Chironimidae*) weakened physical condition were observed.

Since the cyanide pollution has passed the Hungarian part of the river Tisza, we have found live specimens of all macroinvertebrate taxa recorded previously in the given sections. These results so far show that some of the macroinvertebrate fauna of the river Szamos and Tisza has survived the cyanide pollution. Characteristic surviving species include some molluscs.
(Lithoglyphus naticoides), bivalves (Unio crassus, U. pictorum, U. tumidus), larvae of river Odonata species (Gomphus vulgatissimus, Ophiogomphus cecilia, Stylurus flavipes, Platythemis pennipes, Calopteryx splendens), Ephemeroptera larvae (Ametropus fragilis, Heptagenia-species), and the Palingenia longicauda.

On the basis of the investigations made in March it can be stated, that with the exception of Crustacea each group of the animals were present in considerable number of taxa in the springtime period, both in the River Szamos and the Tisza river. The presence of molluscs in the Tisza river, both upstream and downstream of the confluence of the River Szamos, might indicate that the cyanide pollution incident did not damage their population significantly. The species, which have suffered a substantial damage, devastation, during the cyanide pollution, are regenerating well along the entire Hungarian river reach, since significant populations of several taxa were identified in the stations investigated.

**EVALUATION OF THE HEAVY METAL POLLUTION INCIDENT**

The dam of the settling pond of a mining company of Baia Borsa, Romania, broke through on the 10th of March, 2000, upon the effect of heavy rainfall and simultaneous snowmelt. About 20 thousand tons of slurry, contaminated with heavy metals was discharged into the creek Vasér, then through the stream Viso in the Tisza river.

The pollution wave, carrying lead, copper and zinc, arrived to Hungary on the 11th of March through the Tiszabecs station of the Tisza river.

In the night of the 11/12 March the maximum concentrations of total lead and zinc was 2.9 mg/l, while that of the copper was 0.86 mg/l. On the 10th of March, before the arrival of the pollutant wave, the concentrations of the above named heavy metals were characteristically below 0.1 mg/l.

The first pollution wave passed the Tiszabecs station until the 13th of March, lasting for about one and half days. The quantity of lead transported through the border station Tiszabecs during this period can be estimated as 50 tonnes, while that of the copper and the zinc as 20 t and 70 t, respectively.

On the 15th of March the second pollution wave crossed the Tiszabecs station. This wave carried smaller concentrations and load and lasted shorter than the previous one.

As the pollution wave travelled downstream the heavy metal concentrations were gradually decreasing, due to the effect of settling, longitudinal dispersion and to the dilution water provided by the tributary streams. Downstream of the confluence with the River Szamos (Vásárosnamény) the concentrations of lead, copper and zinc did not exceed the limit values of water quality class "good".

The pollution wave reached the downstream border station Tiszasziget in eight days. At this station the concentration of lead and copper was below 0.1 mg/l, while the total zinc concentration varied in the range of 0.2-0.3 mg/l, slightly higher than the background concentrations.

The impact of the pollution wave on the composition of the sediment was detectable at some location in the upper Hungarian reach of the Tisza, in the zones of sedimentation.

Results of the analysis of sediment samples, made until the end of March, showed that heavy metals deposited from the pollution wave of high concentrations increased the lead concentration of the sediment to about 900 mg/kg at the stations Tiszabecs rkm 744 and Tiszaköröd rkm 728 (Figure 4). Copper and zinc concentrations of the sediment of the same stations were about 500 mg/kg and 1 400-1 500 mg/kg, respectively. These data represent about ten-fold concentration increase in comparison to background data (characteristic concentration ranges of the non-polluted bottom sediment of the Upper Tisza are 20-70 mg/kg for lead and copper and 100-400 mg/kg for zinc.

The above results are in harmony with the composition of the suspended sediment of the pollution wave: On the 12th of March the lead concentration of the suspended sediment was 1500-1800 mg/kg at Tiszabecs, while the respective values of copper and zinc were 900-1 100 mg/kg and 2 700-3 300 mg/kg. A further complementary information – supporting the
measurement results of Tiszabecs – is provided by the data of the silt samples taken from the Creek Vasér (which have received the heavy metal laden slurry discharge of the failed reservoir). In these samples the concentration of lead was 1 500 mg/kg, while that of the copper and zinc was 1 100 and 4 400 mg/kg, respectively.

Figure 4 Concentration of heavy metals (lead and copper) in the bottom sediment of the Tisza river in March 2000

Comparing the suspended sediment and bottom sediment contamination data to the respective soil quality limit values (Table 1) (Persaud et al (1992), MI-08-1735-1990 (1990), MI-10-420-83 (1983)) one can derive the following statements:

- The background (baseline) concentrations of lead, copper and zinc of the sediment of the Tisza river are lower than the "heavy impact level" of the sediment qualification system for these metals. They are also lower than the highest allowable level of soil qualification systems (considering the tolerance levels of plants);
- Lead, copper and zinc concentrations of the suspended sediment, as arrived with the pollution wave, were exceeding the limit values of the qualification systems of sediment, soil and sewage sludge;
- Contamination levels measured at Tiszabecs and Tiszakörőd, upon the impact of heavy metals, which were deposited from the pollution wave, were higher at some locations than the limit values of the soil- and sediment qualification systems.

Table 1. Limit values of lead, copper and zinc in sediment, soil and wastewater sludge

<table>
<thead>
<tr>
<th>Sludge, soil, sediment</th>
<th>Unit</th>
<th>Maximum allowed concentration</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wastewater sludge for agricultural disposal (MI-08-1735-1990 Hungarian Technical Guidelines)</td>
<td>mg/kg dry sludge</td>
<td>1000 1000 3000</td>
</tr>
<tr>
<td>Soil (criteria for plants) (MI-10-420-83 Hungarian Technical Guidelines)</td>
<td>mg/kg dry soil</td>
<td>100 100 300</td>
</tr>
<tr>
<td>Sediment of water courses (Persaud et.al. 1992)</td>
<td>mg/kg dry sediment</td>
<td>lowest effect level: 31 16 120 severe effect level: 250 110 820</td>
</tr>
</tbody>
</table>
CONSEQUENCES ON WATER QUALITY MONITORING AND ASSESSMENT

The above noted accidents revealed the lack of information on the potential pollution sources in the Tisza river basin. A relevant inventory of the hazardous industrial activities and potential pollution sources is a basic requirement. The multicountry character of the basin press for the practical implementation of the guidelines on water quality monitoring and assessment of transboundary rivers. The riparian countries should reconsider all stages of monitoring process including information needs, monitoring strategy, network design, laboratory and in situ analysis, information utilisation.

Specific recommendations:

- Up-to-date maintenance of an inventory of potential pollution sources
- Preparation of emergency plans for handling accidental pollution
- Strengthening the regional and international Accident Emergency Warning System
- Installation of automatic sampling stations to collect water samples for analysis
- Installation of automatic toxicity monitoring stations
- Development of emergency survey plan for ecological assessment of the environmental damages.

REFERENCES


INTEGRATED MONITORING AND ASSESSMENT OF SURFACE WATER AS A TOOL FOR INFORMED DECISION MAKING AND SUSTAINABILITY

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The developed monitoring and evaluation system for surface waters is based on 226 chemical, ecotoxicological and microbiological/virological parameters. An integrated evaluation scheme is applied consisting of two complimentary schemes: a) the traditional evaluation, based on the independent evaluation of parameters against existing legal limits which provides technical or legal oriented information, and b) the Amoeba yardstick which is based on the holistic evaluation of indices compared against preset target values. Amoeba provides management oriented information. To this effect interrelated parameters, are integrated and expressed by nine quality or effect indices: The Industrial Pollution, Pesticide Pollution, Nutrient Pollution, Organic Pollution, Bacteriological Index, Toxicity, Biodiversity, Benthic Saprobity and the Irrigation Quality Index. Each index is reflecting a different type of pollution or effect. All indices are integrated in one "yardstick", the Amoeba, by which the status of water resources, degree of deviation from targets and priorities for policy measures are clearly described. Target values for each index were developed based on available limits, ecotoxicological data, expert judgment. These were finally adjusted considering also socioeconomic factors. The main elements of the design and development of the system and its potentials to support effective monitoring for surface waters will be discussed.

INTRODUCTION

The ultimate goal of monitoring is to provide information for decision making. It has therefore to go beyond providing only data or fulfilling regulatory obligations. It has to allow for prediction rather than remaining retrospective (Environment Water Task Force DG XII, 1998). To this effect, it has to evaluate the existing situation, identify trends and emerging pollution at early stages of their development, and determine possible threats or effects in the ecosystem. From a study done in 1995 by European Laboratories (Villars, 1995) it was found that existing monitoring activities throughout Europe are associated with the following three major problems: 1. High cost / low efficiency and effectiveness, 2. Deficiencies in the information gained, (300-350 millions ECU are spend for monitoring water quality in EU, but still the information gained is not sufficient), and 3. Lack of harmonization of monitoring throughout Europe. Because of these problems and despite the improvement of EU Regulations and control measures, European Water sources continue to show signs of deterioration and the evidence of threats to human health and the ecosystem is increasing continuously (Water Task Force DG XII, Working document June 1997). One of the main reasons of the deficiency in information and the low effectiveness is the very limited implementation of biological monitoring and the known shortcomings of the physicochemical monitoring. Many authors e.g. Hendriks et al. (1994, Verhaar et al.(1994), Leeuwen (1994) pointed out the need for complementary chemical and biological monitoring approaches, which can provide a more holistic evaluation of water quality and more concrete information for risk assessment and risk management.

Within an EU project LIFE 95/CY/B2/CT/868MED (Michaelidou, 2000(a)), an Integrated Monitoring and Early Warning System was developed in Cyprus in order to sustain quality and multifunctionality of surface waters, for man and the ecosystem The project also aimed to increase the cost effectiveness of the existing monitoring and improve the integration of the results into the decision making process. Cyprus is an island situated in the northeastern corner of the Mediterranean basin at the crossroads of Europe, Asia, and Africa and it has always suffered from droughts often lasting up to several years. Due to climatic conditions, rivers flow mainly from December to April. In order to prevent loses to the sea and maximize storage, 101 dams have been built on the island. The project covered the 8 major dams, their contributing rivers and one stream from a "virgin" area, representing 84% of the total capacity of the 101 dams. Main focus of the project was on the development of its early warning and trend detection capabilities. Its ultimate goals were the holistic investigation, early identification...
of emerging pollution and priorities for prevention, and provision of adequate information for decision-making.

Integration and multidisciplinary approach form the landmark of the project. The two pillars on which the project was built were the holistic investigation approach and the integrated evaluation.

The holistic investigation approach has enhanced capabilities for trend recognition and early identification of emerging problems. It comprises of chemical specific methods, aggregate variables, toxicity tests and bioassessments. By integrating toxicity testing with state-of-the art analytical chemistry and microbiology, possible effects and interactions of the "cocktail" of water pollutants on Man and the Ecosystem, can be addressed. It is based on 226 microbiological, chemical and ecotoxicological parameters.

The integrated evaluation scheme is capable to provide not only data for legal compliance, but also information geared to address needs for management, forecasting and prevention. It comprises two complimentary schemes: the conventional evaluation and the Amoeba Yardstick (Triverdi, et al, 1993, & De Zwart, et al 1995)

METHODS

The holistic investigation

The investigation scheme covered both water samples and sediments. In the investigation 226 parameters were included. These parameters shown below in Table 1, include chemical specific and group chemical parameters, toxicological ones (for acute toxicity and genotoxicity), microbiological, virological and biomonitoring parameters. A seasonal sampling was applied for water whilst sediments were sampled only once. In total 214 water samples and 43 sediments were collected. Each water sample was divided to 10 different sub-analytical samples in order to address the 226 chemical, microbiological, toxicity testing and biomonitoring parameters. In total 2140 water sub-samples were collected. 450 additional samples were analyzed for QC/QA. A tier analytical approach was developed to cope cost-effectively with the extended investigation scheme. Standard methods from the EPA and international literature were adopted, along with methods developed, validated and registered as State General Laboratory standard operating procedures (SGL/SOPs) according to the international Quality Assurance (Q.A.) criteria and the in-house Quality Assurance Program. In addition, a strict Quality Control Scheme was implemented. According to our SOPs, app. 20-30% additional samples were analyzed for Q.C.

Table 1: Monitoring parameters in water and sediments

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<tr>
<th>Water</th>
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<th>Chemical Analysis</th>
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<td>Toxicity Tests</td>
<td>Microtox</td>
<td>pH, DO, BOD₅, COD, Turbidity</td>
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<td>Total Kjeldahl Nitrogen, Ammonium</td>
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<td>Mutatox after S9</td>
<td>Total. Phosphorous</td>
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<td>Nitrates &amp; Nitrites</td>
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<td>Sodium &amp; metals</td>
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<td>general FID profile</td>
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<td>Faecal Streptococci</td>
<td>PCBS</td>
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<td>Bacteriophages</td>
<td>Pesticides</td>
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<td></td>
<td>Intestinal Parasite Eggs</td>
<td>◊ OCL</td>
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<tr>
<td></td>
<td>Enteroviruses</td>
<td>◊ OPS</td>
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<td>Sediments</td>
<td>Mechanical analysis</td>
<td>◊ riazines</td>
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<td>Acute Toxicity (Microtox solid phase)</td>
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<td>General chromatographic profile</td>
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<td>OCL and PCBs</td>
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The integrated evaluation

The evaluation comprises two complimentary schemes: the conventional evaluation and the Amoeba Yardstick: a) Under the conventional evaluation, the 226 parameters were analyzed and evaluated independently from each other for compliance with existing legal limits. This evaluation scheme provides both legal and technical oriented information. b) The Amoeba Yardstick aims to aggregate hundreds of data into concrete information and provide clear identification of problems and priorities for action to the decision-makers. To this effect all parameters, which are related to the same pollution cause or effect, are included in one index. The 226 parameters are integrated to nine indices. On Table No 2 the parameters included in each Index are presented.

The Index calculation (De Zwart personal communication 2000): The measured analytical results of different variables can only be combined into a single index value when they are brought to the same unity. This is accomplished by transforming the analytical results to a 0 to 100 % acceptability scale on the bases of a "parameter acceptability curve". For observed effects and non-toxic compounds the transformation have been guided by scientifically or empirically justifiable common sense in combination with a thorough appraisal of existing standards. The concentrations of toxicants have been scaled to acceptability by using the concept of Species Sensitivity Distributions (SSD) (Suter and Posthuma, in press)., the SSD approach allows to express environmental risk as the percentage of species exposed to a concentration exceeding their chronic No Observed Effect Concentration (NOEC), the so-called Potentially Affected Fraction of species or PAF. The SSDs for individual compounds are constructed by fitting a cumulative sensitivity distribution (log-logistic) to a wide variety of toxicity data obtained from world literature (De Zwart, in press). The indices pertaining to toxic pollution (PPI and IPI) are calculated from the acceptability scores of the individual chemicals by rule of effect addition:

$$100\left(\frac{\text{acceptability}_1}{100}\right) \times \left(\frac{\text{acceptability}_2}{100}\right) \times \ldots \times \left(\frac{\text{acceptability}_n}{100}\right)$$

Effect addition is modelling the assumption that there is no correlation between the responses of the collection of species towards different chemicals.

All other index values are calculated by geometrically averaging the acceptability scores of the individual variables belonging to that index. According to Bach (1980), the geometrical average is applied as a measure to produce a higher impact to variables indicating adverse conditions.

Individual indices are expressed on a scale from 0 to 100, where 0 indicates the worst imaginable environmental condition, and 100 stands for a totally natural environment which is entirely not influenced by mankind.

The Amoeba is based on the holistic evaluation of these indices against target values. As a first step the target value for each index is developed based on all available limits of parameters included in the index. When limits do not exist, eco/toxicological data from the international literature are considered. This is essential because limits exist for only few parameters over the hundreds that are present in the aquatic environment and which may pose toxic threats to the Ecosystem. This preliminary setting is based only on scientific criteria. Based on these, policy makers can decide on the final target values after considering socio-economic factors, needs for sustainable development and existing water quality and its potential use. These policy oriented target values are expressed as % of the target developed on purely scientifically basis. Eventually this % can be adjusted accordingly. After 4 years monitoring data, the target values that were set at the beginning of the project (preliminary ones), were eventually adjusted to higher scores approaching the scientifically derived targets. Both the preliminary and final target values are presented in Table 2.

The Amoeba is presented on a radar plot where each index is assigned to a specific sector (figure 1). The magnitude of both individual index and target values is expressed in a variable radius. The index value is plotted "white" on top of the target "gray" circle. Each target value is the desirable value of the respective Index at which maximum protection or sustainability of the Ecosystem is achieved. As the index value increases and approaches the set target, the water quality improves and vice versa. When the target circle is still visible, this indicates environmental conditions negatively deviating from target conditions. If no "dark gray" is visible on the plate, the environmental conditions are equal to or better than the set targets and no further action is necessary.
The figure No 1 is an example of the Amoeba presentation of one dam, Polemidia dam and its inflowing Garyllis river. The changes in the quality of water between the river and the dam are very clearly indicated both in terms of the indices changed and the degree of their respective changes. The biodiversity index BDI and BSI were not determined for the river because Garyllis, as most of the "rivers" in Cyprus, is a stream flowing only during winter and beginning of spring, remaining almost dried for the rest of the year.

This sector wise presentation and the direct link to causes, provide management orientated information and facilitates the integration of monitoring results into the policymaking. Based on this information, proper action for sanitation, prevention and control can be generated.

**Figure 1: Amoeba presentation and changes of the water quality between Polemidia dam and the inflowing Garyllis river**

### Table 2: Indices, Parameters and Target values

<table>
<thead>
<tr>
<th>Index</th>
<th>Parameters per Index</th>
<th>Target Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>OPI: Organic Pollution</td>
<td>BOD₅, COD</td>
<td>80</td>
</tr>
<tr>
<td>NPI: Nutrient Pollution</td>
<td>NH₃, Chlorophyll-a, NO₃, total -P, Turbidity, Total N</td>
<td>70</td>
</tr>
<tr>
<td>PPI: Pesticide Pollution</td>
<td>70 individual Pesticides integrated into 23 groups &amp; Total Pesticides</td>
<td>90</td>
</tr>
<tr>
<td>BQI: Bacteriological Pollution</td>
<td>Bacteriophages, Salmonella Enteroviruses, Faecal coliforms, Faecal streptococci</td>
<td>90</td>
</tr>
<tr>
<td>IPI: Industrial Pollution</td>
<td>Metals (8), NO₂, VOCs (56), Phthalates (6), PCBs (17), GC profiles (24), PAHs (15)</td>
<td>90</td>
</tr>
<tr>
<td>IQI: Irrigation Quality</td>
<td>Conductivity, pH, Sodium Adsorption Ratio (SAR)</td>
<td>70</td>
</tr>
<tr>
<td>BSI: Benthic Saprobity</td>
<td>BMWP-average</td>
<td>70</td>
</tr>
<tr>
<td>BDI: Benthic Diversity</td>
<td>Diversity, SDS</td>
<td>70</td>
</tr>
<tr>
<td>TOX: Toxicity</td>
<td>PAF = % of potentially affected species and includes tests on Algae, Daphnia and Photobacterium phosphoreum, Genotoxicity: Mutatox direct &amp; Mutatox after S9 activation</td>
<td>90</td>
</tr>
</tbody>
</table>
MAIN RESULTS & EVALUATION

The results were evaluated according to the integrated evaluation scheme, based on the 226 parameters. According to the conventional scheme the results were grouped into three main categories i.e. microbiological, chemical, toxicological, and evaluated independently from each other. All the results and the conventional evaluation are given in the nineteen Data base reports (one for each waterbody) (Michaelidou et. al 2000 (b)). The results from all three categories were then evaluated holistically under the AMOEBA yardstick.

The main conclusions drawn from the integrated evaluation of all results from the 4 years of monitoring (Michaelidou, 2000) are summarized as follows:

- The degree of pollution in dams and rivers of Cyprus ranged within acceptable levels and in most of the cases compliance with existing limits and the EU Directive 75/440 was found. Higher pollution was found in Polemidia Dam and is contributing river Garyllis that were selected as the polluted references,

- The impact of agricultural pollution on water quality was low and eutrophication was mainly experienced in Polemidia dam. With the exception of this dam and its contributing river Garyllis total Pesticide concentration was always less than 0,5 ug/L. Triazines were the predominant pesticides found in most of the samples. Nitrates concentration was ranging from 0,5 to 11 mg/L in the 7 dams and from 0,5 to 31 mg/L in the 9 rivers. However an increasing trend in the concentration of Nitrates was shown from 1996 to 1999,

- The Bacteriological quality complied with the EU Directive 75/440. No enteroviruses were detected,

- Deviations of industrial pollutants and other organic pollutants from limits occurred occasionally. Maximum values were measured from the end of 1997 up to the end of 1998,

- In most of the cases Toxicity was measured and respective criteria for toxicity and for the ecological quality were not met,

- In the sediments high acute toxicity was measured and many pesticides and industrial pollutants which were non detectable in water were also found. Endocrine disrupting and potential genotoxic compounds were identified in the sediments, indicating the need for continuous future monitoring of those compounds in water. It is evident from the 4 years of monitoring that future activities should pay a special attention to the assessment of the presence, origin and risks associated with genotoxic and endocrine disruptive compounds.

CONCLUDING REMARKS

The results and the experience gained from this project clearly show that the application of a battery of complementary approaches is an essential prerequisite for cost effective monitoring, early identification of emerging pollution, and optimization of monitoring schemes. For example Toxicity testing signaled the presence of potential genotoxic compounds whilst these were not detected chemically, either due to low concentration i.e. below detection limits, or limitations of the implemented investigation scheme. Furthermore, the application of the nine indices and the Amoeba Yardstick facilitated the integration of monitoring results into the decision making process as hundreds of data were transformed into concrete information on water quality, changes, trends, deviation from targets and finally on main priorities for action.

ACKNOWLEDGEMENTS

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GUIDELINES ON MONITORING AND ASSESSMENT OF TRANSBOUNDARY GROUNDWATERS

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During the 2nd Meeting of the Parties to the Convention on the Protection and Use of Transboundary Watercourses and International Lakes1, which was held in March 2000, Guidelines on Monitoring and Assessment of Transboundary Groundwaters were presented for endorsement. For the drafting of these guidelines, which was one of the activities of programme area 3: "Integrated management of water and related ecosystems", a core group was established under the responsibility of the ECE Task Force on Monitoring and Assessment.

The groundwater guidelines are supported by supplementary documentation. An inventory, based largely on questionnaire responses, was made of transboundary groundwaters in the ECE region (figure 1) and their current monitoring and assessment practices. Background reports were produced on the inventory and on the present approaches to monitoring which contributed to the drafting of the guidelines. In preparation for and support to the drafting of the guidelines, additional activities included work to identify indicators for groundwater assessment, a review of the use of models and a study to the state of the art on monitoring and assessment of groundwaters. In this way, four supporting technical reports have been produced.

Co-operation has been sought with various international organisations and institutions to make best use of existing programmes and link ongoing activities in the field of monitoring and assessment. In the coming years it is hoped that the guidelines will be implemented and tested through a series of pilot projects. After a period of some three to five years, and taking account of experience gained in the pilot projects, review and possible revision of the guidelines will be carried out.

This paper gives a brief overview of the groundwater guidelines. Some highlights and recommendations are presented. These guidelines form part of a series of such guidelines on monitoring and assessment of rivers, groundwaters, lakes and estuaries, which follow the general approach of the monitoring cycle developed for the river guidelines (UN/ECE, 1996).

PROBLEM DEFINITION

Groundwater supports various important functions some of which, like nature and agriculture, are directly related to the occurrence of groundwater. For other functions, such as drinking water and industrial water supply, groundwater is used as a production factor, because of its normally good and constant quality. However, high and increasing population density, continuously growing industrialisation and intensive agriculture will have a negative effect on
the quality of soil and groundwater. Over-abstraction of groundwater results in declining groundwater tables. In recent times, soils have become increasingly polluted by private and public waste dumps, air pollution, agricultural chemicals (fertilisers, pesticides, herbicides). In shallow groundwaters this pollution can easily be transported to locations where it may be harmful to one or more of the functions referred to above. These problems do not occur only within countries, but can also have transboundary impacts. Then there is a requirement for agreed and consistent cross-border monitoring and assessment activities to identify the origin of the pollution and evaluate its current extent and future evolution. Furthermore, measures may be needed to avoid these undesired developments and monitoring to assess their effectiveness, within, as well as between countries through joint groundwater bodies. The integral basin area approach, or ecosystem approach, which was adopted as a guiding principle in the Convention on the Protection and Use of Transboundary Watercourses and International Lakes, was also the basis for structuring the guidelines on monitoring and assessment of transboundary groundwaters.

**OBJECTIVES AND CHARACTER OF THE GUIDELINES**

The objectives of the groundwater guidelines is to provide a framework for the problem identification, the specification of information needs and the use of applicable tools and the setting up of a monitoring and assessment systems for transboundary groundwaters. The guidelines will focus on the implementation of the relevant provisions of the Convention. The core elements of the Convention concern the protection of groundwater resources against pollution and over-use, the protection of the ecosystem, which is closely connected with shallow groundwaters, and the protection of sources of drinking water supply.

The character of these guidelines is strategic rather than (operational) technical. For technical details, the background reports prepared by the Core Group Groundwater, and international literature and handbooks on operational practices of monitoring and assessment should be consulted. The guidelines are not legally binding. They are intended to assist ECE governments and joint bodies in developing harmonised rules for the setting up and operation of systems for transboundary groundwater monitoring and assessment. The target group comprises decision makers and planners in ministries, organisations and institutions responsible for environmental,
water or hydrogeological issues and all those who are responsible for managing transboundary groundwaters. The guidelines also aim to provide advice to those who are responsible for and those who are involved in the development of sustainable water management schemes. These guidelines are intended to be concise and realistic; they are not intended to be prescriptive. They provide an approach for the identification of problems and a guidance to meet the information needs. The guidelines deal mostly with monitoring and assessment needs that arise from the Convention and as far as possible from the Protocol on Water and Health to the same Convention.

STRUCTURE OF THE GUIDELINES

In these groundwater guidelines the general approach of the monitoring cycle (figure 2) has been followed. The monitoring cycle can be seen as a sequence of related activities which starts with the definition of information needs and ends with the use of the information product. It offers a readers’ guide and a valuable approach when drawing up programmes for the monitoring and assessment of transboundary groundwaters. An exchange of information (and joint assessment/modelling) between riparian parties is meaningful only when the data are comparable, which can be achieved when all components of groundwater monitoring activities on both sides of the border use similar principles or adopt an approach such as the monitoring cycle.

![Figure 2 Monitoring cycle](image)

SOME HIGHLIGHTS

In this paper some of the highlights of the groundwater guidelines will be mentioned and briefly explained.

- Specific aspects of groundwater monitoring
  When implementing transboundary monitoring and assessment programmes, it is essential to present the hydrogeology in conceptual models and/or in graphic schemes. This should comprise a characterisation of the transboundary aquifer (geometry), the flow conditions, including recharge and discharge areas, and the evolution of the groundwater quality.
The characterisation and description of relevant transboundary aquifer systems are a prerequisite for the monitoring and assessment of transboundary waters in general but of transboundary groundwaters in particular. Features that influence the way groundwaters are monitored and assessed and that distinguish them from surface waters are:

- the slow movement (residence times);
- the interaction between the aquifer material;
- the variability of groundwater flow (intergranular or through fractures);
- the presence of recharge and discharge areas.

Knowledge of the groundwater flow system means in particular the locations of groundwater recharge and discharge zones and the way groundwater flows through aquifers from zone to zone. Figure 3 illustrates that activities in the recharge areas on one side of the border can adversely affect the quality and quantity of groundwater on the other side of the border.

![Figure 3 Transboundary groundwater flow systems](Image)

To characterise groundwater occurrence, information on geology, geophysics and hydrogeology in the transboundary area is needed. It is also very important that the dynamics of the groundwater flow system, such as seasonal or longerterm responses and variations and changes in flow rate or direction caused by human activities, particularly groundwater abstractions, are understood and agreed by the parties at both sides of the border.

**Integrated approach**

Groundwater should be assessed in an integrated manner, based on criteria that cover water quantity and water quality for different human uses as well as requirements of ecosystems. The harmonisation of surface water and groundwater monitoring networks must be envisaged, in order to manage and to protect transboundary water resources effectively. In developing monitoring programmes, especially in a transboundary context, the integration of data gathering and storage, the interaction of groundwater and surface water and the relations between groundwater quantity and groundwater quality are important aspects to consider for further integration.

Core elements of groundwater management are functions and uses of the groundwater bodies (aquifers), issues (problems) and pressures (threats) and the impact of measures on the overall functioning of the water body.
• Information needs
Information needs deals with the questions why and how information is needed. Therefore different concepts like the policy life-cycle and the DPSIR-framework (driving force, pressure, state, impact, response) (see figure 4) are recommended as approaches to define information needs. Proper identification of information needs requires that the concerns and decision-making processes of information users are defined in advance. In these guidelines a special attention has been given to the role of indicators, because they can play an important role to communicate monitoring results with decision makers since indicators are closely linked to relevant recommendations for management action.

Figure 4  DPSIR framework

• Data management
Monitoring data collected by riparian countries in transboundary aquifers should be comparable, available for integration with information from variety of sources and easily aggregated spatially and temporally.
Data produced by groundwater monitoring programmes should be validated, stored and made accessible. The goal of data management is to convert data into information that meets the specified information needs and the associated objectives of the monitoring programme.
For modelling purposes of transboundary aquifers, the standardisation of the accessibility of data (interfaces to databases and (GIS) is more important than the standardisation of the software used. If both the conceptual model and the basic data are reliable, the results will be comparable even when the software used is not the same.

• Joint or co-ordinated action and institutional arrangements
The successful drawing-up and implementation of policies, strategies and methodologies on groundwater management crucially depends on institutional aspects. These include the organisation, structures, arrangements for co-operation and the responsibilities of institutions and organisations involved. In transboundary groundwater management, international co-operation is governed by the provisions of the Convention, which stipulates that the socio-economic conditions in the riparian countries should be taken into account when deciding on the specific institutional arrangements.
IMPLEMENTATION OF THE GROUNDWATER GUIDELINES

After a period of some years the groundwater guidelines will reviewed on the bases of experiences gained with the implementation in pilot projects in some transboundary aquifers in the ECE region. The objectives of the pilot projects are to demonstrate the application and to illustrate from experiences the process and the difficulties of the implementation of the guidelines, to assist countries in the implementation and to identify gaps and incompleteness and consequently to propose improvements.

Recently, the EU Water Framework Directive came into force. It can be expected that these groundwater guidelines can play an important role in the implementation of this Water Framework Directive.

Another development is the recent co-operation with UNESCO, IAH and FAO in the TARM (Transboundary Aquifer Resource Management) project. For this project UNESCO, IAH and the FAO adopted the ECE groundwater guidelines.

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INDICATORS AND INTEGRATED ENVIRONMENTAL ASSESSMENT FOR REGIONAL SEAS OF EUROPE; PERSPECTIVES FOR A MULTI TRACK APPROACH

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The European Topic Centre on Marine and Coastal Environment (ETC/MCE) of the European Environment Agency (EEA) and the European Marine Regional Conventions/ Action plans collaborate in data/information exchange, the development of indicators and in the harmonisation of reporting. The chain monitoring-data-information-assessment-reporting has been gone through in the preparation of Integrated Environmental Assessments of Regional Seas in Europe. The effectiveness of the different steps of the chain is apparent from the reporting. In this respect lessons can be learned from the reports ‘Environment at the turn of the Century’ of the European Environment Agency (EEA), the ‘Quality Status Report 2000’ of OSPAR and ‘State and pressures of the Marine and Coastal Mediterranean environment’ of the EEA and UNEP/MAP. The common accepted analytical framework of Pressure State Response (OECD, Indicators for sustainable development) or the Driving Forces- Pressure-State-Impact-Response of the EEA are successfully used for the assessment of major environmental themes related to pollution. Clear operational objectives for the abatement of pollution, contributed to development and improvement of the tools. This paper presents the status and potentials of Indicators and Integrated Environmental Assessments for the Coastal and Marine environment. The indicator focus is on the conceptual and technical aspects. The view on Integrated Environmental Assessment addresses the institutional and technical (ICT) arrangements to improve on the steps of the monitoring to reporting chain.

INTRODUCTION

Policies to improve marine waters in Europe started in the seventies and eighties at the regional seas level. Five Conventions and action programmes, OSPAR (North East Atlantic and North Sea), HELCOM (Baltic Sea), AMAP (Arctic waters), Barcelona Convention/ UNEP/MAP (Mediterranean Sea), Bucharest Convention/ UNEP/BSEP (Black Sea) now cover European regional seas. The European Environment Agency (EEA) was established in the nineties by the European Union to develop European environmental information. Collaboration of the EEA with the Conventions/action programmes is ongoing. In the 21st century, European water management policies will impact the monitoring and assessment activities within the regional seas conventions and at the national level. Assessment and monitoring of the quality of the marine environment are a key to achieving the aims of the Regional Marine Conventions and Action Programmes in Europe. Their activities have roots in the abatement of pollution and extended to major policy issues like biodiversity, inter alia, to implement global agreements of the 1992 Rio Earth summit with Agenda 21. The European Union is contracting party of the conventions. The EU has recognised the importance of good ecological quality of the coastal zones and marine waters and the need for measures to ensure environmental and economic sustainability. The challenge of managing coastal zones lies in its strategic importance: a home to a majority of citizens an increasing percentage of our economic activities. Its resources however are subject to growing pressures, and urgent European level action is identified. A comprehensive management approach is needed with the development and exchange of information and knowledge as a prerequisite for action.

The European Environment Agency and its Topic Centre Marine and Coastal Environment provide the European commission and the public with information on the current state and future trends of regional Seas of Europe. To enhance the monitoring to reporting chain within the regional seas conventions/action programmes, institutional arrangements for collaboration now exists between EEA and international institutions (see Box 1). Legal instruments such as the recent adopted EU Water Framework Directive mean also a concrete step forwards to harmonisation of monitoring and reporting on landbased pressures and on the state of the coastal waters at the European level.
Box 1 Institutional arrangements supporting marine information supply on the European level

The Inter-Regional Forum (IRF) was set up in 1995 by the European Environmental Agency (EEA) with the main objectives to facilitate the exchange and the possible integration of existing data and information produced by the European Marine Regional Conventions/Actions Plans with the EEA and the European Topic Centre on Marine and Coastal Environment (ETC/MCE) of the EEA. The following marine conventions/ action programmes take part (figure 1): the 1992 OSPAR Convention for the Protection of the Marine Environment in the North East Atlantic, covering the North East Atlantic, the North Sea, the Norwegian Sea and parts of the Barents Sea, the Helsinki Commission (HELCOM), covering the Baltic Sea, the Arctic Monitoring and Assessment Programme (AMAP), the Mediterranean Action Plan (MAP), and the UNEP Black Sea Environment Program (BSEP). Memoranda of understanding between the conventions/programmes and EEA are put in place with OSPAR and HELCOM and are in preparation for MAP and ICES. To meet the aims to improve working relations and task sharing, according to EEA’s mandate of providing reliable, harmonised and objective information on the state of the European environment, working groups on Indicators, Data management and GIS were established at the end of 1999.
EUROPEAN WATER MANAGEMENT AND INFORMATION FOR ASSESSMENTS

The EEA mission is to deliver timely, targeted, relevant and reliable information to policy-makers and the public for the development and implementation of sound environmental polices in the European Union and the other EEA member countries. The aspects timely, targeted, relevant and reliable are a real challenge for the EEA assessments for the marine waters. The reporting is based on data from international databases, published reports and scientific literature. It is aimed not to duplicate work already done by other international working groups.

Main environment policy questions for the EU in general concern the improvement on the state of the environment, the integration of environmental concern in sector policies and sustainable development for sectors and for the environment as a whole. The relevance of information for the public is stressed. Better access to information and citizens’ participation in the political process will promote sound environmental policies.

Although in general the pressures on the coastal and marine waters are generic, the sensitivity of the marine environment and coastal zones differ. The recent integrated reports on pressures, state and impacts of regional seas, identify differences and similarities in the major challenges and problems. Different priorities exist in the information need at the regional seas level which are also valid on the European policy level. These differences have to be taken into account in the efforts of the EEA and the Conventions/action programmes in respect of the harmonisation in monitoring, in the development of indicators, and in the production of assessment reports.

To assess the main challenges and problems for the marine and coastal waters, the Inter-Regional Forum Working Group on Indicators has framed the information needs following the general conceptual framework of the EEA, known as the DPSIR approach. DPSIR stands for Driving Forces, Pressures, States, Impacts and Responses (box 2).

Box 2 DPSIR approach for integrated environmental assessments

| Driving forces | describe the developments in human activities and economic sectors playing a key role in driving environmental change. If the contribution of the different driving forces to an environmental problem can be shown, more understanding of the coastal system and presumably, more adequate measures, the responses are the result. |
| Pressures | describe direct stresses on the environment such as emissions to water, total input of substances to the coastal zone. States describes environmental (geo-physical, chemical and biological) variables which characterise the conditions of coastal zones. Impacts describe the changes in ecosystems, resources and human health. Due to the resilience of the ecosystem, changes in the environmental pressures do not always result in changes within the ecosystem. Moreover, changes in the state of the environment are so gradual that changes in the system are difficult to identify and often there is a time lag before changes become visible. The responses of policies can be defined specific in terms of measures affecting driving forces, pressures, state and impact or more generic like the adoption of integrated coastal zone management or the ecosystem approach in fisheries. |

Indicators that span the whole DPSIR categories provide an insight into the interactions between environmental policy making, developments in major sectors and the state of the environment.

The DPSIR framework for issues relevant for the marine and coastal environment covering the major themes within the marine conventions/action programmes and the EU is presented in table 2. The framework describes the information demand on the European level. The grey boxes show the fields of the present development of indicators. Indicators for the input of nutrients and hazardous substances and the concentrations of nutrients and contaminants in organisms are now being tested by the ETC/MCE. Indicators for pressures have been developed by EUROSTAT independently. OSPAR is identifying indicators for biological state variables to be applicable for the development of ecological quality objectives and HELCOM started with indicators for eutrophication in the Baltic Sea.
The identification and development of indicators for Driving forces within the EEA is done more in general and is related to the sectors dominating the Gross Domestic Product. Although there is in some European countries a trend towards policy monitoring (performance indicators), the identification of possible response indicators for the marine themes needs more attention. At present clear formulated policies like the reduction aims in input of hazardous substances to the sea offer the best premises for accountability.

### INDICATORS FOR THE MARINE AND COASTAL ENVIRONMENT ON THE EUROPEAN LEVEL

DPSIR indicators for monitoring changes to evaluating the effectiveness of policies are building blocks for environmental assessment reports and serve public communication. The purpose and level of indicators determine the amount of detail in which they are described. Experiences in the EEA, the Topic Centres and the UK Ministry of the Environment make clear that indicators relate to policy-objectives. The approach thus far is visualised by EEA using a pyramid or objective-criteria hierarchy (Figure 2). In such hierarchy:

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**Table 2** Major themes and indicators within the DPSIR framework for marine waters and coastal zones. The themes radioactive substances, oil pollution, micro-biological pollution, waste dumping and the introduction of non-indigenous species are also identified within the DPSIR framework covered.
• the top layer contains Headline-indicators that are used for European policy,
• the middle layer represent the elaboration of these at system-level or national level, focusing on translation of European policy to these levels.
• the bottom layer represents local management and the operationalisation to monitoring activities, availability and organisation of data-sets etc.

Figure 2 Pyramid of indicators. The scheme is an ideal structure. It considers aggregation of both qualitative and quantitative information and can be applied in the context of the production of yearly based indicator reporting and integrated assessment reporting.

At present Headline-Indicators are being developed by a taskforce, comprising representatives of the Commission, the EEA, the Joint Research Centre (JRC) and voluntary contributions from Member States. Topic centres of the EEA concentrate on the indicator systems of the middle layer for the different environmental themes.

The EEA report ‘Environmental signals 2000‘(2000) is the first regular indicator report showing the results of these exercises for the major environmental themes.

The topdown approach and bottom up approach for indicator development for the Marine and Coastal Environment need to come together. For the marine and coastal environment Ecosystem Health could be developed as a headline indicator.

Indicators are especially important tools for accountability and transparency. This requires that they are limited in number, relevant, responsive, simple and policy-related. Further it is required that specific databases on indicators need to be developed through sustained data management.

The Topic Centre Marine and Coastal Environment studied the availability and access to data for the development of pressure and state indicators for the coastal waters at the European level. Data were gathered and parameters for the themes eutrophication and hazardous substances, were tested on its potential for indicators. As a pilot a database was made to investigate its potential for EEA reporting.

The resulting database showed clearly the gaps in easy available marine environmental information on the European level. At present the list of potential indicators, which comply to the criterion of European comparability and easy data processing on a regular basis, is restricted to a selected set of parameters available in the international databases.

The theme eutrophication is best covered through data on nutrients concentrations (state indicators, Figure 3) and information on direct and riverine input of nutrients (pressure indicators). For the theme hazardous substances data on direct and riverine input of heavy metals (cadmium, mercury, lead and zinc) and organic micropollutants (lindane and sum PCB7) are used as indicators (Figure 4). Furthermore the concentrations of these substances in organisms are assessed.

From assessment of the international databases of OSPAR on contaminants in biota it became clear that despite the international monitoring programmes and available international databases, the assessment of these data is not easy expeditable. Because of further quality assurance and the application of non standardised statistical methods, trend analysis is not a routine. The OSPAR QSR 2000 also reflects in its overall assessment the imperfections in the nature of the data of contaminants in biota and regrets that the assessment of so many time series in biota has failed to derive statistical trends.
Improving existing monitoring and assessment tools is ongoing in OSPAR. Some international marine data bases like that of contaminants in sediment seem yet not suitable for use in indicator assessments which follow the present working procedure of EEA for Environmental Signals. Although this database is built on the JAMP (OSPAR Joint Assessment and Monitoring Programme) different analytical procedures are allowed and this restrict its use for holistic assessments. Sediment Guidelines on normalisation were recently revised, a prerequisite for the further use of monitoring data in indicator presentations. From this assessment it became clear that, considering the marine data, the gathering of data and information on the European level, should preferably be organised in collaboration with the Conventions/action programmes. Not only for reasons of efficiency in data collection on the European level, but also to guarantee proper quality assurance on the data (sampling and analysis) and sustained data management. Additional data which need to be collected directly from countries, require quality assurance procedures and adequate meta data descriptions before use in indicator assessments.

Figure 3 Changes in Phosphate concentrations in OSPAR and HELCOM coastal waters 1985-1998 (EEA, 2000b). Data were aggregated in squares of 100km$^2$. The total number of squares in each area is given in brackets. Source: ICES, Finish National Focal Point

Figure 4 Change in sum of direct and riverine inputs of hazardous substances into the North East Atlantic and North Sea in the period 1990-1998, based on OSPAR data (EEA 2000c). Sum of input of each substance in 1990 is 100%
The structure of present international databases needs to be adapted in order to meet the requirements for quick scanning on indicators by international data experts. National reportings from the member states of the Conventions should be further harmonised with the needs of yearly reportings of the EEA. Enforcement of the data management of international databases is a prerequisite for the production of timely information. This might result in the development of indicator data bases within each convention. The national implementation of the strategies of the conventions and action programmes are a prerequisite for information on the level of regional seas and contribute to the overall picture of all regional seas at the European level.

LESSONS LEARNT FROM MARINE INTEGRATED ASSESSMENT REPORTING BY EEA AND OSPAR

Several integrated reports on the impacts of the human activities and the effectiveness of measures show the state of the art of the monitoring to reporting chain. In 1998 the EEA presented the report ‘Europe’s Environment: The Second Assessment’. Main subjects identified as themes for concern in the marine and coastal environment chapter are: eutrophication, contamination, over-fishing and degradation of coastal zones. The data retrieval on eutrophication and chemical substances in the report was based on a selected set of data from the Conventions. A graphical representation and classification of yearly figures was given for concentrations and (direct discharges and riverine) loads of nutrients, pollutants and oil. The results made clear that in view of regular reporting on these data, a proper methodology for the data processing and presentation had to be developed.

In the comprehensive EEA state and outlook report ‘Environment in the European Union at the turn of the century’ (1999), the actual and foreseeable state of the environment in the EU and the Accession countries has been assessed. The outlook has been based on socio-economic and environmental policies that are assumed to be implemented by 2010. It describes the interrelations between human activities and the environment; it serves to inform policy makers on developments in environmental parameters and the effects of measures taken. As such the report is a background for strategic policy development. The main challenges and problems in the coastal zones of the four regional seas, the North EastAtlantic, the North Sea, the Baltic and the Western Mediterranean are reviewed. Pressures and states are analysed, but quantitative data reflecting the effectiveness of environmental and specific marine policies on the European level are restricted. This is partly due to what is identified as main problems encountered: the different reporting obligations, the different pace in EU and non EU countries of the Mediterranean and the lack of resources for the Black Sea.

The report ‘State and Pressures of the marine and coastal Mediterranean environment’ (1999) of EEA and UNEP/MAP is the first comprehensive report for a regional sea which follows the DPSIR approach of EEA. It is based on scientific literature, data and technical reports from UNEP/MAP and other international organisations and at the national level. It is considered a milestone towards an overall state and outlook assessment of the environmental situation of the whole Mediterranean basin. Through this overview of best available information also the gaps in knowledge become visible.

OSPAR is finalising the publication of the first Convention Wide Quality Status Report, the QSR 2000. OSPAR is perhaps the most effective regional convention world-wide. Regional task teams, with national experts from the countries involved, were set up to write regional reports for the five areas within the Convention. This approach resulted for each report in a thorough analysis of pressures, state and impacts on the biota of the regions. The major causes of environmental degradation are identified as well as the improvements achieved. Recommendations are given for managerial and scientific actions. The different reports reflect clearly the major problems and lack of knowledge encountered. The overall assessment summarises the impact of human activities, evaluates the effectiveness of measures, reflects the limitations of knowledge and identifies priorities for action. Efforts are needed in all fields to improve the efficiency and effectiveness of capturing and analysing data. Cost-effective monitoring systems and other means of gathering information and a better allocation of available resources to the various needs is essential. Looking at the working procedures of OSPAR integrated assessments, these can be described as participative, reflecting feedback and communication and common agreement on the final text.
The time taken for the communication process is substantial. The working procedures of the EEA integrated assessments are less participatory. National expert checks are mostly done afterwards so the whole process can be speeded up.

If both working procedures come more together in terms of effective and efficient communication and feedback, there is a better chance in the aims of the Inter-Regional Forum to harmonise the reporting systems.

From the above integrated assessments it became clear that an intensive collaboration between EEA and Conventions is required to produce comparable, compatible and verifiable information on the state of the marine environment at the European level.

On the technical level, the development of indicators within the context of the DPSIR framework within the Conventions contributes to transparency of integrated analysis.

On the institutional level, the role of the Inter-Regional Forum and its working groups need to be enhanced.

TOWARDS A COMMON MARINE INFORMATION AND COMMUNICATION STRATEGY

Harmonisation of monitoring and assessment for nutrients and harmful substances is ongoing on the European level. Also the ecosystem characteristics of key species and habitats are identified and international collaboration in the monitoring to reporting chain for biodiversity is beginning to develop. Many working groups of ICES and within the Conventions are elaborating specific stages of the information cycle (Figure 5). The European Commission (2000) identified the need for more coherence in national and European research activities.

For the Marine and Coastal information, prioritisation of themes per regional sea, integrated monitoring, and a common system of scientific and technical references are among the issues to be addressed.

Now, as collaboration on the European level is growing, time has come to review the whole information and communication cycles in the context of integrated state and outlook assessments at the European level and to stimulate public communication and debate.

The need for transparent and comparable information has put a greater emphasis on the relevance of well maintained and accessible international databases and more frequent reporting on relevant issues for the regional seas.

The need for a better exchange of marine and coastal information at the European level, taking into account also lower levels of information aggregation (Conventions, regions) has been expressed at the INFOCOAST conference (1999) and in the context of the European Integrated Coastal Zone Management strategy. The European CoastBase project (2000-2001) is developing a pilot technical architecture for an easy search and access to distributed data and information about the marine and coastal environment. This innovative tool serves not only general or specific endusers of marine and coastal information but also the data providers and data centre administrators.

Conventions could contribute considerable if they adapt their monitoring and assessment requirements and data management to the needs of data and information provision at the European level. Strategic aspects of co-operation between the Marine Conventions and the European Community are now being identified.

Figure 5 The information cycle for Marine and Coastal information at the European level
The Integrated Coastal Zone Management Strategy of the Commission is a step forward towards sustainable management of coastal zones. It requires good access and use of information and knowledge. A common strategy for information and communication on the European level in which not only the institutional arrangements and content is reflected but which takes also into account the strategic possibilities of the concept of CoastBase as the European virtual Coastal and Marine Data Warehouse is recommended.

REFERENCES


ENVIROMENTAL INFORMATION PERSPECTIVES IN CENTRAL EUROPE
EMERGING VIEW OF INFORMATION NEEDS

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This paper on “Environmental Information Perspective in Central Europe” has been prepared for the 3rd Monitoring Tailor Made Workshop and will include information about the situation related to environment management in Romania.

Having in mind that the countries situated in Central and Eastern Europe have belonged in the past to the same political system, and from an economic point of view the development has been based on the same framework the conclusions presented in this paper are referring to the Romania but in many respects can be considered valuable for other countries from the region as well. However the differences from country to country existed and still exist

Keywords: monitoring/ information/ quality of environmental factors / water quality / Danube River Basin / management of environment/

BACKGROUND

As was mentioned above, the present papers is referring mainly to environment management in Romania. Romania is located in the Central-Eastern part of the Europe being crossed by the 45 parallel and limited in the west by the 20 meridian and the east by the 30 meridian. In the South, Romania is bordered by the Danube River and the south - east by the Black Sea: it/ has as neighbours, Hungary and Yugoslavia to the west, Bulgaria in the south, Moldavia and Ukraine in the east and North. Nearly all the country is part of the Danube cachment area (98%), the rest 2% being part of the Black Sea cachment area.

The area of the country is 237.500 sq. km in which can be found mountains and mountain plateau, large plains around the mountains, and network or rivers measuring about 125.000Km length, out of which 75.000 km are in the hydrological monitoring network and about 22.000km in the water quality monitoring newtork. In the country, natural lakes cover about 1.1% of the surface area; there are about 550 impounding reservoirs in which one third of the annual average water storage for the country is kept and than 130 dams higher than 15 m. In the European context, taking into consideration the geographical position, it might be concluded that Romania is the main final receiver (the Danube River via Delta and the Black Sea territorial waters) of pollutants coming from the Danube River riparian countries as well as taking up the main part of its own pollution impact.

About 22,5 million inhabitants are living in the country, and the impact created by their activities in the environment are monitored by a number a sector - oriented monitoring systems.

THE PRESENT ENVIROMENTAL MONITORING SYSTEM

The information system related to the environment is provided by a number of sector - oriented monitoring systems, this provides a basic information which the people need to fulfil obligations in connection with the management of the respective environment sector (water, air radioactivity, wastes, soil, etc). These systems check the state of environmental indicators and warn if there are any problems so that, users of systems can make various decisions: define strategic goals, make management decisions, or take remedial or prevention action.

In most of cases the results of the monitoring activities are data that represent in a numerical or alphabetical form the result of the measurement/ analyses. Information product is also data, which has been processed into a form that makes them useful for many purposes. In this way the information can therefore be categorised in accordance with the types of needs to be fulfilled. In this way the environmental monitoring system may be viewed as an information system rather than as a producer of environmental data.
In this respect, when the environmental system is designed as an information system it will produce a direct flow of information from the various tasks such as sampling, laboratory analysis, data processing, data analysis, reporting and information utilisation. For each environmental medium interface and between them the elements of the monitoring systems has to be evaluated and adjusted in accordance with the new needs.

The integrated approach to monitoring the environment is the new direction in which, starting from 1993, the MWFEP has started to organise the sector oriented systems. The goals of the integrated monitoring system in Romania are:

- knowledge of the state of the environment
- elaboration of decision - making structure for sustainable social development
- supervision and description of the changes in environment and resources
- assessment of these changes and forecast connected with the preparation of necessary steps for preventing as well as for connecting the negative tendencies with the possible consequences.

In order to harmonise the meaning of how this is going to be applied, a number of specific principles for Romania were agreed at different levels of decision and execution. These are:

a) The monitoring system should provide information about both the sources of pollution (the emissions) and the environmental quality changes resulting from those emissions, and the use and state assessment of the resources themselves.

b) The monitoring system should provide information about not only the specific sectors (water, air, soil, etc) but also about the linkages between the respective sectors.

The implications of these objectives are that the design and implementation of such an integrated monitoring system entails adding, restructuring and reallocating human and technical resources. Because of the limited resources available and the need to develop monitoring systems in sectors not presently monitored, a monitoring system that is developed step by step and not dependent or fragmented, diverse and unrelated activities will be far more cost effective and professionally more rewarding for the workers, experts and decision - makers involved in the environment monitoring in Romania.

As an example, water monitoring in Romania was focused on the protection of water-users interests (e.g. drinking water resources rather than protection of the ambient water quality) For instance, surface waters are classified and monitored according to water - usage oriented quality standard (STAS 4706-88).

Mainly physical and chemical variables are included in the monitoring programs and this part of the monitoring programme is working reasonably well. Only a limited number of biological variables are measured. Physical and chemical water quality variables are well suited for describing the momentary usability of the water resource for a specific purpose and they are easy to sample and analyse, but this measure of the water quality is only at a single moment in time, usually monthly. To obtain an integral measure of the water quality at a certain place, it is envisaged to monitor a media that is continuously exposed to the flowing water, e.g. sediment samples, macrozoobenthos samples, fish stock and the impact of respective pollution on the biota. Efforts are made to strengthen and upgrade the monitoring of these kind of water- quality variables. A particular attention is paid in the new upgrading of the monitoring programmes to the evaluation of pollution transport. Calculation of pollutants loads is not regularly reported, either from point sources, non- point sources or tributaries. The necessary data (concentration and discharges) are generally available but the collection of these is not done in way that may allow the mass-balances to be established.

Generally, the basic framework and structure of an effective monitoring programme for water already exist. What is be to improved, is a more stringent fulfilment of the established monitoring objectives, a strengthening of sediment and biological monitoring, and standardisation and inclusion of quantity monitoring speciality dedicated for loads assessments.

The above comment to the rivers, as well as measures to be taken regarding physical, chemical and biological monitoring apply as well to the lakes monitoring. A special emphasis in the case of lakes is the sediment monitoring diversification, because as is well known, the sediments play a major role in eutrophication as a consequence of sedimentation / resuspension processes of nutrients and also of different pollutants in lakes.
With reference to the waste waters and in the context of grant authorisations process as well as for calculation of the fees and fines, the quality and quantity of the discharges are monitored. It can be said that the monitoring of aquatic emissions are performed in a relatively complete way. The main gaps that needs to be specified and corrected are referring to the quality control of the data obtained from a very heterogenic system, and especially in the organisation and responsibility structure.

The information flow and data availability are not optimised and the responsibilities of the water suppliers and the controlling authorities are not well prescribed in the current application of the law.

AN EMERGING VIEW OF INFORMATION NEEDS

One of the main objectives of Romania is accession to the EU, and as a result of this the Romanian government in generally, and MWEPF in particular has oriented the policies in order to meet the EU requirements and to comply with the targets and directives which has been set for that. In respect with this target the Romanian Governments and respectively the Ministry of Water Forest and Environment Protection have placed these tasks in the forefront of their activities. Some examples of what have been achieved in this respect, are:

- the integration of near all environmental factors within the responsibility and jurisdiction of only one body, the Ministry of Waters Forest and Environment Protection
- a continuous activity and work for harmonisation of the environment related legislation with the EU legislation.
- a very active participation in all co-operation activities in the CEE region for improving the State of Environment
- provision of documented answers to the different requests from a number of international environmental organisations like European Environmental Agency (EEA) for CORINNE programmes, Dobris Reports, Helsinki Convention- Tasks Force, International Commission of the Danube River Protection Convention, UNDP/GEF Danube Pollution Reduction Programme

Because the situation is in a continuous flux at the economical and structural level in the Central and Eastern Countries, new policies for environment protection and an improved management were adopted as overall objectives. As a result of this, new needs have appeared because of:

- new conditions created by the co-operation at the regional level (CEE)
- restrictions and conditions imposed for accessions to the European Union
- harmonisation of the Environmental policies to the EU, UN and other international organisations to which the countries are Signatories Parties.

The new needs that appeared are:

- improvement of the data compatibility in respect with the requirements for compliance with the national and international data gathering, processing and reporting obligations
- strengthening of the data and information quality, based on the level of performances imposed by interregional utilisation
- adoption of the monitoring programmes (based on the integrated management and assessment approach) based on systematic approach according to the tasks for which the information will be used or based on a problem-oriented approach in which the information production is oriented for finding the solution of the specific problem.

In Romania changes at the political level have also influenced the environmental policies in respect with a number of new and specific needs:

- the identification and rehabilitation of the affected areas – hot spots
- to respond to a greater number of international and regional engagements for producing and delivering the information for global and regional environmental assessment
- to the elaboration and implementation of the local and regional Environment Protection Strategies, Strategic Action Plans etc.
- the correct information of the multiple users and also the public, with regard to the real state of environmental factors mainly for those that are used as resources for socio-economical activities (water, air)

In this respect the environmental monitoring programmers as information providers were directed to cover those needs mentioned above.
At the national level the water-monitoring system is providing information connected with the quality assessment and the management ways of both emissions and imissions. The primary objective for ecological protection is to ensure all water users with the quantities and quality that is in accordance with the national standards and laws.

HOW THE SYSTEM WORKS IN THE CASE OF AN ACCIDENT

The accidental pollution with cyanide and heavy metals from AURUL Baia Mare January 2000

The causes of the accidental pollution

The accidental pollution with cyanide was caused by the bursting of the dam of the sedimentation pond, containing cyanide, used by the S.C.AURUL S.A. Baia Mare plant. The pond has a surface of 93 ha and is situated at a distance of 1.2 km far from the Lapus river. The water resulting from flotation process remains in the sedimentation pond and by the applied technology is totally recycled. The technology used for the gold extraction by the S.C.AURUL S.A. Baia Mare is a new technology which was used for the first time in USA at the Homestake mine in 1971 and in present is used in Canada, USA, South Africa, France, etc.

The melting of the snow layer, with a depth of 43 cm and about 35.7 mm precipitation fallen during the day of 30.01.2000 caused the quick rise of the water level in the pond. The inner dam broke on a length of approximately 25 m and a height of about 3 m, and the exterior dam broke also on a length of about 5 m an a height of about 1 m, being protected by a layer of PEHD.

The spilling of the dam begun in the day of 30.01.2000, around 22.00 h, and by the created break, a volume of approximately 100,000 m³ flooded a surface of 14.78 ha of land close to the pond and then flew through the de-watering channels into Lapus River and from here to Somes, Tisa and Danube Rivers.

During the morning of 31.01.2000, due to the undertaken measures, the accidental discharged flow was about 200 l/s, with a tendency of decreasing, being in the day of 01.02.2000 at the 8.00 h, at approximately 50 l/s. At S.C.AURUL S.A. Baia Mare the operation of the plant was stopped in the day of 30.01.2000, 23.00 h. Also the measures for closing the break were taken as well as for neutralisation of the cyanide spilled from the Aurul pond, with sodium hypochlorite.

Starting of the alert

The polluter S.C.AURUL S.A. Baia Mare didn’t warned directly the territorial units of the National Company "Romanian Waters", respectively Maramures Water Management System or the Somes-Tisa River Authorities. Beginning with the day of 31.01.2000, 8.00 h, the Somes-Tisa River Authority, was announced by the Environmental Protection Agency from Baia Mare, and they warned the County Authorities, the Ministry of Water, Forests and Environmental Protection, the National Company "Romanian Water" (C.N.A.R.), the Hungarian authorities and the downstream water users from the Somes river.

Also the local authorities were alerted about the affected water courses and the interdiction to use the river water for any domestic needs or for animals drinking supply, as well as for the river fish consumption.

Taking into consideration that the Hungarian authorities were announced with about 16 hours earlier that the pollution wave reached the Hungarian territory, the head of the Hungarian delegation at the extraordinary meeting of the bilateral Water Quality Sub-Commission from 08.02.2000 at Satu Mare, agreed that in the meeting's Minutes to be mentioned: "The water quality sub-commission ascertains that between the hydro-technical bodies of Romania and Hungary has been carried on an exemplary co-operation regarding the exchange of information and the laboratory analysis". The same appreciation has been made by the Hungarian Ministry of Environment and other politicians.

According to the Provisions from the Operation Rules of the Romanian Warning System for Accidental Pollution, created within the International Commission for the Protection of Danube River with EU financial support, in the case of an accidental pollution with transboundary effects, the Ministry of Waters, Forests and Environmental Protection (M.A.P.P.M.) is the coordination and the decision authority.

By PIAC (Principal International Alert Centre), there were announced the countries from Tisa
and the Danube catchments area: Hungary, Yugoslavia, Bulgaria, Moldova and Ukraine. According to the provisions of the same Rules, the territorial authorities of the National Company "Romanian Waters" has started the data transmission system and also started the monitoring activity with the clear tasks to assess the propagation of the pollution wave and to alert accordingly the downstream water users.

M.A.P.P.M was involved as well in this activity, in charging the National Reference Laboratory which is the National Research Institute for Environmental Protection (I.C.I.M.) Bucharest, with the validation of water quality data. The National Reference Laboratories–Network was created in the same international frame of ICPDR with EU support and is part of the Transnational Monitoring Network (TNMN) of the Danube catchments area involving 12 countries from the region.

At the County level, the Prefect of the County which is in the same time the Head of the Disaster Committee, was involved in coordinating the activities for reduction the pollution effects of cyanide wave together with the Municipal waters authorities and the Inspectorate for Sanitary Protection.

In Annex 1 there are presented the actions and measures, that were taken by the Romanian Ministry of Waters, Forests and Environment Protection.

Figure 1. The main water quality monitoring sections on the Somes, Tisa and Danube River

The monitoring of the event

The monitoring sections have been chosen, mainly, upstream of the water supply intakes, for the purpose of warning and taking the necessary actions to close this in the cases of exceeding the maximum admissible limit which in Romania is 0.01 mg/l for cyanide concentration, according to the Romanian standard STAS 4706/1988.

In particular the following steps were taken:
- water samples with a frequency of 6 in 6 hours till the concentration reached the limit of 0.01 mg/l and with 2 in 2 hours in the time when the limit has exceeded this, for the cyanide analysis and of 6 in 6 hours for the heavy metals; from the samples taken before, during the pollutant wave trespassing and after the passing of the wave, there were analysed also the biological parameters in order to establish the size of the created impact.
- bottom sediments samples were collected, before and after the pollution wave passed in every station
- fish and mollusc were collected, before and after the pollution wave passed;
- samples with weekly frequency were collected, from the wells and from some domestic wells situated in the plain, near the river;
- water and sediment samples, fish and mollusc were collected in the standard profiles situated on the Danube and also in Black Sea coast, south of the Danube discharge through the Danube Delta.

Generally the sampling from rivers has been done near the bank and in the middle of the wet section (hydrologic profile). The performed measurements has shown the facts that cyanide concentration was not constant in the whole control sections, indicating also that the mixture of water-cyanide wasn’t homogenous, taking in consideration the unpolluted water inflows into the main river from the tributaries. In order to get better knowledge’s about the cyanide concentration evolution, in some sections complete measurements were performed. In figure 2 there is shown the cross section profile and the points on verticals where the sampling was performed. This measurements were made for the control section Giurgiu on the Danube in the day of 22.02.2000, 17.00 h. It can be mentioned that the concentrations values measured vary in the section between 0.025 and 0.082 mg/l. The average calculated concentration was 0.054 mg/l, while the measured concentration for the usually sampling points was 0.055 mg/l, very close to the average concentration.

For the control section situated far from the fix laboratories, the cyanide concentrations were measured by means of portable laboratories and also samples for lab analysis were collected. The samples analysed both in the laboratories and "in situ" with the portable laboratory, allowed the determination of the correction coefficient for the measurements made with portable laboratories.

Water samples was analysed, mainly, in the National Company "Apele Romane" territorial units and in parallel together with the sediment samples, fishes and mollusc- muscular tissue- were analysed in the laboratory of the Research-Development National Institute for Environmental Protection – ICIM Bucharest-which is the NRL in Romania. In some cross-section analyses were done also by the local units of the National Water Company and by the Environment Protection Agency.

![Figure 2. The cross section and the verticals for cyanides concentrations measurements at the control section Giurgiu on the Danube River](image)

The characteristics of the pollution wave

The analysis performed by the Maramures Water Management System of C.N.A.R, on the samples taken from water spilled from the AURUL sedimentation pond, has indicated that 126 mg/l CN- and a total cyanide content of 405 mg/l. In the river Lapus the measured concentration was 19.16 mg/l, after the dilution of the spilled water from the sedimentation pond, with the Lapus river flow, which had 13.8 m3/s during the period 31.01.2000 - 01.02.2000.
The temporal evolution of the cyanide concentrations in the main control sections of the rivers Somes, Tisa and Danube are presented in figures 3, 4 and 5. In tables 1 and 2 are shown the cyanide pollution wave characteristics for the main sections on the Somes, Tisa and Danube rivers. The maximum concentration of the pollution wave decreased from 13.26 mg/l in the control section Cicârlau, to 7.8 mg/l in the control section Satu Mare on the river Somes, and to 1.49 mg/l in the control section Tiszasziget (at the Hungarian-Yugoslavian border) on the Tisa river. About 0.342 mg/l were found in the section Bazias (entering point of the Danube into the Romanian territory) and about 0.049 mg/l in the control section Sulina, on the Danube river. The declining of the maximum concentration of the pollution wave, during its downstream propagation was possible due to the dilution and dispersion processes. On the Somes, Tisa and Danube rivers, the average propagation velocities of the pollution wave were about 0.63 m/s (Somes), 0.73 m/s (Tisa) and 0.95 m/s (Danube). The characteristics of the river sections, the duration and the average propagation velocities of the peak of the cyanide pollution wave are shown in tables 1 and 2. The measurements performed before, during and after the propagation of the pollution wave, regarding the heavy metals concentrations, showed a significant increase of the heavy metal concentrations, in comparison with the admissible limits for the Lapus and Somes rivers, referring to the Cu, Zn, Pb, Fe and Mn (table 3). On the Danube river, the most important values, exceeding the limits were recorded for Cu, in the control section Bazias.

The preliminary evaluation of the effects

The pollution of the rivers: Lapus, Somes, Tisza and Danube had in principle effects on:
• water users;
• water environment
On the Romanian territory water supply was interrupted in 24 localities presented in the table no.4.
Besides the inconveniences caused to the citizens, the pollution generated supplementary costs in the sanitary field by the transportation and treatment of the patients in other localities, in industry, by the diminution or even interrupting of the production process. The pollution affected the fish resource, on the Romanian sector of the Somes River the death fish quantity was relative small, in comparison with the quantities declared by the Hungarian Authorities, on the Tisa River, in spite of the fact that the maximum cyanide concentration determined there were less with 2-5 times, in comparison with those that were identified in Romania. On the Danube River there were no fish mortalities observed.
It can be mentioned the following experiment done: in a 2 mg/l solution of cyanide, which is the maximum concentration measured on the downstream sector of Tisa River, it was introduced a fish from Somes River, which after 2 days was still alive. In the figures no.6 - 9 there are presented the evolution in time of the phytoplankton and zooplankton, in the control section Satu-Mare on Somes River and control section Chiciu (Ostrov) on Danube River.
On Somes river the phytoplankton decreased from 250.000 number of organisms per liter, determined on the day of 20.01.2000, to 0 organisms per liter, on 01.02.2000, after which, it increased to 145.600 organisms per liter on 16.02.2000. Those results showed that the phytoplankton was regenerated in 16 days in the proportion of 60%. A similar evolution had also the zooplankton, (figure no.7 ) which was completely regenerated in 16 days.
The analysis of the samples taken on 28.02.2000 by the Romanian experts, who accompanied the UNO experts on the Hungarian territory, shows also the regeneration of the phytoplankton and zooplankton on the Somes River, downstream Satu Mare and Tisa River (table 5).
On Danube River in the control section Chiciu, the density of phytoplankton (figure8) decreased to 26% in the moment of the peak concentration and after 4 days increased to 75%.
The zooplankton (figureno.9) had a similar evolution, decreasing to 6,8% in the maximum concentration moment and after 4 days increased to 68%.

This work was elaborated based on the data transmitted, in real time, till the date of 29 February 2000, by the territorial units of the National Company "Apele Romane" and based on the analyses of those data, regarding their homogeneity and comparability and based on there corroboration with the data obtained by the United Nations experts and with the results of analyses produced in the National Reference Laboratory - ICIM Bucharest.
Table 1. The characteristics of the cyanide pollution wave in the main control-sections on the Rivers: Somes, Tisa and Bazias on the Danube

<table>
<thead>
<tr>
<th>River</th>
<th>Control-section</th>
<th>Distance between sections (km)</th>
<th>Peak Time</th>
<th>Peak Concentration (mg/l)</th>
<th>Discharge (mc/s)</th>
<th>Travel time (hours)</th>
<th>Velocity (m/s)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Somes</td>
<td>Cicârlau</td>
<td>-</td>
<td>31.01</td>
<td>11.00</td>
<td>13.26</td>
<td>111</td>
<td>-</td>
</tr>
<tr>
<td>Somes</td>
<td>Satu Mare</td>
<td>56</td>
<td>1.02</td>
<td>11.30</td>
<td>7.8</td>
<td>148</td>
<td>24.5, 0.63</td>
</tr>
<tr>
<td>Tisa</td>
<td>Szolnokxx</td>
<td>414</td>
<td>9.02</td>
<td>4.00</td>
<td>2.85</td>
<td>805</td>
<td>183.5, 0.63</td>
</tr>
<tr>
<td>Tisa</td>
<td>Csongradxx</td>
<td>94</td>
<td>10.02</td>
<td>12.00</td>
<td>2.9</td>
<td>840x</td>
<td>32, 0.81</td>
</tr>
<tr>
<td>Tisa</td>
<td>Mindszetxx</td>
<td>26</td>
<td>10.02</td>
<td>20.00</td>
<td>2.0</td>
<td>1170x</td>
<td>8, 0.92</td>
</tr>
<tr>
<td>Tisa</td>
<td>Tiszaszigetxx</td>
<td>57</td>
<td>11.02</td>
<td>12.00</td>
<td>1.49</td>
<td>1800</td>
<td>17, 0.93</td>
</tr>
<tr>
<td>Danube</td>
<td>Bazias</td>
<td>295.6</td>
<td>15.02</td>
<td>14.00</td>
<td>0.342</td>
<td>8700</td>
<td>98, 0.83</td>
</tr>
</tbody>
</table>

x – estimate values
xx – values provided by Hungarian Authorities

Table 2. The characteristics of the cyanide pollution wave at the main control sections on the Danube River, on the Romanian territory

<table>
<thead>
<tr>
<th>CONTROL SECTION</th>
<th>Km</th>
<th>Distance between sections (km)</th>
<th>Peak Time</th>
<th>Peak Concentration (mg/l)</th>
<th>Q (m³/s)</th>
<th>Lag Time Hours</th>
<th>Propagation velocity (m/s)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bazias</td>
<td>1075</td>
<td>-</td>
<td>15.02</td>
<td>14.00</td>
<td>0.342</td>
<td>8700</td>
<td>-</td>
</tr>
<tr>
<td>Gura Vaii</td>
<td>941</td>
<td>134</td>
<td>17.02</td>
<td>18.00</td>
<td>0.158</td>
<td>8600</td>
<td>52, 0.71</td>
</tr>
<tr>
<td>Calafat</td>
<td>795</td>
<td>146</td>
<td>19.02</td>
<td>19.00</td>
<td>0.112</td>
<td>8500</td>
<td>49, 0.83</td>
</tr>
<tr>
<td>Bechet</td>
<td>679</td>
<td>116</td>
<td>20.02</td>
<td>21.00</td>
<td>0.106</td>
<td>8600</td>
<td>26, 1.23</td>
</tr>
<tr>
<td>Corabia</td>
<td>635</td>
<td>44</td>
<td>21.02</td>
<td>9.00</td>
<td>0.103</td>
<td>8680</td>
<td>12, 1.01</td>
</tr>
<tr>
<td>Turnu Magurele</td>
<td>597</td>
<td>38</td>
<td>21.02</td>
<td>20.00</td>
<td>0.103</td>
<td>8810</td>
<td>11, 0.96</td>
</tr>
<tr>
<td>Zimnicea</td>
<td>554</td>
<td>43</td>
<td>22.02</td>
<td>8.00</td>
<td>0.10</td>
<td>8840</td>
<td>12, 0.99</td>
</tr>
<tr>
<td>Giurgiu</td>
<td>495</td>
<td>59</td>
<td>22.02</td>
<td>22.00</td>
<td>0.095</td>
<td>8860</td>
<td>16, 1.02</td>
</tr>
<tr>
<td>Oltenita</td>
<td>430</td>
<td>65</td>
<td>23.02</td>
<td>15.00</td>
<td>0.95</td>
<td>9025</td>
<td>17, 1.06</td>
</tr>
<tr>
<td>Chiciu</td>
<td>375</td>
<td>55</td>
<td>24.02</td>
<td>5.00</td>
<td>0.093</td>
<td>9120</td>
<td>14, 1.10</td>
</tr>
<tr>
<td>Gropeni (Braila)</td>
<td>197</td>
<td>178</td>
<td>26.02</td>
<td>5.00</td>
<td>0.086</td>
<td>-</td>
<td>48, 1.03</td>
</tr>
<tr>
<td>Galati</td>
<td>159</td>
<td>38</td>
<td>26.02</td>
<td>15.50</td>
<td>0.075</td>
<td>10000</td>
<td>10.5, 1.01</td>
</tr>
<tr>
<td>Tulcea</td>
<td>71</td>
<td>91</td>
<td>27.02</td>
<td>16.00</td>
<td>0.059</td>
<td>4100</td>
<td>24.5, 1.03</td>
</tr>
<tr>
<td>Sulina</td>
<td>3</td>
<td>68</td>
<td>28.02</td>
<td>10.00</td>
<td>0.049</td>
<td>-</td>
<td>18, 1.05</td>
</tr>
</tbody>
</table>
Table 3. The values of heavy metals concentrations determined in the period January - February 2000 in the AURUL Pond and in the principal control sections on the Rivers: Lapus, Somes and Danube

<table>
<thead>
<tr>
<th>RIVER</th>
<th>CONTROL SECTION</th>
<th>DATE / HOUR</th>
<th>Cu, La=0.05</th>
<th>Zn, La=0.03</th>
<th>Pb, La=0.05</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lapus</td>
<td>Pond-AURUL</td>
<td>31.01.’00 – 10.00</td>
<td>285,5</td>
<td>32,6</td>
<td>0,183</td>
</tr>
<tr>
<td></td>
<td>Busag</td>
<td>20.01.’00</td>
<td>0,030</td>
<td>0,352</td>
<td>0,015</td>
</tr>
<tr>
<td></td>
<td></td>
<td>31.01.’00-12.00</td>
<td>19,5</td>
<td>1,68</td>
<td>0,278</td>
</tr>
<tr>
<td></td>
<td>Cicalau</td>
<td>20.01.’00</td>
<td>0,021</td>
<td>0,196</td>
<td>nd</td>
</tr>
<tr>
<td></td>
<td></td>
<td>31.01.’00-11.00</td>
<td>10,5</td>
<td>0,419</td>
<td>nd</td>
</tr>
<tr>
<td></td>
<td></td>
<td>01.02.’00-20.30</td>
<td>0,638</td>
<td>0,182</td>
<td>0,160</td>
</tr>
<tr>
<td></td>
<td></td>
<td>10.02.’00-17.30</td>
<td>0,116</td>
<td>0,160</td>
<td>0,151</td>
</tr>
<tr>
<td>Somes</td>
<td>Satu Mare</td>
<td>20.01.’00</td>
<td>0,110</td>
<td>0,237</td>
<td>0,022</td>
</tr>
<tr>
<td></td>
<td></td>
<td>14.02.’00</td>
<td>0,099</td>
<td>0,017</td>
<td>0,007</td>
</tr>
<tr>
<td></td>
<td></td>
<td>18.02.’00</td>
<td>0,009</td>
<td>0,158</td>
<td>0,007</td>
</tr>
<tr>
<td></td>
<td>Bazias</td>
<td>06.01.’00</td>
<td>0,009</td>
<td>0,173</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>14.02.’00</td>
<td>0,023*</td>
<td>0,035*</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>15.02.’00</td>
<td>0,155*</td>
<td>0,057*</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>16.02.’00</td>
<td>0,035*</td>
<td>0,042*</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>17.02.’00</td>
<td>0,027*</td>
<td>0,041*</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Moldova Veche</td>
<td>06.01.’00</td>
<td>0,006</td>
<td>0,019</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>15.02.’00</td>
<td>0,084*</td>
<td>0,047*</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>16.02.’00</td>
<td>0,025*</td>
<td>0,038*</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>17.02.’00</td>
<td>0,024</td>
<td>0,035</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>19.02.’00</td>
<td>0,046</td>
<td>0,029</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>20.02.’00</td>
<td>0,041</td>
<td>0,024</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>21.02.’00</td>
<td>0,037</td>
<td>0,026</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Chiciu</td>
<td>21.02.’00</td>
<td>0,0215</td>
<td>0,018</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>22.02.’00-11.00</td>
<td>0,079</td>
<td>0,015</td>
<td>0,013</td>
</tr>
<tr>
<td></td>
<td></td>
<td>25.02.’00-14.00</td>
<td>0,042</td>
<td>0,067</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>25.02.’00-22.00</td>
<td>0,086</td>
<td>0,053</td>
<td>-</td>
</tr>
</tbody>
</table>

Legend: * - daily average values; La - the limit of x compound from the Romanian Standard STAS 4706/1988 for the first water quality category

Table 4. Localities from the Danube region, where the water supply systems were interrupted, during the cyanide pollution wave propagation period

<table>
<thead>
<tr>
<th>No.</th>
<th>Locality</th>
<th>Population (no. inhabitants)</th>
<th>Water intake Danube Q daily (l/s)</th>
<th>Water intake ground water Q daily (l/s)</th>
<th>No. wells</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1</td>
<td>Moldova Noua</td>
<td>15700</td>
<td>-</td>
<td>154</td>
<td>19 wells near the Danube River</td>
</tr>
<tr>
<td>2</td>
<td>Svinita</td>
<td>1600</td>
<td>6,6</td>
<td>Springs</td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>Tr. Severin</td>
<td>118000</td>
<td>837,1</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>Calafat</td>
<td>21400</td>
<td>117,2</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>Turnu Magurele</td>
<td>36000</td>
<td>140,0</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>6</td>
<td>Oltenita</td>
<td>31900</td>
<td>168,0</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>7</td>
<td>Calarasi</td>
<td>77900</td>
<td>312</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>8</td>
<td>Slobozia</td>
<td>56925</td>
<td>380,0</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>9</td>
<td>Fauréi</td>
<td>4600</td>
<td>9,5</td>
<td>9,5</td>
<td>4</td>
</tr>
<tr>
<td>10</td>
<td>Ianca*</td>
<td>12600</td>
<td>70</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>11</td>
<td>Braila</td>
<td>235000</td>
<td>1400</td>
<td>540</td>
<td>30</td>
</tr>
</tbody>
</table>
### Table 5. The evolution of the phytoplankton and zooplankton on the Somes and Tisa River

<table>
<thead>
<tr>
<th>River / control section</th>
<th>Date</th>
<th>Phytoplankton</th>
<th>Zooplankton</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Number of organisms/dm³</td>
<td>Number of species/dm³</td>
</tr>
<tr>
<td>Somes / Oar (border section)</td>
<td>24.02.2000</td>
<td>170,000</td>
<td>21</td>
</tr>
<tr>
<td>Somes/ Csenger</td>
<td>28.02.2000</td>
<td>248,000</td>
<td>22</td>
</tr>
<tr>
<td>Tisa, upstream conf. Somes</td>
<td>28.02.2000</td>
<td>150,000</td>
<td>12</td>
</tr>
<tr>
<td>Tisa, downstream confl. Somes</td>
<td>28.02.2000</td>
<td>230,000</td>
<td>17</td>
</tr>
</tbody>
</table>

* From the same water supply system are supplied also the localities Gropeni and Movila Miresei

Figure 3. Evolution of the cyanide concentrations at the main control sections on the Somes River
Actions and measures which have been taken by the Romanian Ministry of Waters, Forests and Environmental Protection

On February 14, in Baia Mare, a meeting took place between a delegation of the Ministry of Waters, Forests and Environmental Protection from Romania, leaded by Mr. Anton Vlad, the Secretary of State, and a delegation of the Hungarian Ministry of Environment leaded by Mr. Pal Pepo, the Minister.

During the meeting at the Company AURUL SA, the co-operation and action ways of responsible authorities from both States were analysed, starting with the moment of accidental pollution on the Somes River.

On February 17, Mrs. Margot Wallstrom, the European Commissioner on Environment Protection, together with Mr. Romica Tomescu, the Romanian Minister of Waters, Forests and Environmental Protection and Mr. Pal Pepo, the Hungarian Minister of Environment, visited the affected location in Baia Mare area. With that occasion, a Task Force was set up leaded by a nominee representative from the European Union Commission. The Task Force was established in order to act to the following actions:

- analysis of the conditions in which the accidental pollution was produced
- assessment of damages on the Lapus, Somes, Tisa and Danube aquatic ecosystems;
- setting the necessary actions to be carried out in order to avoid further new accidental pollution.

Task Force includes the representatives of United Nation Environment Protection (UNEP), of the Environment Ministries from Romania and Hungary, of the E.U., of the International Commission for the Danube River Protection, and also of WWF representatives.

On this occasion the European Union representative has expressed again the availability of the EU support for Romania and Hungary in order to mitigate the effects of the accident.

On February 18, 2000, within the Ministry of Public Works and Territorial Planning, the extraordinary meeting of the Operating Office of the National Commission for the Dams and Other Hydrotechnical Works Safety (CONSIB) took place. During the meeting was analysed again the situation created by the technical accident from the Aurul-Baia Mare settling pond. Also, in the framework of this meeting were analysed the technical aspects of the accident taking into account the first conclusions of the EU Commission experts, which preliminary controlled the affected objective and ordered the measures for diminishing the risk of producing a new accident and also for setting the objective into a safety state.
In the period 26-27 February 2000, an international expert group, under the UNEP, has made a visit in the Baia Mare area, took samples and discussed, with all involved factors, about the concern of the causes and consequences of the SC Aurul SA. accident at. At this meeting, local and national officials and civil society representatives were present as well. The Romanian Ministry of Waters, Forests and Environment Protection has taken the following measures:

- using the Principal International Alert Centre in case of Accidental Pollution on the Danube River (PIAC 08), alerted all the downstream countries and then these were informed about the event. Also the International Secretariat of the International Commission for Danube River Protection was informed asap.;
- has monitored permanently the situation of water quality and the efficiency of the measures taken for stopping the pollution source;
- has informed in written the Baia Mare Court of Justice-Prosecutor’s Office;
- has organised an extraordinary meeting of water quality Sub-commission within the Romanian-Hungarian Hydro-technique Joint Commission, for the inspection of the accident location;
- has nominated a commission of experts in order to analyse the safety of the settling pond protection dam and to establish the further necessary measures that has to be taken. The conclusion of the Commission were analysed within the Operating Office of the National Commission for the Dams and Other Hydrotechnical Works Safety, being jointly agreed on the necessary measures to be applied in order to avoid another new accidental events;
- during the meeting held in Oradea, on February 10, 2000, together with the Hungarian authorities representatives, were analysed the circumstances of the accident and the cooperation measures were agreed between both parties in order to monitor the state of Tisa River water pollution, including the common water sampling programmes.

The Ministry of Waters, Forests and Environment Protection, will strengthen the control regarding all similar objectives, and will propose to amend the existing regulations, in order to include this type of objectives in the corresponding importance class according to their risk. Also, the National Action Plan for Environment Protection will take in the consideration the economic objectives that present impact risk in trans-boundary context. In this respect, we will try to use with priority the existing technical, economical and financial assistance for achieving the objectives.

Figure 5. The evolution of the cyanide pollution wave on the main control sections on the Danube River in the period 14.02-29.02.200
Figure 6 The correlation between cyanides concentrations, number of organisms and species of phytoplankton in control section Satu-Mare on Somes River in the period 21.01-16.02.2000

<table>
<thead>
<tr>
<th>Date</th>
<th>Cyanide (mg/l)</th>
<th>Phytoplankton (ths.org.dm³)</th>
<th>No. of species/dm³</th>
</tr>
</thead>
<tbody>
<tr>
<td>21.01.2000</td>
<td>0.01</td>
<td>250</td>
<td>13</td>
</tr>
<tr>
<td>01.02.2000</td>
<td>7.8</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>04.02.2000</td>
<td>0.09</td>
<td>120</td>
<td>8</td>
</tr>
<tr>
<td>14.02.2000</td>
<td>0.02</td>
<td>133</td>
<td>14</td>
</tr>
<tr>
<td>16.02.2000</td>
<td>0.02</td>
<td>145</td>
<td>16</td>
</tr>
</tbody>
</table>

Figure 7 The correlation between cyanides concentrations, number of organisms and species of zooplankton in control section Satu-Mare on Somes River in the period 21.01-16.02.2000

<table>
<thead>
<tr>
<th>Date</th>
<th>Cyanide (mg/l)</th>
<th>Zooplankton (ths.org.dm³)</th>
<th>No. of species/dm³</th>
</tr>
</thead>
<tbody>
<tr>
<td>21.01.2000</td>
<td>0.01</td>
<td>250</td>
<td>13</td>
</tr>
<tr>
<td>01.02.2000</td>
<td>7.8</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>04.02.2000</td>
<td>0.09</td>
<td>120</td>
<td>8</td>
</tr>
<tr>
<td>14.02.2000</td>
<td>0.02</td>
<td>133</td>
<td>14</td>
</tr>
<tr>
<td>16.02.2000</td>
<td>0.02</td>
<td>145</td>
<td>16</td>
</tr>
</tbody>
</table>
Figure 8 The correlation between cyanide concentrations, the number of organisms and species of phytoplankton at the control section Chiciu, on Danube River in the period 22.02-28.02.2000

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Cyanides (mg/l)</td>
<td>0.006</td>
<td>0.028</td>
<td>0.093</td>
<td>0.01</td>
<td>0.004</td>
</tr>
<tr>
<td>Phytoplankton (ths.org.dm³)</td>
<td>3058</td>
<td>960</td>
<td>184</td>
<td>1420</td>
<td>2322</td>
</tr>
<tr>
<td>No. of species/dm³</td>
<td>23</td>
<td>11</td>
<td>9</td>
<td>15</td>
<td>18</td>
</tr>
</tbody>
</table>

Figure 9 The correlation between cyanide concentrations, the number of organisms and species of zooplankton at the control section Chiciu, on Danube River in the period 22.02-28.02.2000

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Cyanides (mg/l)</td>
<td>0.006</td>
<td>0.028</td>
<td>0.093</td>
<td>0.01</td>
<td>0.004</td>
</tr>
<tr>
<td>Phytoplankton (ths.org.dm³)</td>
<td>13</td>
<td>11</td>
<td>1</td>
<td>4</td>
<td>9</td>
</tr>
<tr>
<td>No. of species/dm³</td>
<td>6</td>
<td>4</td>
<td>1</td>
<td>4</td>
<td>5</td>
</tr>
</tbody>
</table>
A major collaboration programme has been undertaken by the Environment Agency in the UK (in collaboration with the Scotland and Northern Ireland Forum for Environmental Research (SNIFFER)) and the Po River Authority in Italy, resulting in the development of a common Manual of Best Practice for the design of water quality monitoring programmes. The Manual has the purpose of supplying a guidance for organisations charged with monitoring activities, and it has been organised in different versions for UK and Italy. Both versions refer to common principles, but different approaches to implementing monitoring programmes are recognised. For a given monitoring objective, the Manual gives the user step-by-step guidance through the choice of an appropriate monitoring strategy, deciding what to measure and how and when to measure it, as well as how to analyse the resulting data and generate management information. 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.

INTRODUCTION

The purpose of this document is to highlight a major collaboration programme undertaken by the Environment Agency in the UK (in collaboration with the Scotland and Northern Ireland Forum for Environmental Research (SNIFFER)) and the Po River Authority in Italy, two of the major regulatory authorities in Europe. This has resulted in the development of a common Manual of Best Practice for the design of water quality monitoring programmes. The collaboration addressed the need to develop efficient monitoring programmes to meet the requirements of EU and national legislation, as well as local operational needs. As costs of water quality monitoring are relevant, opportunities for securing better value for money and potential savings through more efficient monitoring programmes could be considerable.

The development of a Manual of Best Practice is seen as an important contribution to the needs of Member States and in support of the role of the European Environment Agency in promoting best monitoring practice.

THE NEED FOR WATER QUALITY MONITORING

Water quality monitoring may be undertaken to obtain information to satisfy the requirements of EU legislation, international agreements, national legislation, classification schemes and local water management activities.

A number of EU Directives and Decisions (75/440, 76/464, 77/795, 78/659, 86/574, 91/271, 91/676, 2000/60) have been adopted that require monitoring of rivers 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 importance of environmental monitoring is stressed also by the activity of the European Environment Agency: the Agency’s Topic Centre on Inland Waters investigates the status of European water bodies through Eurowaternet, a monitoring network that collects data about the quality and management of surface waters and groundwater in the member countries.

Monitoring is undertaken also to satisfy national requirements resulting from the implementation of specific legislation or to support more general water management activities at the operational level.
In Italy the Regions are charged with water monitoring activities, and the national government is empowered with supervision, co-ordination and regulation tasks. The basic legal framework for water quality protection and monitoring is established by:

- Legislative Decree 152/99, which transposes the EEC Directives 91/271 and 91/676 and defines the main requirements for water quality monitoring in inland waters, coastal water, estuaries and lagoons. For the first time in Italy, the Decree sets environmental and functional objectives for water bodies;
- Inter-ministerial Committee Resolution of 4/2/77, on the protection of water from pollution, which regulates any topic that is not thoroughly defined by the Legislative Decree 152/99;
- Law 183/89, which establishes river catchment as the ecosystem within which environmental protection activities have to be designed and performed, and creates river authorities;
- Law 36/94 concerning the re-organisation of the public services that are charged with water abstraction, water supply and distribution, and wastewater treatment;
- Law 61/94 concerning the re-organisation of environmental controls and instituting the National Agency for Environmental Protection (ANPA) and Regional Agencies (ARPAs)

In England and Wales the legal framework for river monitoring stems from the Water Resources Act 1991 and the Environment Act 1995. Various Sections and Schedules of these Acts require the Environment Agency to undertake monitoring as follows:

- To determine applications for consents to discharge and, although there is no explicit legal requirement on the Agency to monitor discharges to assess compliance with consent conditions, in practice this is the approach taken;
- To meet any Water Quality Objectives (WQOs) applied to protect certain uses of the aquatic environment and maintain them on a public register. River monitoring must be undertaken to gauge the degree of compliance with WQOs. In practice, most WQOs have arisen from the introduction of European Union legislation.
- To “monitor the extent of pollution in controlled waters”. This basic requirement has been taken to mean that river monitoring must be done to gauge compliance with specific environmental quality standards, in order to protect the various uses of water.

Under the Water Act 1989 (subsequently consolidated into the Water Resources Act 1991), use related and classification water quality schemes were separated into two new systems; Water Quality Objectives (Rivers Ecosystem Classification) incorporating the use-related elements, and a General Quality Assessment (GQA) scheme for classification purposes.

The GQA scheme has been 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.

In addition to the monitoring requirements outlined above, monitoring is undertaken also to satisfy the data requirement to support local water management activities.

In Italy, responsibility for environmental protection and monitoring has been largely devolved from the national level to the Regions, provided that the minimum requirements of the national legislation are satisfied.

The Po River Authority has made concerted steps towards the co-ordination of water management activities at the basin level. On 1 July 1993, the Po River Authority’s Institutional Committee approved the project for a basin-scale monitoring system for surface water classification. This agreement was signed by the Authority and the organisations responsible for water management in the Po catchment: six Regions, an autonomous Province and the Meteorological Service of Air Force.

In England and Wales, local monitoring programmes are undertaken by the eight Environment Agency regions for local or regional water quality management purposes. The design of these types of monitoring programmes is undertaken at the local level, on a case-by-case basis. The range and extent of programmes vary but local monitoring can be divided into different categories, such as discharge impact assessment and pre-consenting studies, development impact assessment, diffuse source impact assessment, trend detection and general quality characterisation, national and regional research and development, post pollution incidents, real-time water quality management, model development and validation.

The diverse origins of the eight Environment Agency regions means that, approaches to monitoring under these categories differ.
In terms of expenditure, the most important categories are discharge impact assessment and pre-consenting studies; post-pollution incidents; and detection of trends and general quality characterisation. Comparison of the current scale of river monitoring in the Po River Basin and England and Wales indicates that river monitoring is undertaken on a larger scale in England and Wales. More monitoring sites, in terms of both average river length and catchment area, are currently operated in England and Wales (average 1 site/7 km) and a greater number of samples per year are taken at each monitoring site than in the Po River Basin (average 1 site/9 km). The higher density of monitoring sites in England and Wales reflects a larger population density and a more intensive use of rivers. A further explanation of a larger monitoring requirement is the adoption of the Environmental Quality Standard approach to water management rather than the Uniform Emission Standard approach.

DEVELOPMENT OF A MANUAL OF BEST PRACTICE

Our collaboration has resulted in detailed information collected on current monitoring practices from the Environment Agency and the Po River Authority. This has included identifying the requirements of those involved in the design and interpretation of monitoring programmes and the range of tools and procedures used. This information formed the basis of a draft Manual of Best Practice for water quality monitoring. Following this stage, extensive testing of the principles of the Manual was made in a series of test catchments in both the UK and Italy. The final versions of the Manual were completed in both countries in early 1999. We have agreed that while principles enshrined in both manuals must be common, differences in approach to implementing programmes must be recognised. So we have produced two versions of the Manual:

These give step-by-step guidance through all the stages of a monitoring programme. For a given monitoring objective, the user is guided through the processes of choosing an appropriate monitoring strategy, deciding what to measure and how and when to measure it, as well as how to analyse the resulting data and generate management information. 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.

A key feature of the approach offered by the Manual of Best Practice is the ability to design a cost-effective monitoring programme within the constraints of the staff and other resources available.

The UK Version of the Manual of Best Practice

The Manual of Best Practice consists of the manual itself and nine associated software tools. Through the use of the software tools, the principles of the Manual are put into practice when planning a monitoring programme.

The Manual itself is split into four parts.

Part A, Concepts, lists the acknowledged business needs for water quality monitoring. Six monitoring strategies that can be used to address these needs are then defined: quality characterisation, spatial comparison, spatial trend detection, temporal comparison, temporal trend detection, and before-after control impact (BACI). The process of designing a monitoring programme to address any particular Strategy is broken down into a sequential series of logical steps. A vital element of this process is an understanding of the types of variability present in the water body of interest (e.g. seasonal, tidal, diurnal), and the mechanisms that help to cause those variations (e.g. temperature, rainfall, intermittent discharges, flow). Part A provides a comprehensive review of this important theme, together with a general overview of the planning process.

Examples of diurnal and seasonal variations are shown below (Figures 1 and 2).
Part B then supplies a more detailed treatment of the planning and analysis activities. First, a
general introductory section deals with each of the steps in turn, guiding the user through the
decision-making processes with a combination of general advice on factors to consider and,
where appropriate, detailed step-by-step guidance through statistical procedures. Details
specific to the individual strategies are then provided in the subsequent six sections. Two key
aspects of the manual are its use of a resource-driven approach reflecting what usually happens
in practice, and its emphasis on assessing the likely value of the information to be generated by
the proposed monitoring programme. This ensures that a cost-effective programme can be
designed for fixed resources.
The nine software tools produced in support of the methodology are set out in the Manual. Four
of the tools deal with the planning aspects of a programme, the other five are to help with
generating management information through the statistical analysis of data gathered by the
monitoring programmes.

The tools are:
1. Tables of typical variability for a series of parameters at a range of sites across the UK. This
tool is designed to obtain information on typical components of variability in the system
that is proposed to be monitoring in the absence of existing supporting data.
2. Programme Planning Worksheets. These are used to enter the details of the proposed
programme and the components of known variability within that system. This then gives a
value for the residual variation left over in the system not covered by the programme as
designed.
3. Precision Tool Worksheets. The information generated in the Planning Tool is used to
predict the expected performance of the planned programme. This is where the user can see
what will be obtained from the programme, e.g. what trend will be detected over what time period if the planned number of samples are taken, and can then adjust the programme to meet the initial aim. An example output is shown below (Figure 3):

**Figure 3 Programme Planning output**

In this case this shows that the selected programme will detect a year on year trend provided that the difference in means is greater than 0.393 units each year.

4. Monitoring Programme Simulator. Allows the user to simulate a monitoring programme based on the known components of variability and gives a vivid demonstration of its precision and bias.

The Data Analysis Tools are:
1. Understanding Continuous Monitoring Data. This tool provides a comprehensive analysis of high frequency data (e.g. that obtained from a continuous monitor typically taking readings every hour);
2. Analysis of Spatial Comparisons. This tool carries out a paired analysis of any determinand common to two data sets. It is particularly useful in comparing data between two sites in a catchment to detect differences in quality or trends in quality;
3. Temporal Trend Analysis. This tool enables the user to detect both linear and step-change trends in a data set at a given site and carries out a statistical test to decide which of these is the most significant;
4. Analysis of temporal (before-after) comparisons. This tool allows the user to decide at which point in a data series to analyse if there is a statistically significant difference in quality. This is particularly useful if for example a known impact has taken place at a given time, e.g. an improvement to a discharge came on line, and you want to know if this has resulted in an improvement in water quality.

An example is shown below (Figure 4):

**Figure 4 Temporal (before-after) comparison**
5. Analysis of spatial trends in biological data. Compares biological data e.g. number of taxa, BMWP scores, average score per taxa, EQIs, at a series of sites along a river system. Allows the user to detect statistically similar quality or where there is a trend in quality. This helps with the identification of potentially redundant sites or where extra monitoring stations may be needed.

Part C of the Manual presents the results of a series of case studies undertaken in the UK during the development of the Manual. The case studies are examples of the various Monitoring Strategies, illustrating the application of the Manual itself and then use of the nine software tools.

The final part of the Manual comprises a series of Appendices comprising a Glossary of statistical terms, an explanation of the need for the “Statistical Approach” to monitoring programme design, background to designing monitoring programmes, typical chemical and biological methods, estimation of mixing zones in rivers, tables of typical variations for selected determinands at some continuously monitored sites, statistical tables, technical details for planning procedures, summary of available software; and user guides to the nine software tools for use in association with the Manual.

**The Italian Version of the Manual of Best Practice**

The Italian version of the Manual of Best Practice refers to monitoring applications mainly in watercourses (rivers, streams, canals) and lakes. It is divided into four parts.

Part A provides general information about each phase in the definition and analysis of a water quality monitoring programme. The approach used for the optimum design of a monitoring programme is based on the concept of using six different monitoring strategies to reach nine identified monitoring objectives.

Part B describes the operational procedure for the design of optimum monitoring programmes for surface water quality. The section describes the logical steps in designing a water monitoring programme and assessing its effectiveness. Particular attention has been paid to the description of sources of environmental variability. A step-by-step procedure is used and the initial set-up can be reviewed and the monitoring programme re-configured accordingly to achieve the optimum cost/benefit ratio.

![Figure 5 The Enza and the Sesia basin](image)
Part C presents the results of the monitoring programmes carried out in the Enza stream and the Sesia river according to the guidelines of Parts A and B. Experimentation with sample basins was designed to verify and fine-tune monitoring techniques, to improve their effectiveness and to identify the determinands defining a water body and the surrounding habitat. The final analysis of effectiveness provided important feedback data for the design stage for future monitoring programmes of surface waters (Figure 5).

The Manual includes also the results of monitoring programmes in Lake Iseo and Lake Viverone, two lakes with different limnological and territorial features.

Part D is divided into five appendices that provide technical and statistical information in order to optimise monitoring programmes and data processing.

**CASE STUDY: THE ENZA STREAM**

The Enza is an Apennine stream whose lower basin is mainly characterised by intensive land farming and cattle harvesting.

The monitoring programme was designed to assess the impact on water quality of floods, of soil leaching following first rainfall and of occasional pollution.

The first station was located in Cedogno, at the outlet of the mountain basin. The location of the second station in Coenzo is a compromise between the need to monitor pollutant loads at the basin outlet and to avoid bias by Po backwaters (Figure 5).

The two stations were provided with flow gauge and automatic samplers; in Coenzo a multiparameter probe was installed to disseminate alarms in case of occasional pollution events. The sampling strategy included fixed-period sampling by sampling crews, storm chasing and occasional pollution chasing by automatic samplers. The main determinands were general physical-chemical parameters and nutrients; additional analyses included also sediment quality and biological monitoring.

Automatic samplers were remotely activated according to the results of flood forecasting, which was performed by processing remote rainfall and water level data through a hydrological model. By integrating fixed-period sampling (generally performed for statutory requirements) with storm chasing, more precise estimates of pollutant loads were provided and important information on their distribution through time and through hydrological regime were given (figure 6).

![Figure 6 Total N load and total P load in Coenzo during a flood event](image)

The multiparameter probe proved to be completely ineffective as an alarm system for emergency intervention during occasional pollution events. This type of system is not sufficiently sensitive to detect the determinand variations caused by pollution events, unless they reach such a level as to cause acute effects. Furthermore the high natural variability of Apennine flow regimes may hide the effects of pollution events.

Sediment quality analyses gave no significant results. The sediment sampling technique was shown to be one of the main sources of variability, particularly in the mountain sections where the particle size is relatively coarse and uneven along the transect.

Biological monitoring provided relevant information on river stretch characterisation and on the effects of structural and functional modifications to the river ecosystem through the study of benthonic macroinvertebrates, diatoms and fish. The results highlighted the difficulties associated with biological monitoring in the potamon reaches of a watercourse, where heavy silt deposition and the poor habitat diversity may simplify biological diversity.
The results suggest investigating the determinands that have proved to be most significant in characterising water quality: nitrates and faecal coliforms, correlating respectively with organic diffuse pollution (arable and livestock farming) and microbiological pollution of mainly civil origin. This represents an easy to interpret, low-cost monitoring and early warning system to complement current procedures, enabling controls and quality trend analyses to be intensified on both the spatial and temporal scale (medium and long term). To supplement the chemical controls, indispensable to identify and quantify sources of pollution, the effects of pollution on the river ecosystem could also be assessed through the use of biological monitoring and river functionality indexes.

**CASE STUDY: THE SESIA RIVER**

The Sesia is an alpine river whose basin is characterised by relevant urban settlements and intensive agricultural activity based largely on rice cultivation.

Water quality monitoring was designed to analyse the interaction between surface water and groundwater, as well as the path of pollutants through these two water bodies. The most relevant pollutants of this area are soluble herbicides, antiparasites and nitrogen fertilisers, which easily seep into groundwater when the rice paddies are submerged, and that come back to surface water through a system of karst springs. The aim of the study included also the optimisation of monitoring methods, frequency and the spectrum of determinands to be measured in relation of the above mentioned phenomena.

The sampling activities were performed over the span of a year in order to monitor the various phases associated with existing crops, fallow periods and the exchange dynamics between surface and underground water. The monitoring network consisted of 13 stations for surface water sampling and 4 stations for spring water sampling, integrated by groundwater sampling carried out by the Piedmont Region.

Five stations for surface water monitoring were provided with continuous flow recording and automatic samplers; in the other ones sampling crews took instantaneous water samples and flow measurements. The four stations for spring water monitoring were provided with automatic samplers and instantaneous flow measurements.

The sampling strategy for surface water involved a weekly mean sample for the detection of herbicides. All the other chemical determinands were analysed on instantaneous samples. The sampling strategy for spring water involved a composite mean sample collected by an automatic sampler over six hours.

Flow measurements were one of the critical elements of the study, mainly due to the looseness of the existing flow gauge network. It is good practice to verify, at least a year before, the suitability of the sites for water quality data and flow measurements and calculate the stage-discharge relation. Even stations apparently without problems must be re-verified. Furthermore, the location of monitoring stations near the basin outlet should be carefully evaluated, in order to monitor all significant pollutant loads and at the same time avoid the biases caused by backwater sampling.

The results of the monitoring programme enabled to eliminate stations with poor significance and optimise the choice of determinands and sampling frequency. Almost 25% of the existing monitoring stations showed poor significance or provided little relevant information for water quality assessment, mainly due to inadequate location. As a result of similar reasoning applied to the monitored tributaries, some of them proved to be of little significance in determining pollutant loads and need not, therefore, be monitored.

The optimum sampling frequencies proved to be monthly sampling; for some stations this frequency had to be improved to weekly sampling during the rice cultivation period (from April to August). The most effective sampling strategy was instantaneous sampling, integrated by mean sampling in Palestro, where the total pollutant loads of the Sesia basin were calculated.
THE VALUE OF WATER MONITORING DATA FOR THE DEVELOPMENT OF NATIONAL NATURAL RESOURCES MANAGEMENT POLICY

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Data obtained from the national routine monitoring system for water quality control should provide answers to the following:

• to determine priority pollutants for certain areas, and their quantitative parameters
• to estimate pollutant loads on the ecosystems
• to provide data on seasonal distribution of pollutants loads, when and where the deterioration of water quality is most severe
• to explain why water quality differs between areas
• to tailor monitoring programs in the most sufficient way

The content of nitrogen and phosphorus were studied in atmospheric precipitation (snow) and bottom sediments in order to estimate nutrient loads on urban landscapes and especially on water ecosystems. Capital town of Moldova was selected as a case-study area for estimation of pollutant loads on urban water ecosystems. Knowledge of the processes of the forming of water quality is the key issue in understanding how to develop water quality targets and monitoring programs in the policy-relevant activities like development of Action Plans, programs etc.

INTRODUCTION

Environmental standards for water quality have been implemented over last decades in order to provide customers with water of certain quality for different purposes (drinking water supply, fishing, bathing etc). Based on these data some protective measures have been undertaken in order to support the water quality standards in certain areas of the country. Actually some of them, due to the changes happened in national economy and shifting of the priorities, are now being recognized as questionable.

First of all financial constraints caused a necessity to modify the actual monitoring network. Water users, especially in the Eastern European countries are not able to support high cost monitoring programs. Large areas of the important water courses in this region are not monitored with the frequency and number of parameters, which have been analyzed before 1990. It raises a question, if the reducing of the number of monitoring stations will not lead to the loss of valuable information? Determining which contaminants are likely to be present in large amounts for certain areas and focusing monitoring activities on them will provide acceptable level of information for developing protective measures and will cost less.

Secondly, tailoring of the monitoring network could be developed based on the spatial variation and adaptability to the contamination. Some areas like wetlands, hilly landscapes etc have different types of soils, rocks, vegetation etc, that creates different conditions for migration and accumulation of contaminants in different parts of the landscapes. Different geochemical barriers in soils and landscapes also lead to variation of contaminant concentrations in different parts and components of the environment.

The understanding of the real state of water resources provided by properly designed scientific monitoring programs could form the basis for tailoring of monitoring network within the country. The objectives of a monitoring network could be:

– to describe current state of water resources for water ecosystems in different landscapes affected in different levels by pollution
– to provide information on the main features of the water quality changes in different regions and landscapes
– to distinguish human and natural impacts on the state of water quality, especially in the case of diffuse sources of pollution
– to develop proposals for decision makers, programs for Action Plans, improvement of water resources management practice etc
METHODS

Analyses of nitrogen and phosphorus were performed according to the standard methodology (Ammonia – ISO 5664, nitrites – ISO 6777, nitrates – ISO – 7890, phosphorus – ISO – 6871).

Field research was conducted on the base of sampling of main river, lakes situated in the basin and atmospheric precipitation.

For sampling of atmospheric precipitation main functional zones of urban landscapes were identified:

- recreational zones (parks, green areas, forested landscapes etc)
- living areas (multistoried bloc building areas, private houses etc) in urban areas
- industrial areas
- traffic areas

Snow was sampled in these areas. First sampling took place immediately after the snowfall in parallel with the sampling of the same material in a nature protected area (Plaiul Fagului) as a reference area located in 100 km from the town. The analyses performed did not show any significant differences between different urban zones and nature protected area. The concentrations of nitrogen and phosphorus detected in these samples were used as a reference for further investigations.

The next sampling was undertaken 10 days after the snowfall. There were no precipitation during this period of time and that is why the difference in concentrations was a base for the calculations and conclusions for pollutant loads in the case-study region (the capital town of Moldova – Kishinev), with the population of around 800000 inhabitants.

RESULTS AND DISCUSSIONS

The primary objectives of the study were identified based on the available monitoring data. So as due to the financial and other constrains a number of stations and parameters were out of the monitoring programs for last 10-12 years next objectives were determined:

- assess and evaluate conditions at the actual Bic river monitoring sites in order to identify their actual representativeness for collecting of water quality information
- to analyze of additional components of river ecosystems, which are not included in the routine monitoring programs
- to estimate factors and calculate pollutants loads, which influence the state of the water quality
- to update the methodology of material accounting for development of nutrient balances

Prunici and Drumea (1999) studied the content of different forms of nutrients (nitrogen and phosphorus) in different types of water ecosystems of urban landscapes. According to the monitoring rules sampling had to take place each 2 weeks. Detections varied as a function of land use, season of the year etc, and the nitrogen and phosphorus concentration in water and sediments. Special attention was given to the estimation of the concentration of these elements in the atmospheric precipitation in winter period. Low risk areas were granted a full sampling only for one year, while high risk areas were monitored for two years and medium for one year. Resources (financial, human etc) were covered by the savings obtained from reduced sampling frequency. Areas where nutrient concentrations are expected to be the most severe are clearly identified (see table 1 and 2

<table>
<thead>
<tr>
<th>Water ecosystems</th>
<th>(N) NH₄⁻ mg/l</th>
<th>(N) NO₂⁻ mg/l</th>
<th>(N) NO₃⁻ Mg/l</th>
<th>N min. mg/l</th>
<th>N total mg/l</th>
<th>(P) PO₄³⁻ mg/l</th>
<th>P total mg/l</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lakes</td>
<td>0,47</td>
<td>0,12</td>
<td>1,04</td>
<td>1,63</td>
<td>1,72</td>
<td>0,013</td>
<td>0,039</td>
</tr>
<tr>
<td>Superficial runoff</td>
<td>1,26</td>
<td>0,08</td>
<td>1,15</td>
<td>2,00</td>
<td>14,41</td>
<td>0,080</td>
<td>0,282</td>
</tr>
<tr>
<td>Atmospheric precipitation's</td>
<td>0,60</td>
<td>0,02</td>
<td>0,35</td>
<td>0,97</td>
<td>1,15</td>
<td>0,023</td>
<td>0,035</td>
</tr>
</tbody>
</table>

Table 1. Nutrients in the lakes and atmospheric precipitation of the case-study region. (by P. Prunici and D.Drumea)
Table 2. Accumulation of nutrients in the strata of snow of the case study region and in the natural protected area (Plaiul Fagului for the period of 10 days (by P. Prunici))

<table>
<thead>
<tr>
<th>Target areas</th>
<th>Number of samples</th>
<th>NH$_4^+$ mg/l</th>
<th>NO$_2^-$ mg/l</th>
<th>NO$_3^-$ mg/l</th>
<th>PO$_4^{3-}$ mg/l</th>
</tr>
</thead>
<tbody>
<tr>
<td>Parks</td>
<td>10</td>
<td>0,25</td>
<td>0,009</td>
<td>0,14</td>
<td>0,014</td>
</tr>
<tr>
<td>Living massive (urban area)</td>
<td>21</td>
<td>0,22</td>
<td>0,009</td>
<td>0,10</td>
<td>0,015</td>
</tr>
<tr>
<td>Industrial areas</td>
<td>10</td>
<td>0,39</td>
<td>0,021</td>
<td>0,36</td>
<td>0,024</td>
</tr>
<tr>
<td>Traffic areas</td>
<td>12</td>
<td>0,48</td>
<td>0,026</td>
<td>0,21</td>
<td>0,025</td>
</tr>
<tr>
<td>Average in the town</td>
<td>53</td>
<td>0,34</td>
<td>0,016</td>
<td>0,20</td>
<td>0,019</td>
</tr>
<tr>
<td>Average in the natural</td>
<td>5</td>
<td>0,05</td>
<td>0,000</td>
<td>0,02</td>
<td>0,000</td>
</tr>
<tr>
<td>protected area Plaiul Fagului</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The highest concentration of N-NH$_4^+$ in the liquid phase of bottom sediments was detected in the lake largely used for recreation – 6,32 mg/l, while the lowest was determined in the lake with underground feeding in a park area - 1,80 mg/l in the autumn time and maximum - 4,43 mg/l till 14,19 mg/l, in the summer period. In the lake from natural reservation Plaiul Fagului, concentration of N-NH$_4^+$ varied from 1,55 mg/l in autumn till 5,85 mg/l in summer.

Concentration of the N-NO$_3^-$ in the liquid part of bottom sediments varied from 0,21mg/l - till 1,25 mg/l in urban landscape, while in the sediments of the nature protected area it was on the level of 0,1-0,2 mg/l.

Concentration of the mineral phosphorus in the liquid phase of the sediments was in 6-20 times more than in the water and varied from 0,26 till 2,63 in urban lakes, while in the nature protected area it was on the level of 0,10 mg/l.

**CONCLUSIONS**

- urban landscapes are a serious source of pollution of water ecosystems with nutrients
- atmospheric precipitation in different parts of urban landscapes have different concentrations of nitrogen and phosphorus, which depend on the type of human activities in a certain area.
- The most important source of pollution of urban water ecosystems are areas with high density of transport units and industry
- superficial runoff formed in urban areas is a major source of nutrients to the water ecosystems
- organic part of nitrogen and phosphorus is around 70% from total amount of these elements in superficial runoff, while in atmospheric precipitation mineral forms predominate over organic ones.
- from the mineral forms of nitrogen N-N03 strongly predominates in all components of urban water ecosystems

The existing monitoring network in the case-study area has several shortcomings, which could be summarized as follows:

- There are serious overlaps in the spatial location of monitoring stations.
- Monitoring programs are poorly harmonized among different users
- Poor efforts to coordinate monitoring activities among main Institutions involved in this process
To overcome these gaps it could be useful to undertake measures aimed at the harmonization of the activities in tailoring the monitoring network in the region:

- to integrate dispersed efforts of different institutions for the harmonizing of the spatial location of the monitoring sites, parameters, methodology used etc
- to improve quality assurance practices in the laboratories and information exchange programs
- to promote independent Institutions to participate in monitoring programs, with involvement of public Institutions and NGOs
- to identify potential for integration of monitoring programs

Scientific assessment is a key element which provides a base material for the estimation of frequency of sampling, spatial location of the monitoring stations in order to estimate main features of migration and accumulation of the contaminants and their possible impact on the state of environment in the region. The efficiency is achieved by better understanding where and when sampling activity must be more frequently and the greatest uncertainty in the estimation of environmental quality exists. Knowing which factors are responsible for the deterioration of the water quality could be the key issues to tailor future monitoring programs and for addressing to the policymakers.

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NEW TOOLS FOR ECOLOGICAL WATER MANAGEMENT IN THE PROVINCE OF LIMBURG (THE NETHERLANDS)

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In the period 1998-2000 a method for systematic evaluation of the water policy in de Province of Limburg has been set up. At first the policy for the regional waters, the brooks and brook valleys, was chosen for evaluation. This policy has been laid down mainly in four documents, that were analysed precisely and combined in one structure; an objectives tree. The objectives tree presents the relations between the main objectives, subobjectives and measures. In the evaluation only the main objectives and the measures were taken into account.

Simultaneously a method has been developed for areaspecific ecological objectives for brooks. Statistical analysis was performed with biotic and environmental data of the Limburg brooks. This resulted in a series of characteristic biotic communities, called cenotypes, that are found under certain environmental conditions. The typology was completed with reference situations derived from historical data in literature. The actual and desired situation in the brooks were expressed by means of these types, together with the desired situation in two ambition levels. This gives areaspecific standards for water quality, water flow and the brook structure (cross profile, possible meanders).

The policy evaluation and the brook typology leaded to indicators for policy evaluation.

INTRODUCTION

In the recent years the Dutch water managers for regional waters, the Provinces and water boards, have developed a system with which the water policy for these waters can be evaluated: the Regional Water System Report (RWSR). A first important step in this process has been made by the Association of the Provinces of the Netherlands (IPO) by writing a Manual RWSR. An important part of this manual is formed by a ‘catalogue’ of indicators allowing standard watersystem reports. The Dutch Provinces and waterboards will work out and apply this type of report in their own region.

\textbf{Figuur 1. Province of Limburg in The Netherlands}
In addition to the RWSR-methodology, the province of Limburg (see figure 1) needed a method to formulate area specific and verifiable objectives for the regional watersystems. Most objectives in the regional water management are formulated in general terms without specifying the area of application, especially in case of the objectives for nature development. This is connected with the diversity of ecosystems, which makes the use of simple verifiable standards impossible. Without a specification of objectives a systematical evaluation of regional water policy is very difficult.

Considering this, the Limburg water managers defined an ambitious study, with the translated title: "Water target values and Regional Watersystem Explorations Province Limburg". This project has been elaborated and carried out by IWACO B.V. in co-operation with Alterra, commissioned by the Provincial authorities of Limburg and the three waterboards in this province: Zuiveringschap Limburg, Waterschap Peel en Maasvallei and Waterschap Roer en Overmaas. The project started in 1998 and was finished in November 2000.

The project was focussed on the brooks as the most important water types in Limburg, and resulted in 3 products:
1. a method for the definition of water target values;
2. a method for water system reports;
3. the application of both methods to 8 catchment areas within the province of Limburg.

The parts of the project are illustrated in figure 2.

![Figure 2. Project lay out](image)

The contents and the connections between these parts is explained here below. Before that, the position and the headlines of the regional water policy is outlined.

**PRESENT-DAY PRACTICE IN REGIONAL WATER POLICY AND MANAGEMENT**

Water policy in the Netherlands is formulated in three different plan-figures which act on a national (the state), a regional (the provinces) or on a sub-regional (the water boards) managerial level. In order to develop objectives for regional water systems national objectives are focussed to regional objectives by the provinces, which in turn are translated into sub-regional objectives and management measures (e.g. hydrological and ecological measures) by the water boards. After the first generation of sectorial water plans several provinces –including the Province of Limburg– are nowadays working on the development of an integrated environmental plan, in which water management is more or less integrated into physical, ecological and economical planning.
Important strategic objectives for the present-day regional water policy are the restoration of the ecological functioning, the restoration of the retention function, and the improvement of water quality of the regional water systems. An important ingredient is the so-called catchment area approach, which states that the entire functioning of the water system should be regarded on the scale of catchment areas (in line with the European Water Framework Directive). For the Netherlands the catchment areas on the national level are those of the rivers Rhine, Meuse, Scheldt and Eems. The regional catchment areas in the province of Limburg are those of the side streams of the River Meuse, of which the largest are the rivers Roer, Niers, Geul, Tungelroyse beek, Grote Molenbeek, Geleenbeek. The strategic regional objectives which are supposed to hold for the entire province have to be laid down and elaborated into the regional catchment areas of the side-streams of the river Meuse.

Since the various regional water systems vary with respect to their original geomorphological, physico-chemical, hydrological and ecological response characteristics and their present-day ecological functioning in relation to land-use, the possibilities of restoration and the ultimate objectives which can be set for these water systems are regionally differentiated. For the translation of these strategic objectives into more operational objectives and measurements, not only basic knowledge on the present-day ecological functioning of the regional water systems is a prerequisite, but also knowledge on the various regionally differentiated environmental variables. Moreover, the entire functioning of the water systems has to be regarded within the context of socio-economic and environmental developments. In other words, besides an analysis of the water policy, also an analysis of other policy fields, such as the physical planning and ecological environment has to be made.

Monitoring of the regional objectives is carried out by regional water boards. However, since the relation between the monitoring efforts, the realized measurements and most policy objectives is rather diffuse, by lack of regionally differentiated objectives, the policy objectives are hard to evaluate. In order to fill in these knowledge gaps and to set regionally differentiated objectives for the various regional water systems the present project has been started. The regionally differentiated policy objectives are supposed to serve as important building bricks for the catchment area orientated water management plans. It is important to note that these objectives must not be interpreted as blue prints, but more as scenarios with which the future water management can set specific, measurable, realistic and achievable objectives.

THE INSTRUMENT FOR WATER TARGET VALUES: A BROOK TYPOLOGY

The first task was the formulation of areaspecific target values, that were formulated in desired aquatic biotic communities. This part of the project is carried out by research institute Alterra, Wageningen. In order to make these target values useful for water management, it is necessary to determine under what conditions a biotic community will develop, in terms of water flow, physical and chemical water quality and the structure of the crossprofile and longitudinal profile. In this project these relations were examined by means of a statistical analysis of a large number of biotic (macro invertebrates) and abiotic data of the Limburg brooks that had been collected by the water authorities in the period 1980 - 1999.

Based on a multivariate analysis of these data characteristic types of biotic communities were identified, called cenotypes. Each cenotype was described by a.o.: name, general characteristics, typical examples in the region, the macroinvertebrate species and the abiotic characteristics. These characteristics are the most important starting point for water policy and management as they can be influenced directly.

Human influence on the water system during many decades resulted in the situation that the valuable biotic communities that could serve as target values often do not exist anymore in the present-day water systems. Therefore the optimal ecological development of brooks under undisturbed conditions up- and downstreams (reference situations) were defined using historical information and comparable natural stream ecosystems elsewhere within the Netherlands. These references were described as a cenotype in a similar way as the actual types mentioned above. Finally cenotypes were defined as intermediate stages between the ‘ideal’ references and the actual types.

The biotic communities can develop in a certain direction when conditions change; one type of community can turn into another in a period of several years. The connection between the cenotypes can be considered as a network (see figure 3).
All cenotypes in the figure are presented by a letter code (Gb, Gc). The directions of development from one to the other type are indicated with their (supposed) most important steering factor, e.g., the flow velocity of the water or the concentration of nutrients or organic matter. In practice a diversity of factors determine the development of a community, and the complete set of conditions should be regarded.

A last step in this method was to assign a certain (arbitrary) quality level to the cenotypes. Expert judgement of e.g.: the rate in which the water flow is influenced by weirs or artificial water supply, the morphological aspects cross and length profile, the chemical water quality and the present number of macrofauna species, led to a ranking in five classes, from low to very high quality. These classes will support the communication with non-experts on the subject of the water target values.

DEFINING THE CONTENTS OF THE WATER SYSTEM REPORTS

Introduction

The second part of the project concerns a method for water system reports (WSR) that must provide and regular insight in the actual situation of the water system in relation to the water policy, and indicate possible bottlenecks which can form the basis of new policy and management. This is a general definition, that is not sufficient for building a WSR; a specification is necessary.

The following delineation was made:

a) The WSR describes the situation per catchment area;
b) The WSR provides information on a strategic - and not operational - level.
c) The evaluation of individual projects is excluded.
d) The WSR refers to the actual situation of the water system and the measures that are taken in the field.
e) The Limburg WSR that was set up in this project refers to the brooks and brook valleys.
f) The WSR includes only data that are related directly to measurements or observations, vary in time and can be influenced by water management.

Water policy analysis

The Limburg water policy and regional water management has been laid down in 7 documents of which 4 are the most important: the water management plan of the Province of Limburg (Provincie Limburg, 1995), translated in 3 regional integrated water management by the waterboards (Waterschap Peel en Maasvallei en Zuiveringschap Limburg, 1997; Waterschap Roer en Overmaas en Zuiveringschap Limburg, 1993 and 1997). The provincial integrated environmental plan (Provinciaal Omgevingsplan) that is framed at this moment wasn’t available yet at the time the policy analysis was carried out.

The objectives in the plans considered were arranged in different ways so that an clear overview of all policy and management objectives was difficult. Moreover, the objectives are related to each other in an hierarchical way, that isn’t indicated clearly in the plans, but certainly is relevant. For instance, the reduction of groundwater withdrawal can serve different
objectives:
- the restoration of deteriorated terrestrial nature
- the increase of base flow in brooks
- sparingly use of water resources (as an general objective).

When the effects of the measure 'reduction of groundwater withdrawal' must be evaluated, it must be clear whether this should be done by monitoring in the nature areas, the base flow in brooks or the total water use.

As a solution to the two points mentioned above an objectives tree was framed.

**Principle of the objectives tree**

The objectives tree departs from an hierarchical order of objectives an related measures (Quade, 1985). Different levels of objectives can be distinguished, with the measures on the lowest level. An example is given in figure 4.

The objectives of the highest level are elaborated in subobjectives. E.g. the objective "Reducing surface water contamination" is translated in several subobjectives, e.g. "reduction of point sources" with the measure "connection of direct discharges to the sewer system". So the objectives tree forces the user to specify where all subobjectives and measures must lead to.

**Figure 4. Principle of an objectives tree**

**Objectives tree for Limburg**

The objectives tree for Limburg was built up starting from the plans mentioned before. For each plan all objectives are listed and given a code; in total about 750 objectives and measures. Then the objectives are selected according to the delineation rules as mentioned above.

The objectives tree was arranged in order of policy themes as far as possible. The water system report will finally be an evaluation of the complete water policy. The themes selected within the framework of this project are:
- surface water contamination;
- natural discharge regime (surface water);
- surface water structure and maintenance.

**Indicators**

Three sources were used to identify indicators: the brook typology, the objectives tree and the IPO catalogue of indicators, as mentioned in the first part of this article.

First the steering parameters resulting from the brook typology were chosen; parameters that appeared to be the most important for the change of one biotic community to the other. This is
in the Limburg situation: the concentrations of nutrients and organic matter, the flow velocity, the cross profile (form and dimensions), the longitudinal profile (degree of meandering), the degree of shadow over the brook and the presence of aquatic plants. The environmental parameters fit well in the objectives tree (see table 2).

Table 2. Relation between the policy themes and brook typology indicators

<table>
<thead>
<tr>
<th>theme</th>
<th>indicator</th>
</tr>
</thead>
<tbody>
<tr>
<td>surface water contamination</td>
<td>nutrients</td>
</tr>
<tr>
<td></td>
<td>organic matter</td>
</tr>
<tr>
<td>natural discharge distribution</td>
<td>flow velocity</td>
</tr>
<tr>
<td>surface water structure and maintenance</td>
<td>cross profile</td>
</tr>
<tr>
<td></td>
<td>longitudinal profile</td>
</tr>
<tr>
<td></td>
<td>shadow</td>
</tr>
<tr>
<td></td>
<td>aquatic plants</td>
</tr>
</tbody>
</table>

The indicators from the typology don’t cover all main objectives of the water policy for the brooks. Contamination by heavy metals and organic micropollutants couldn’t be distinguished as a critical parameter for the biotic communities, mainly because of lack of input data for the statistical analysis. The concentrations of these type of substances were added as indicators.

All indicators mentioned above are related to the main objectives of the water policy. The water system report is completed with an overview of measures taken in the report period (fixed at 4 years).

In fact, the indicators were chosen at the left and the right side of the objectives tree, and not in between (see figure 5).

![Figure 5. Position of the indicators](image)

So for the theme ‘surface water contamination’ the groundwater quality is not used as an indicator. The same applies to reduction of emission by sewer overflows. The increase of storage capacity in the sewer system is included in the WSR as a measure, whereas the volume of sewer overflows is not. The efficiency of measures for a certain objective will be subject for research, the WSR cannot be used for this purpose.

Data on the use of the water system (e.g. water withdrawals and contaminant sources) are only included in the WSR when they can be judged in relation to the water policy. For instance, the water policy contains no rules on the number of sewer overflow points in a catchment area, so including this number in the WSR isn’t useful.

Finally the indicators were compared with the indicators included in the IPO catalogue. As far as they matched, the IPO indicator was used; in other cases a new indicator was chosen.

**DESCRIPTION AND TARGET VALUES OF THE 8 CATCHMENT AREAS**

The methods for water target values and water system reports have been applied to 8 different catchment areas within the province of Limburg. The present state of the brooks was described using the cenotypes and the indicators of the water system report. A proposal for the biotic
target values (expressed in cenotypes again) has been defined by a panel of experts for the medium term - the year 2018 - and a long term - the year 2030. Leading principle was the choice of an ecological ambitious target value that is regarded as achievable within actual the policy for water, environment and spatial planning and the tempo by which changes in the water system take place.

The route from the actual state to the the target value becomes apparent for each site. By comparing the abiotic conditions required for the target values with the actual situation, the task for the water managers could clearly be indicated, and has been elaborated in types of measurements that should be taken in each catchment area. The measurements and the resulting changes in the ecological state of the flowing waters will be followed in the future water system reports.

In the near future the target values will be introduced as a starting point in elaboration of water management plans on the scale level of catchment areas or individual waters.

CONCLUSIONS

Water policy evaluation should be based on an analysis and specification of the information demand on one hand and specialist knowledge on the other. Analysis of the information demand from policy objectives gives a delineation in themes and aspects. However, objectives often aren’t formulated indicators that can be measured; the translation from objective to indicator requires specialist knowledge.

In this case the evaluation was directed to the actual ecological situation of the regional flowing waters. Two methods were used to support the framing of the evaluation: an arrangement of the objectives as laid down in the water policy and management plans using an objectives tree, and a typology of biotic communities connecting biotic target values with abiotic parameters that must be influenced for water management. Both methods complemented each other very well.

By connecting the biotic and abiotic parameters a relation has been established between desired biotic communities as final target values and measurements in water management; this clarifies choices and discussions on ambitions for ecological development, their priorities and feasibility.

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MONITORING HABITAT RESTORATION MEASURES OF NATIONAL FRESHWATER SYSTEMS: WHY AND HOW

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Water management in the Netherlands used to focus mainly on water quality, i.e. toxic agents and eutrophication. Today an important topic in Dutch integral water-management is the rehabilitation of river- and lake ecosystems. Many water systems in the Netherlands are characterised by incompleteness of the ecosystem. This is due to centuries of intensive use by man, which led to the destruction of typical habitats. Due to habitat restoration measures, abiotic conditions that were lost can be re-created and the typical habitats may recover. The effects of water policy and habitat restoration measures can be evaluated with biological monitoring data. Evaluation of water policy is the last part of the monitoring cycle. The monitoring cycle structures the process from water management goals to formulation of concrete information that is required, and subsequently the strategy on how to gain this information. Eventually the monitoring results are reported to the water management; this might lead to a change in water policy and related information requirements. The monitoring cycle starts again.

Since 1992 RIZA has an ‘official’ biological monitoring program. The monitoring cycle has been used to evaluate and improve the program. In time the biological monitoring program grew to 10 parameter groups, which give either information on water quality aspects or on ecological rehabilitation. Nowadays at RIZA the effects of ecological restoration measures are monitored by five parameters: 1. Ecotope maps are made covering all large inland water systems and showing the local hydro- and morpho-dynamics and antropogenic use. 2. The distribution of (both aquatic and bank) flora gives information on the quality of the biotopes. 3. Breeding birds are counted because they are indicative for vegetation structure types. 4. The presence of amphibian species is related to current hydrological conditions. 5. Fish monitoring data give information on the presence of spawning habitats and migration possibilities. The program is an important step towards integrated monitoring because it includes both biological and physical aspects.

INTRODUCTION

Biological monitoring in the large Dutch inland waters started in the nineteen-sixties at the Institute of Inland Water Management (RIZA). At that time, improvement of the water quality was the main topic in water management. The first monitoring activities in rivers and great lakes focussed on phytoplankton. Macro invertebrate sampling on large stones, which defended the riverbanks of the river IJssel, started in the early seventies and still continues. The same is the case for the counting of waterbirds that started at the end of the seventies. These biological time series gave valuable information on the state and development of the ecosystem in a period in which water pollution (with nutrients and toxic agents) reached its peak and measures were taken to improve water quality.

Due to international and national policy and measures, concentrations of nutrients and polluting agents decreased considerably from the 1970s up to the 1990s. This is a great achievement to which monitoring has contributed in an essential way; it supplies the facts and figures. For instance, ratios between green and blue-green algae decreased, waterplants returned and the species composition of fish improved. Despite these positive results, many typical riverine species have not returned to the water systems, due to the absence of specific habitats. Many habitats were destroyed as a result of centuries of human (governmental) interference in all Dutch water systems (lakes, rivers and tidal waters). In the 1990s, in addition to functions like water discharge, transport, fishery and recreation, nature was acknowledged as a ‘function’ of the inland fresh water systems. Ecological rehabilitation became a topic on the political agenda which led to the performance of numerous measures to restore or to rebuild habitats.

In 1996 RIZA extended its monitoring program to be able to follow the effects of ecological rehabilitation measures. Setting up an effective monitoring program is very complicated. More than once monitoring data turned out to be unsuited to evaluate measures because they did not link up directly to water management aims. Due to the differences in the level of abstraction between policy makers and people that set up a monitoring program ecological goals may not
be defined clearly, detailed description of required information may not be formulated, or assessment instruments may not be available. To avoid these problems an instrument must be used to improve the communication between water managers monitoring specialist. In this paper this method will be described and the outcome presented: a monitoring program for Dutch fresh water systems with which water policy concerning ecological rehabilitation can be evaluated.

DESCRIPTION OF DUTCH STATE WATER SYSTEMS

The inland waters on which the monitoring program focuses are large rivers, lakes and tidal waters.

The main functions of the large rivers are water discharge and shipping. Safety from flooding of the surrounding areas is the main goal of river management. Additional functions are agriculture, recreation, industrial use and nature. Due to centuries of regulation, the river became one main stream with silted up river forelands and bordered by high winter dikes. The system lacks specific riverine habitats like side channels or flood plain waters. Furthermore, the river banks are scarcely covered with vegetation. Ecosystem threats are wave attack, extreme water level fluctuations (due to efficient drainage), strong currents, high cattle-grazing intensity and bad quality of water and soil.

The main functions of the major lakes are fishery and water storage. Additional functions are shipping, drinking water, recreation and nature. Due to manipulated water levels (high in summer, low in winter) bank vegetation is not able to develop. Waves induced by wind cause severe bank erosion making circumstances for bank vegetation to establish even less favourable. The ecosystem is characterised by high nutrient concentrations.

The main function of tidal waters is shipping and water discharge; additional functions are agriculture, industry and nature. The situation of tidal waters is partially similar to that of large rivers; safety against flooding plays a major role. In the 1970s, dams were built to protect the land and inhabitants against floods. This resulted in a strong reduction of both the tide-affected area and the tidal amplitude. The typical sand and mud flats of this ecosystem eroded severely and the transition zone from salt to fresh water reduced strongly. Main ecosystem problems are the absence of the brackish water zone, the strong bank erosion and pollution.

METHOD

The monitoring cycle as described in the intermezzo, was applied to the Dutch water policy in order to design a monitoring program. Dutch water management goals serves as input to the cycle; the output was formed by the monitoring program with which the water policy concerning ecological rehabilitation could be evaluated.

Intermezzo

Monitoring cycle

The monitoring cycle (Figure 1) is an instrument that structures the monitoring process from the water policy and ecological goals to the actual measurements in the field. It facilitates the design of a monitoring program because it translates the goals to information required to evaluate these goals; subsequently concrete monitoring parameters are chosen. Good communication between water management and monitoring specialists is crucial.

In summary the successive steps in the monitoring cycle are:

1. Water management: goals are formulated concerning ecological rehabilitation.
2. Information needs; describes what information is required to evaluate the goals.
3. In the monitoring strategy monitoring parameters are determined with which the required information can be obtained.
4. In the network design-phase locations and sampling frequency are determined (what accuracy is required) and the coherence with other monitoring parameters is established.
5. Execution phase includes sample collection, laboratory analysis, data handling, data analysis and report writing. In this paper this phase will not be described.
6. Reporting and information utilisation by water managers are the phases that close the cycle. These results may form the start of a new cycle producing a new policy with a different information need.
RESULTS AND DISCUSSION

Application of the monitoring cycle on the Dutch water policy had the following results (step 4 and 5 will not be considered).

Water management. The general message of water management in the Netherlands is “to have and hold a land safe against flooding and to keep and consolidate a healthy ecosystem in which sustainable use is guaranteed” (Min. V&W, 1997). For the theme ‘ecological rehabilitation’ the following target situation is described: Natural vegetated banks give room for flora and fauna to establish and function as a migration route. Migratory fish species like Salmon and Seatrout can reach their spawning areas without obstruction. Dynamics in a water system are allowed and used to a maximum. Eutrophication problems are solved and there is space for transition zones between land and water, and fresh and salt water. There is a sustainable fish stock.

Information needs. The main subjects of the ecological target situation are ecosystem dynamics, transition zones, migratory fishes, and vegetated banks with accompanying fauna; it focuses on processes and species. The main pressures that resulted in the today poor, incomplete ecosystems are given in the description of the water systems. Habitat restoration measures focus on the reduction of these pressures; they can be subdivided into morphological measures, water level management and land use management measures like grazing, mowing etc.

Examples of morphologic measures along rivers are constructions of side channels and removal of summer dikes and surface clay layers. As a result, both hydro- and morphodynamic gradients become less steep and are extended to larger areas: there is a larger variety of abiotic conditions (current velocity, wave attack, inundation frequency, soil composition etc.). Constructions to protect the banks against wave attack are built in the major lakes. Additionally, banks are made less steep to create a gradual transition zone between land and water with large shallow water areas. Similar bank-protection constructions are built in tidal waters to protect the sand and mud flats and marshes from persistent erosion. Building fish ladders that improve the accessibility of regulated rivers for migratory fishes is also a morphological measure.

The effects of changing the current water-level management to a more natural one, is studied in the major lakes and tidal waters. In lake systems this would highly improve the conditions for settlement and development of bank vegetation. In tidal waters the increment of tidal amplitude would lead to extension of the transition zone between fresh and salt water and to a reduction of bank erosion.

Finally, diminishing the agricultural activities (like reduction of cattle density) in bank zones, highly favours the possibilities for establishment and succession of natural vegetation.

Monitoring strategy: determination of the monitoring parameters.

Water management measures are focused on the increment of hydrological and morphological gradients and the return of natural flora and fauna. Therefore, the monitoring parameters must be related to hydrological and morphological conditions, land use and species composition.

At RIZA a method is developed to monitor processes (hydrology and morphology and land use) on a landscape scale. Ecotopes are landscape units that are homogeneous in hydrologic and morphologic conditions and human influence (Wolffert, 1996). Ecotope maps of all water systems, showing the distribution and area of ecotopes, are made every eight years (Figure 2).
Aerial photographs of 1:10,000 form the basis of the ecotope maps. The photo’s are combined in a GIS with abiotic information, like elevation, inundation duration or water depth, typical for the water system. The minimum mapping unit is 50 x 50 m; for banks this corresponds with lines with a minimum length of 50 m. Morphological changes (like the construction of a side channel) and vegetation development (for instance from shrub to forest) can be surveyed on ecotope maps. In 2000, the first series of maps of inland state waters was completed.

In addition to monitoring landscape units, the quality of the habitats is monitored. Four species groups are chosen in view of either their relationship with habitat conditions or with habitat functionality (as migration route or breeding habitat).

a. Vegetation is directly related to local conditions. The floristic quality on banks of rivers, lakes and tidal waters is determined in 1 x 1 km sections along the water systems once every four years. For each distinguished ecosystem type (like marsh, meadows, flood plain forest) a list of typical plant species is made. Floristic quality is defined as completeness of the typical flora of that ecosystem.

![Ecotope map of the river IJssel; ecotopes are clustered in groups](image)

b. Monitoring the composition of the breeding bird population provides information on the functionality of the habitat. Species composition and numbers of breeding birds are related to vegetation structure and habitat area. Monitoring frequency is once every two years.

c. Fish monitoring focuses on species composition, numbers and fish length. These parameters are related to hydro- and morphodynamic conditions (habitat quality) but also migration and reproduction function of the water system. Monitoring results give information on three parts of the ecological goal: 1. Are migratory fish able to reach there spawning area? 2. Is the fish population sustainable? and 3. Are created habitats suited as spawning or nursery habitat?

d. Monitoring amphibians gives information on hydrological dynamics in river areas; some species are indicative either for low or for highly hydro-dynamic waters. Small waters along the dynamic water systems (large rivers and tidal waters) are sampled every year.

**Reporting and information utilisation.**

Apart from increasing knowledge of ecosystems, monitoring results give information on 1. the state, 2. development of the ecosystem, and it can be used 3. to evaluate water management goals. Since monitoring of ecological rehabilitation measures has started at RIZA only 4 years ago, today the main application of monitoring results is to give a description of the state of the ecosystem. After some years of monitoring the state, the second goal, description of the water system development can also be achieved. Water policy evaluation requires longer time series but also a described target and evaluation tools. For each water system a reference and target situation is described. In the Netherlands, the reference situation is not a natural, unaffected
situation, but the most natural situation under certain conditions; eg, the presence of dikes along rivers is part of the reference situation. The ecological target situation describes the most natural situation (reference situation) taking into account the functions of the water system (water discharge, fishery, shipping). At the moment two evaluation tools are used at RIZA. The first tool formulates the present, reference and target situation of the rivers Rhine and Meuse in terms of ecotope distribution (Figure 3) (Postma et al. 1996). In the target situation the percentage of forest is smaller than in the reference situation because it obstructs water discharge, increasing the risk of floods. In the second tool the reference situation is described with density of indicative species; this method is called the AMOEBE (Ten Brink & Hosper, 1989). The set of indicative species are chosen from different levels of the food web and describe together a particular ecosystem. Comparison of densities of indicative species today with the reference situation shows the success of the water policy.

Figure 3: Ecotope distribution (%) of the River Rhine (ecotopes are clustered).

CONCLUSIONS

The monitoring program that RIZA uses to evaluate ecological rehabilitation seems to be a sound program in which the chosen parameters are well related to the processes to which habitat restoration measures are directed (Figure 4). The monitoring cycle proved to be a useful instrument to design a monitoring program that collects water-policy relevant information. At the moment the monitoring program has been operational for four years and the first results are obtained. Ecotope maps of all water system are completed and an overview of the floristic quality of all water systems can be given. Bird and amphibian monitoring has started only two years ago.

The monitoring methods and first series results are now under discussion; are information users (policy makers and water managers) satisfied with the collected information, what are the main inaccuracies and is it possible to connect different parts of the program. The effects of habitat restoration measures may be evaluated after the second sampling series. The future will show how successful the program is; for now it may be stated that it is a step towards integrated monitoring in which hydrology, morphology and biology is represented.
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NEW INFORMATION TECHNOLOGIES FOR THE CREATION OF INTEGRATED SYSTEM OF RIVER BASIN MONITORING

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This paper aims to contribute to the discussion on methods of the assessment of condition of water bodies – river basin on the basis of the data integration. The main issue is assessment under conditions of uncertainty and incompleteness of information. Creation of integrated systems for monitoring is necessary for the increase of the reliability of assessment of the condition of the water bodies and the evaluation of the impact on water body.

The improvement of the reliability of the assessment is possible due to the integration of the data measurements, which were carried out within separate monitoring programs. For improvement of the reliability of the assessment are used, together with the integrated measurements data, the data base and mathematical models. For creation of the integrated system it is necessary to solve a number of scientific issues, including the issue of development of new mathematical models and new informational techniques.

Such technologies are based on calculation of the optimal values of material flow balance in technogenic and natural environment and are used for considerable improvement of quality water body condition assessment.

INTRODUCTION

In this paper considered as the methodological basis are the following levels of monitoring data integration:

Level 1. Simple combination of monitoring data characterizing the condition of water bodies. For example, integration of data for a river basin obtained under different monitoring programs. Included into this level can be integration of data gathered for transboundary water bodies. Such integration is in great demand for Ukraine and other CIS countries at which monitoring of water bodies is carried out by individual organizations according to their own monitoring programs (e.g. Hydrometeorology Service, Public Health, Environmental Protection, Municipal Economy, etc.). Similar situation exists in other countries of Europe as well. Effectiveness of such data integration is relatively low because it is based only on averaging of available data which have been collected under monitoring programs implemented by different organizations and which are not synchronized in time and space.

Level 2. Integration of data with usage of mathematical models of water bodies enables to compare the results of calculations of initial data obtained as a result of implementation of one monitoring program with the similar results obtained as a result of carrying out another monitoring program. For example, a water quality model for a stretch of a river basin is capable to integrate results of a number of measurements made at different time periods and at different points of a river basin, but it can be done only in the case when such model enables to calculate movement of water and pollutants downstream of the river, both in time and in space.

Level 3. Considerable improvement of the data reliability and results of their integration is possible when the factors influencing the water bodies are taken into account. Among such factors, mentioned should be data on the amount and composition of wastewater discharged into water bodies. Such data are usually owned by operators of wastewater treatment facilities. These data should be incorporated into the general model, which describes effluent discharge into water bodies.

Level 4. The highest level of data integration involves integration of data obtained as a result of measurement of the parameters characterizing the state of water bodies, as well as data on effluent discharge (from point sources of pollution) and available data on impact of non-point sources of pollution on water bodies. Structure of material flow balances in natural and
technogenic environment developed by Brunner-Baccini can serve as a good basis for
development of the required model of monitoring data integration.

Let us consider some aspects of the model, as well as methods and elements of data integration.
Uncertainty of the initial data, lack of sufficient volume of statistic data compels us to use non-
traditional information technologies. Such technologies use both available data bases (in the
form of statistic reports), data of direct measurements and knowledge bases (expert data). All of
it is used for development of so-called functions (basic element of a theory of fuzzy sets of Zade
type). Belonging function determines the reliability of the degree of involvement of a certain
random variable value to a given interval of numbers, that is belonging function is analogue of
the probability. For explain fuzzy set application let’s consider a set X which is a subset of Y set.
Then a fuzzy a set is considered as specified if defined as X and Y sets and an identity function
R(x) of elements x ∈ X for Y set

BASE OF MODELS

At first, let us consider an element of a model for integration of hydrological data (or material
flow data). In this simple case, water balance (material balance) for a part of the river depicting
an equality of the sums of input flows (xi) and output flows (xj) can be presented as:

\[ \sum_{i \in N_1} x_i - \sum_{j \in N_2} x_j = 0, \quad x_i, x_j \geq 0, \quad i \in N_1, \quad j \in N_2, \quad N = N_1 \cup N_2, \quad j \in N; \]  

(1)

Using available statistical material that is on the basis of available monitoring data one can
build an belonging function of the following type:

\[ R_n = f_n(x_n), \quad p_n \leq x_n \leq q_n, \quad R_n \in [0,1], \quad n \in N. \]  

(2)

Were \( p_n, q_n \) – limit of variation for \( x_n \) for example minimum/maximum boundaries of data.

The task of estimation of optimal material flow balance involves the following: finding of \( x_n, \ n \in N \), satisfying (1) and facilitating the best possible value for a minimax criteria, i.e.

\[ \min \max \left\{ R_n = f_n(x_n), \ n \in N \right\} \]

In such a way, there will be found such a balance, which satisfies the best (maximum) reliability
and is estimated at the value equal to minimum value of \( R_n \). It will to satisfy the minimax
criteria and it enable to provide the best evaluation of the monitoring data, which will meet the
laws of nature (balance) and will have maximum possible reliability at available monitoring
data.

In more complicated case, for integration of monitoring data characterizing pollution of water
body and indices of effluent quality, as well as data showing the impacts of other pollution
sources, the following fragment of a model for data integration is used:

\[ Z_k = a_{k-1}Z_{k-1} + b_k u_{k-1}, \quad k \in K, \]  

(4)

\[ p_k \leq Z_k \leq q_k, \quad g_k \leq u_k \leq h_k, \quad k \in K, \]  

(5)

For model (4,5) the following criteria is used.

\[ \min \max \left\{ R_k^Z = f_k(Z_k), R_k^U = \phi_k(U_k), \quad k \in K \right\} \]

(6)

where \( Z_k, Z_{k-1} \) are measured parameters of flows of different materials per unit of time in the
sites of river where monitoring takes place (e.g., BOD, NH4, SO2); \( u_k \) - indices of material flows
entering a water body together with effluents and from other sources; \( a_{k-1}, b_k \ldots \) - coefficients
characterizing processes taking place in a water body. If reliable information about coefficients
\( a_{k-1}, b_k \ldots \) is absent, the task of finding the best evaluation of material balance will acquire
non-linear (bilinear) nature.
INPUT DATA AND CALCULATION RESULTS

For calculation we used next sources of information official reporting data: number of population, number of animals (different type), land use square (arable lands, grass land ect.)


Expert data: specific value (average, min / maximum) for N,P – flow from population , from animals, from lands, from bottom sediment, to bottom sediments. This information used for construction belonging function in triangle form, because we have date about three points: average, minimum and maximum limits only.

EXAMPLE OF APPLICATION

Problems of European lakes eutrophication especially lakes in Finland became the base for creation and adaptation mathematical models and demonstration methods of integration monitoring data for nitrogen and phosphorus Lake Halsianjarvi (Finland) has typical structure (Fig 1.). It systems "rivers-lake" includes two in put rivers (Penninkjoki and Venetjoki) and one out river (Halsianjarvi). These systems include only three point of monitoring for measurement concentration nitrogen and phosphorus and water flow. Incompleteness information about all material flow and uncertainty information defined necessity application new information technology. Let's consider two-steps process adaptation models.

On the first step we can to create model for system "rivers-lake" in common case.

The equation of balance for system " input - output " for nitrogen and phosphorus looks like

$$\sum_{l \leq L} x_l + \sum_{k \leq K} y_k = W$$ (7)

Legend:
- $X_1$ – Flow from r. Penninkjoki; $X_2$ – Flow from r. Venetjoki; $X_3$ – Flow from bottom sediments;
- $X_4$ – Flow from atmosphere; $X_5$ - Flow from lake catchment area (arable land, grass, woods ect.)
- $X_6$ – from farm (cow, pigs ect.); $X_7$ - from animals (foxes, minks); $X_8$ – from population
- $Y_1$ – Flow to r. Halsuanjoki; $Y_2$ - Flow to atmosphere; $Y_3$ – Flow to bottom sediments; $Y_4$ – Water supply
- MR1, MR2, ML – points of monitoring

Figure 1. N,P - flow for lake Halsianjarvi
<table>
<thead>
<tr>
<th>No.</th>
<th>Sources</th>
<th>Solution %</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Penninkjoki river</td>
<td>32.9%</td>
</tr>
<tr>
<td>2</td>
<td>Venetjoki river</td>
<td>53.2%</td>
</tr>
<tr>
<td>3</td>
<td>Atmosphere</td>
<td>1.9%</td>
</tr>
<tr>
<td>4</td>
<td>Anthropogenous</td>
<td>12.0%</td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td>189 275</td>
</tr>
</tbody>
</table>

Figure 2 Input N-flow from different sources Lake Halsuanjarvi, kg

<table>
<thead>
<tr>
<th>No.</th>
<th>Sources</th>
<th>NX</th>
<th>Min</th>
<th>Average</th>
<th>Max</th>
<th>Solution %</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Flow from Penninkjoki river</td>
<td>NX1</td>
<td>51 479</td>
<td>62 747</td>
<td>71 877</td>
<td>62 265 23.3%</td>
</tr>
<tr>
<td>2</td>
<td>Flow from Venetjoki river</td>
<td>NX2</td>
<td>74 412</td>
<td>101 879</td>
<td>153 202</td>
<td>100 705 37.7%</td>
</tr>
<tr>
<td>3</td>
<td>Flow from bottom sediments</td>
<td>NX3</td>
<td>27 010</td>
<td>81 030</td>
<td>94 535</td>
<td>78 721 29.4%</td>
</tr>
<tr>
<td>4</td>
<td>Flow from atmosphere</td>
<td>NX4</td>
<td>2 960</td>
<td>3 700</td>
<td>4 440</td>
<td>3 668 1.4%</td>
</tr>
<tr>
<td>5</td>
<td>Flow from land</td>
<td>NX5</td>
<td>4 929</td>
<td>19 823</td>
<td>32 866</td>
<td>19 186 7.2%</td>
</tr>
<tr>
<td>6</td>
<td>Flow from domestic animals</td>
<td>NX6</td>
<td>1 091</td>
<td>2 182</td>
<td>3 272</td>
<td>2 135 0.8%</td>
</tr>
<tr>
<td>7</td>
<td>Flow from wild animals</td>
<td>NX7</td>
<td>342</td>
<td>684</td>
<td>1 026</td>
<td>669 0.3%</td>
</tr>
<tr>
<td>8</td>
<td>Flow from population</td>
<td>NX8</td>
<td>550</td>
<td>650</td>
<td>700</td>
<td>646 0.2%</td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td></td>
<td>162 773</td>
<td>272 695</td>
<td>361 918</td>
<td>267 995</td>
</tr>
</tbody>
</table>

Table 1. Input-output N-flow, Lake Halsuanjarvi, kg/year

| Disbalance – Discrepancy | Z  | 8 619 | 8 983 | -17 243 | 0    |
| Reliability             | R  | 0.95725|

Table 1. Input-output N-flow, Lake Halsuanjarvi, kg/year
**Input**

<table>
<thead>
<tr>
<th>No. Sources</th>
<th>PX</th>
<th>Min</th>
<th>Average</th>
<th>Max</th>
<th>dx</th>
<th>ex</th>
<th>fx</th>
<th>Solution</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow from Penninkjoki river</td>
<td>PX1</td>
<td>2 137</td>
<td>3 885</td>
<td>4 857</td>
<td>3 788</td>
<td></td>
<td></td>
<td>16.2%</td>
<td></td>
</tr>
<tr>
<td>Flow from Venetjoki river</td>
<td>PX2</td>
<td>4 377</td>
<td>6 456</td>
<td>11 490</td>
<td>6 341</td>
<td></td>
<td></td>
<td>27.0%</td>
<td></td>
</tr>
<tr>
<td>Flow from bottom</td>
<td>PX3</td>
<td>2 701</td>
<td>8 103</td>
<td>27 010</td>
<td>7 802</td>
<td></td>
<td></td>
<td>33.3%</td>
<td></td>
</tr>
<tr>
<td>Flow from atmosphere</td>
<td>PX4</td>
<td>296</td>
<td>370</td>
<td>444</td>
<td>366</td>
<td></td>
<td></td>
<td>1.6%</td>
<td></td>
</tr>
<tr>
<td>Flow from land</td>
<td>PX5</td>
<td>1 328</td>
<td>4 483</td>
<td>5 389</td>
<td>4 307</td>
<td></td>
<td></td>
<td>18.4%</td>
<td></td>
</tr>
<tr>
<td>Flow from domestic animals</td>
<td>PX6</td>
<td>273</td>
<td>545</td>
<td>818</td>
<td>530</td>
<td></td>
<td></td>
<td>2.3%</td>
<td></td>
</tr>
<tr>
<td>Flow from wild animals</td>
<td>PX7</td>
<td>86</td>
<td>171</td>
<td>257</td>
<td>166</td>
<td></td>
<td></td>
<td>0.7%</td>
<td></td>
</tr>
<tr>
<td>Flow from population</td>
<td>PX8</td>
<td>120</td>
<td>150</td>
<td>180</td>
<td>148</td>
<td></td>
<td></td>
<td>0.6%</td>
<td></td>
</tr>
<tr>
<td>Total</td>
<td></td>
<td>11 317</td>
<td>24 164</td>
<td>50 444</td>
<td>23 448</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Output**

<table>
<thead>
<tr>
<th>No. Sources</th>
<th>PX</th>
<th>Min</th>
<th>Average</th>
<th>Max</th>
<th>dx</th>
<th>ex</th>
<th>fx</th>
<th>Solution</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow to Halsuanjoki</td>
<td>PY1</td>
<td>6 274</td>
<td>12 764</td>
<td>24 879</td>
<td>13 439</td>
<td></td>
<td></td>
<td>57.3%</td>
<td></td>
</tr>
<tr>
<td>Flow to atmosphere</td>
<td>PY2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
<td></td>
<td>0.0%</td>
<td></td>
</tr>
<tr>
<td>Flow to bottom sediments</td>
<td>PY3</td>
<td>3 241</td>
<td>9 724</td>
<td>14 856</td>
<td>10 009</td>
<td></td>
<td></td>
<td>42.7%</td>
<td></td>
</tr>
<tr>
<td>Flow to water supply</td>
<td>PY4</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
<td></td>
<td>0.0%</td>
<td></td>
</tr>
<tr>
<td>Total</td>
<td></td>
<td>9 515</td>
<td>22 487</td>
<td>39 734</td>
<td>23 448</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Disbalance – Discrepancy**

<table>
<thead>
<tr>
<th>Z</th>
<th>1 802</th>
<th>1 676</th>
<th>10 710</th>
<th>0</th>
</tr>
</thead>
</table>

**Reliability**

| R | 0.9443 |

Table-2. Input-output P-flow  Lake Halsuanjarvi, kg/year

Where $x_l$ - entering flows for everyone $l$ - source of input substances; $y_k$ - leaving flows for each $k$-direction; W-discrepancy between an input and output, which should be equal to zero ($W=0$) in case of complete balance "input - output".

Additional equation for measured values looks like:

$$x_l - m_l = q_{bl}, l=1,L$$  

$$y_k - n_k = s_{yk}, k=1,K$$  

$$\alpha_l \leq x_l \leq s_l, cx_l$$  

$$dy_k \leq y_k \leq fy_k$$

Where $m_l, n_k$ - data of monitoring, $q_{bl}, s_{yk}$ - discrepancies, $ax_l, cx_l, dy_k, fy_k$ - minimal and maximal allowable values of volumes of substances flows. The general task setting consists in the following. To find the decisions of system (7-11), so to satisfy conditions of balance (7), i.e. $W=0$. Except for that discrepancies values between the monitoring data $m_l, n_k$ and accounting values of substance flows should be minimal.

**On the second step** we create fuzzy set model (FS-model) for system "river-lake". On the basis of monitoring data the belonging functions can be build.

$$R_{xl} = f_l (x_l) \in [0,1]; R_{yk} = f_k (y_k) \in [0,1]$$

These functions define a degree of belonging of number $x_l$ to a subset interval $X_l$ and number $y_k$ to a subset interval $Y_k$. Simultaneously these functions define reliability of accounting values of balance.

FS-model looks like:

$$\min \max \{R_{xl} = f_l (x_l), l \in L, R_{yk} = f_k (y_k), k \in K \}$$

$$\sum_{l \in L} x_l + \sum_{k \in K} y_k = 0$$
In practice this means, that the minimal values of reliability of an estimation of balance are maximal (best) of all possible variants of accounts.

In the elementary case $R_{xl}$, $R_{yk}$ looks like triangles, so at $R_{xl} = 1$, $R_{yk} = 1$, $x_l$, $y_k$ are equal to average values of monitoring data, at $R_{xl} = 0$, $R_{yk} = 0$ $x_l$, $y_k$ are equal to the minimal or maximal values.

Results of calculation demonstrated on the Table 1,2 and Figure 2.

As result we calculated assessment of complete balance N-P-flow in condition on uncertainty information. We used different type in put information for mathematical models and calculation optimized assessment of N-P-flow and condition of water body.

REFERENCES


ECOLOGICAL MONITORING FOR WATER BODY MANAGEMENT

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The biomonitoring methods commonly used in the aquatic environment (biotic and diversity indices, indicator species) are briefly illustrated and their drawbacks discussed. Although biomonitoring is limited to some select biological compartments, without taking into account the variables of the physical environment, it is useful as a "warning signal" of environmental changes. On the other hand, biological monitoring cannot be used as a basis for deciding on the best way of managing an environment to reach the specific goals of the conservation of the environment and its uses. A kind of monitoring is required which can provide the information necessary for the development of a programme of management. To this end, what might be called "ecological monitoring" must take into consideration biological as well as chemical and physical variables. In addition, when necessary, both the water body and its watershed must be considered. Ecological monitoring must be adapted to specific problems of environmental management; as a consequence, there is no universal type of monitoring technique, but many types according to the characteristics of the environment and the goals to be reached.

Key words: saprobic scale, biotic and diversity indices, concentration factor, pollutant capacity of the environment

INTRODUCTION

Chemical and biological monitoring methods have been applied to water bodies to estimate their level of degradation deriving from anthropogenic impact (e.g. disposal of excessive amounts of nutrient and toxic substances) and natural events. Chemical monitoring can identify the potential causes of biological effects and the consequent changes to the ecosystem, but gives no information on the actual damage to the biota. Conversely, biological monitoring may evaluate the actual effects on the biota, produced by changes to the environment, but without providing information on their causes whether anthropogenic or natural (Ravera, 1998). These considerations suggest that there is an evident advantage in applying both chemical and biological monitoring to the same environment at the same time (Cairns, 1995). This combined monitoring has been applied with some success in Ireland (Flanagan and Toner, 1972) and in Switzerland (Perret, 1977).

On the other hand, even combined biological and chemical monitoring is of limited usefulness in planning the management of water bodies. The ecosystem to be managed must be monitored by using selected variables (physical, chemical and biological) to show its degree of wellbeing and its variations over time. If the monitoring is correct, the results obtained may be used to identify the type of management best suited for the solution of specific environmental problems. Since the water body is greatly influenced by its watershed, with which it forms a unit, the energy and/or the material loads from the watershed into the water body must be considered. This paper aims a) to discuss the advantages and disadvantages of the biomonitoring methods commonly used b) to propose a method of ecological monitoring designed for the management of aquatic ecosystems.

BIOMONITORING

Because there is no ideal biomonitoring method, a great variety of techniques have been developed. According to Hynes (1994), continuing to search for some universal indices, the reality of which is at least questionable, is a waste of resources.

Kolkwitz and Marsson (1908, 1909) and Kolkwitz (1950) developed a biomonitoring method they called "Saprobiensystem", which is based on the structure of microorganism associations (e.g. bacteria, fungi, protozoa, algae) in relation to the level of sewage pollution in the running waters where they live. As a result, running waters may be classified according to their pollution level and ordered on a scale based on biological analyses. The "Saprobiensystem"
method has been used in an increasing number of geographical areas. The technique has over
time undergone many modifications (e.g. Fjerdingstad, 1964; 1965; Rothschein, 1962;
Sládecek, 1971; 1972; 1973) and was adopted by several agencies and administrations,
especially in Central and Eastern Europe.

Richardson (1928) was the first to evaluate the water quality of running waters (e.g. Illinois
River) on the basis of the different degree of sensitivity of some benthic macroinvertebrate taxa
to pollutants and especially to degradable organic matter (sewage). This technique has been
successively modified by a number of scientists (e.g. Woodiwiss, 1964; Graham, 1965;
De Pauw and Vanhooren, 1983) giving rise to many "Biotic indices" (Washington, 1984; De
Pauw et al., 1991). These methods are based on the presence (or absence) of some taxa
considered at various taxonomical levels such as species, genus, family and some also on their
relative abundance (e.g. Chandler, 1970).

The technique of the biotic indices, based on macroinvertebrates, is generally limited to
shallow running waters, commonly with a stony bottom. This type of monitoring may be
extended to deep rivers with muddy sediments by temporarily laying artificial substrates in the
segment of river to be monitored. Macroinvertebrates which have colonised the substrates are
collected after a set time and identified to calculate the value of the biotic index. To follow the
seasonal variations of the index, the macroinvertebrates must be collected several times during
the year.

Many taxonomic groups with a relatively high number of species have been used as indicators
of pollution; for example, Oligochaeta (Brinkhurst, 1966; Rosso et al., 1994; Prygiel et al.,
2000); Trichoptera (Schumacher and Schremmer, 1970; Bazzanti and Bambacino, 1987);
Diptera (Rivosecchi, 1973); aquatic macrophytes (Seddon, 1972; Schmedtje and Kohlmann,
1987); algae (Nygaard, 1949; Kam, 1986; Somashakar, 1988); Diatomeae (Besch et al., 1972;

The index of biotic integrity (IBI) takes in to account a series of characteristics of the structure
and function of fish communities living in streams and small rivers (Karr, 1981; Karr et al.,
1986). The index, applied with success by Simon and Lyons (1995) to warm water fish
populations, was modified to be applied to dunal palustrine wetland (Simon, 1998) and to
lagoon fish communities (Simon and Stewart, 1998).

Another method consists in measuring the survival of a group of animals (e.g. molluscs, fish)
kept for an appropriate time in cages submerged in different zones of a polluted water body.
This technique may be useful for mapping a polluted area and measuring the polluted gradient
along a water course. The same technique may be used as an "alarm test".

Measuring biological diversity is another method used for monitoring environmental pollution
(Magurran, 1988). Diversity has been used by several authors as a powerful indicator of the
wellbeing of the ecosystem (e.g. Bechtel and Copeland, 1970; Egloff and Brakel, 1973; Patrick,
1973; Shafer, 1973; Rosenberg, 1976; Wu, 1982; Shaw et al., 1983). This method is based on
the observation that in a polluted environment the diversity value is generally lower than it is in
a non polluted one. In fact, pollution eliminates a number of the species which are most
sensitive to this stress and this number increases with the increasing pollution level; as a result,
the diversity of the community decreases.

From the various monitoring methods developed to test the effects of pollutants on the
enzymatic system and other biochemical variables it is very difficult to predict the ultimate
consequences on the populations to which the material tested belongs (Gill et al., 1991;
Kufesad et al., 1994; Sternberg et al., 1996; Nascimento et al., 1998; Soto-Galera et al., 1998).
Standardising, or at least harmonising, the various biomonitoring methods should make
comparison of the results easier. Attempts to do this have not had much success, because of the
great variety of environments, biogeographical differences and other more or less scientific
reasons. The Commission of the European Communities (C.E.C.) sponsored three campaigns to
monitor some English, German and Italian running water environments by the various
biomonitoring methods commonly used in European countries (Tittizer, 1975; Woodiwiss,
1978; Ghetti, 1979; Ghetti and Bonazzi, 1980). The aim of these campaigns was the
harmonisation of the various biomonitoring techniques, that is, the development of a common
scale to compare the results obtained by using different methods. The studies achieved only
limited success.
The above monitoring methods are based on the biological changes induced by pollution. In contrast, monitoring through "accumulator organisms" ("scavengers") is based on the fact that some taxa, which are very resistant to certain pollutants, can accumulate these pollutants in their body at concentrations far higher than those present in their environment, without apparent damage. These bioindicators were initially used to monitor radioactive contamination of the environment and subsequently also to monitor non-radioactive pollutants such as heavy metals, organic micropollutants and hydrocarbons (e.g. Ravera and Riccardi, 1997; Ravera, 1998). The pollution level of the environment for a certain pollutant is evaluated by analysing the pollutant content in the organism.

The results of biomonitoring, commonly those of the "Saprobiensystem" or "Biotic indices", are often used to represent on a map the state of pollution of the hydrological network of a region and, if monitoring is repeated, its variations over time. With this aim a colour is assigned to each pollution level, corresponding to each type of biological association. On the map the segments of the water courses with the same pollution level are labelled with the same colour. The pollution map shows the general view of the state of wellbeing of the water resources of a region. Its practical value is strictly dependent on the accuracy and frequency of the sampling plan and the quality of the analyses.

THE MAJOR DRAWBACKS OF BIOMONITORING

The most important drawbacks of "biotic indices" based on macroinvertebrates are the following: a) the assumption that a species is an entity stable over time and space, and consequently that the populations of a species cannot be modified by adaptation and selection mechanisms; b) attributing the same sensitivity to the pollutants to all the species belonging to the same genus or family considered by the protocol; c) the influence of the physical environment on the relationship between the pollution level and the biotic characteristics is not taken in to account and d) the method must be restricted to shallow running waters polluted by degradable organic substances and not to any kind of pollutant (e.g. Washington, 1984). Some considerations on these drawbacks follow.

The presence of a species in a given environment is evidence: a) that it has developed such positive (e.g. symbiosis, cooperation) and negative relationships (e.g. predation, parasitism, competition) with other species of the community as permit its survivorship and reproduction and b) that the characteristics of the physical environment are tolerated by the species. As a consequence, the species may be considered an indicator of the environmental conditions. Conversely, the absence of a species may be the consequence of other causes, apart from pollution; for example, the unsuitable conditions of the physical environment, the presence of stronger antagonistic species (e.g. competitors, predators, parasites), or the extinction of the species due to later powerful immigrants.

The assumption that there is always a relationship between the presence (or absence) of some taxa and the pollution level of the environment must also be made with caution, which is justified by the following considerations. The microbial strains resistant to antibiotics and those of insects not influenced by biocides, demonstrate that some resistant strains to certain pollutants may be selected in a relatively short time. Recent studies present evidence that natural populations of aquatic organisms may acclimate to various toxic metals (e.g. Stubblefield et al., 1999; Millward and Grant, 2000). As a consequence, different populations of the same species may show different degrees of sensitivity to the same pollutant.

If there is a certain variability within the same species, a far wider variability among species belonging to the same genus, family or class may be expected; for this reason the taxa higher than genus and also species level (such as those used for some indices) have a rather doubtful value as indicators (e.g. Chandler, 1970). This is strongly supported by the use of biomonitoring methods based on the different degrees of sensitivity of species belonging to the same taxonomic group (e.g. Trichoptera, fish). According to Hilsenhoff (1977), identification to species level increases the sensitivity of the monitoring techniques, and it is always necessary in detecting water bodies with low pollution levels. This is one of the reasons why Graham’s index has poorer sensitivity than the poor sensitivity of Woodiwiss’s index (Balloch et al., 1976). It is well known that the same pollutant concentration produces different effects on the same species under different environmental conditions. For example, the sensitivity of a given species to several pollutants decreases with temperature; and the availability of heavy metals to fish
(except the detritus feeders) decreases as the concentration of suspended matter increases. This is due to the adsorption of the metal onto the suspended particles, which sedimenting, cause the decrease of the metal concentration in the water column. As a consequence, applying the same biotic index to communities living in environments with the same pollutant level but with different natural characteristics, may give very different results. It is therefore evident that the value of the biotic index is rather poor if the environmental conditions are not considered. Although the biotic indices have been generally developed for evaluating pollution by organic matter, such as sewage (e.g. Washington, 1984), in practice they are applied, without any modification, to environments contaminated by metals, pesticides, oil and other pollutants. This is due to the erroneous belief that some taxa are sensitive to any stressor. This inappropriate use of biotic indices may produce anomalous results. For example, Ephemeroptera, Plecoptera and Trichoptera are generally very sensitive to degradable organic matter and their decomposition products, whereas red-chironomids and other Diptera are not. On the basis of this observation these taxa were included in several biotic indices (e.g. the Woodiwiss, Graham and Chandler indices). The response to other types of pollutants (e.g. heavy metals) by the same taxa are different (e.g. Clements et al., 1992; Clements, 1994).

There is another important drawback with the "Biotic index" based on the use of artificial substrates. This method consists in recording information on several species which do not naturally live in the soft sediment and have drifted from farther upstream, a zone generally characterised by a stony bottom. This means that the index is calculated on species which usually do not live in the river segment considered and are thus not reliable indicators (Chutter, 1972).

The submerged cage method is generally a short time test (few hours), is very simple to apply and taxonomical experience is not required because the species used are chosen by the researcher. This method has two disadvantages: a) generally the test is carried out on one species, rarely on two or three; and b) the animals used must be transferred from another environment. Consequently, it is doubtful that one or two species may be sensitive to all types of pollutants (e.g. Cairns and Mount, 1990) and there is not sufficient time to adapt the specimens to the new environment.

The major advantages of using diversity as a biomonitoring method are the following: a) diversity is a natural characteristic of the community and one of the most important topics of basic and applied ecological research; b) the degree of diversity, which expresses the community complexity, independently of the characteristics of the composing species, may be quantified by one number and c) since the diversity measure may be applied to any community (or a part of it, such as macroinvertebrates, fish, plankton) and environments (e.g. streams, rivers, lakes, estuaries), the diversity of very different communities may be easily compared. In spite of these advantages, some authors do not believe that diversity is a useful method for evaluating the pollution level of the environment. For example, Washington (1984) stated that diversity level is a measure of community structure and not an indicator of environmental pollution. Hynes (1994) does not believe that any numerical index (e.g. saprobic scale, biotic indices and diversity indices) can produce results which are useful for evaluating the pollution level of the environment. In addition, this author affirms that diversity indices may indicate only differences between stations over distance or time, and enable upstream to be compared with downstream, or one place in different years or seasons. Conversely, many authors use the diversity index as a powerful measure of environmental wellbeing on the assumption that pollution decreases the diversity level of any community. To this rule, called by Hynes (1994) "a current dogma", there are some exceptions if in calculating the diversity index, in addition to the number of species (richness), the distribution of the individuals among the species (evenness) is taken into account. For example, when a small number of dominant species are eliminated because of their high sensitivity to the pollution, several rare species, more resistant to this kind of stress, may take advantage of the decreased competition and increase in number. As a result, the small decrease in richness may be widely counterbalanced by the great evenness and the degree of diversity increases.

As with "Biotic indices" the search for new diversity indices is still going on. Unfortunately, there is a very limited number of studies dealing with comparison among the various indices and their ecological importance (Washington, 1984). Selecting the best diversity index for a specific purpose is not very easy, but the main errors are associated with sampling (e.g. sample size, sampling frequency in relation to the biological cycle of the species studied). Another major error is the acritical use of the diversity value without taking its meaning into account.
For example, the highest diversity value of the benthic community from an area of the Venice lagoon was calculated for the samples collected during the anoxic period. The high diversity measured in this period was due to the effect of the greater decrease of the individuals than that of the species number (Ravera, 1995).

ECOLOGICAL MONITORING

In spite of the drawbacks of biological monitoring, this technique may be useful for acquiring preliminary information on the state of the environment and its evolution over time, on the basis of the biological effects produced by both natural and anthropogenic stress. The discriminant power of diversity indices is generally higher than that of biotic indices. On the other hand, because biotic as well as diversity indices are applied exclusively to the biota without taking into account either the physical environment or the fundamental influence of the watershed on the water body, they are of limited usefulness in planning the management of aquatic environments.

There is a need for a method of monitoring which can identify the most appropriate way of managing an aquatic ecosystem to reach the goals set, and which can evaluate the actual results obtained in relation to those expected. These goals are ecosystem conservation and the uses of the water body, among which the most important are: supply of water for drinking, irrigation and industry, production of electricity, and the provision of suitable environmental conditions for fishing, aquiculture, recreation and tourism.

This type of monitoring may be called "ecological" because it takes into account biological as well as physical and chemical variables. Ecological monitoring may take into consideration the water body and its watershed, which is the main source of nutrients and pollutants. There is no universal ecological monitoring with a rigid protocol, and the variables to be measured depend on the characteristics of the ecosystem and the ultimate aim which management is designed to achieve.

For each problem, monitoring must be carefully planned to prevent too onerous a programme of sampling and analyses. With this aim in mind, the "critical compartments" (biotic and/or abiotic), the "critical seasons" and the "critical sites" must be identified. For example, in a stream contaminated by heavy metals the "critical compartments" are the macrophytes, the sediments and the bivalve molluscs, because these compartments concentrate most of the metals. Benthic populations are the critical compartments for monitoring running waters, while the quality and quantity of phytoplankton and zooplankton are the critical compartments in standing waters. The "critical seasons" for running waters correspond to the periods of the maximum and minimum flow-rates, and for lakes to the periods of full-circulation and stratification. For streams and rivers the "critical sites" are the collecting stations upstream and downstream of the point source of the pollutants; for lakes the critical site will be the area of maximum depth, if the lake is medium or small in size, but if it is large and divided into several basins, each basin must be separately investigated. In a deep lake contaminated by organochlorine biocides, the "compartments" are the fish (fatty tissue) and crustacean zooplankton, the "seasons" correspond to the period of full-circulation and stratification and the "site" may be the sampling station in the area of the maximum depth.

Some examples may clarify the use of ecological monitoring.

- A preliminary evaluation of the trophic level of a lake may be made by measuring two variables: dissolved oxygen (DO) concentration and temperature. During the stratification of the water, DO concentration values which are too high in the epilimnion and too low, down to zero, in the hypolimnion are a good indicator of high trophy. If, at the end of the stratification, the DO concentration in the metalimnion is lower than that in the upper layer of the hypolimnion, the lake can definitely be classified as eutrophic. If during the full circulation the DO concentration in the epilimnion is very low (< 4 mg O₂ l⁻¹) the situation is critical and mass mortality of fish may be expected (e.g. Ravera et al., 1986). In addition, the seasonal variations in water transparency may be a good indicator of algal bloom distribution over time and thus of trophic state of the lake. These data can form the basis for planning a chemical monitoring of nutrient substances and chlorophyll.

- DO analysis allows the eutrophication trend of a lake to be followed. For example, from 1946 to 1970 at the end of the summer the DO concentration in Lake Lugano at 50 m
depth gradually decreased from 7.4 mg DO l\(^{-1}\) to 2.7 mg. Significantly, during the same period of time there was a steady increase in the number of inhabitants in the lake watershed (Ravera, 1973). In figure 1 this relationship is schematised.

Before planning the management of a polluted stream, the flow-rate and current speed and their seasonal variations must be known. In addition, the chemical monitoring must be planned following an inquiry based on an inventory of the pollutant sources in the watershed (e.g. mining, industry, agriculture), to select the most important pollutants to be analysed; in this context the pollutant accumulator organisms ("scavengers") may also be utilised.

![Figure 1 - Relationship between the number of inhabitants in the watershed of Lake Lugano and the oxygen concentration at 50 m depth in the lake water (Jaag and Märki, 1970; Ravera, 1983).](image1)

![Figure 2 - Phosphorus and nitrogen mass-balance estimated in the eutrophic Lake Lugano (Ravera, 1983). The values are expressed in metric tons.](image2)
The management of a eutrophic lake requires quantitative information on the nutrient mass balance, i.e. the nutrient input and output in a time unit. To estimate the nutrient output, chemical monitoring of the outlet must be applied and the water flow-rate of the emissary known. The nutrient input may be defined by two methods: a) chemical monitoring of the tributaries to establish their nutrient concentrations, followed by calculation of the nutrient loads on their flow-rate; b) an inquiry to estimate the nutrient sources in the watershed. Because each method has its own advantages and disadvantages (Ravera, 1980), it is advisable to adopt both and compare the results which emerge. In addition, the nutrient load from atmospheric depositions must be taken into account. The nutrient loads to the lake may then be estimated. The amount of nutrients retained in the lake is the difference between the nutrient load to the lake and that of the output. An example of the nutrient mass-balance estimated in a eutrophic lake is provided by the results of a study carried out in 1973 on Lake Lugano, a deep water body on the border between Switzerland and Italy (Figure 2). The mean annual nutrient load amounted to 1306 t TN and 130 t TP and the output to 802 t TN and 57 t TP. The amounts of nutrients annually retained in the lake were 504 t TN and 73 t NP. Both of the above methods (chemical monitoring and inquiry) were used. The order of magnitude of the data emerging from the inquiry was in good agreement with those from the chemical monitoring (Ravera, 1983). The same method may be applied for any pollutant which has no volatile forms; for example, oil.

One of the most crucial problems in the management of a water body is that of predicting the maximum load of a noxious substance which may enter an aquatic environment without its concentration in the water producing undesirable effects. An answer to this problem is provided by Vollenweider’s model (1968; 1975; 1979) as regards lake eutrophication, and the concept of “Radiological capacity of the environment” formulated by the C.E.C. (Recht, 1969) as regards radioactive contamination.

Vollenweider’s model is focused on the ratio between the mean depth of the lake and its hydrological renewal time and the annual phosphorus load to the lake. This provides a basis for predicting the reduction of phosphorus load needed to recover the eutrophicated lake. With this aim, monitoring essentially consists in evaluating the phosphorus load from the watershed and analysing the total phosphorus in the lake waters. These data are necessary to identify the most appropriate type of management which will lead to the recovery of the lake; for example, waste water treatment, diversion of the waste waters outside the watershed (Figure 3).

![Figure 3 - U.S.-OECD data applied to modified Vollenweider phosphorus loading diagram (redrawn from Rast and Lee, 1978).](image-url)
During the sixties, to minimise the risk of environmental contamination by the effluents from nuclear power plants, the C.E.C. developed the concept of "radiological capacity", i.e. the amount of radioactive material that may be disposed in an aquatic system without damaging the ecosystem and its resources (e.g. water, fish). For this information to be available, the water body must be periodically monitored by radioactivity analyses on the water and other selected biological (e.g. aquatic plants, fish) and abiotic (e.g. sediments) compartments. The data obtained form the basis for estimating a relationship between the radioactive load from the power plant effluents and the radioactivity in the water body. In addition, the "concentration factor" (C.F.) for each radioisotope of each important species (e.g. fish) may be calculated, that is, the ratio between the radioisotope concentration in an organism and that in the water. This type of monitoring may also be applied to any aquatic ecosystem contaminated by toxic substances (e.g. heavy metals, organic micropollutants) with the "radiological capacity" being changed to "pollutant capacity" of the environment.

**DISCUSSION AND CONCLUSIONS**

According to Washington (1984), biotic indices must be used exclusively to evaluate the state of streams polluted by easily degradable organic matter (e.g. sewage) and not for all types of pollutant, whereas diversity and similarity indices are indicators of community structure and not of environmental pollution. Hynes (1994) stated that neither the biotic nor the diversity indices can indicate the pollution level of the environment. On the other hand, it is unbelievable that an important change in the physical environment (due to pollution or other anthropogenic or natural causes) should have no effect on the biota. As a consequence, biotic and diversity indices used for monitoring the aquatic ecosystem, while they are not specific pollution indicators, reflect the state of the biota and, consequently, that of the physical environment. The absolute value of these indices is less important than their variations over time, because they may have the function of a "warning signal" if a major change appears in the biota. In conclusion, in spite of the drawbacks discussed in this paper, biotic as well as diversity indices, if used with caution, may be useful in evaluating the state of the environment and its evolution over time.

The management of an aquatic ecosystem is designed to yield well defined practical results, such as environmental conservation and availability of resources. To this end there is a need for information on the biotic and abiotic variables relating to the goals of the management. These variables cannot involve exclusively one type of parameter, for example biological or chemical. The functioning of the ecosystem, and its decline and recovery, is the result of the interaction of physical, chemical and biological compartments. The need for an integrated approach to identify the most appropriate management for reaching set goals is justified by the great complexity of the ecosystem (e.g. Chandler, 1970; Burton, 1999) and the multiple uses of the aquatic environment.

This concept is supported, for example, by the "Trophic State Index" (T.S.I.) developed by Carlson (1977), which is based on three variables: biological (chlorophyll-a concentration), physical (water transparency) and chemical (phosphorus concentration). This index has been successfully adopted to evaluate the trophic degree of the lakes (e.g. de Bernardi et al., 1984; 1985).

Another useful trophic index (TRIX), developed by Vollenweider et al. (1998) has been used to classify the trophic level of the marine coastal zone. This index is based on the most important variables connected with the causes (phosphorus and nitrogen concentrations) and the effects (oxygen and chlorophyll concentrations) of the ecosystem trophic evolution.

In addition, management of a water body often involves acquiring information on the watershed to evaluate the water, nutrients and toxic pollutant loads. As a consequence, biological monitoring, consisting of biotic and diversity indices, is not sufficient to produce useful information for planning a management programme.

There is a need for "ecological monitoring" which cannot be confined to the rules of a rigid protocol, because it must vary with the characteristics of the ecosystem and the aims of the management. This type of monitoring may take account of biological as well as chemical and physical variables, and may also involve the watershed of the ecosystem to be managed. Ecological monitoring, although based on ecological concepts, is not an academic research, but a pragmatic technique confined to measuring selected biotic and abiotic variables which...
highlight some important processes for planning a given management in relation to specific practical goals. Ecological monitoring is also useful for evaluating the results obtained from applied management. In conclusion, this kind of monitoring is designed for the acquisition of an adequate knowledge of the environment to be managed, and to prevent too onerous a programme of analyses.

ACKNOWLEDGEMENTS

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REFERENCES


Investigations have been carried out to set up the tool called Water-SEQ (System for Evaluation of the Quality of rivers, with the three components of ecological quality as described in water framework directive: (1) Water quality : Water-SEQ, (SEQ-Eau in french) (2) Biological quality : Bio-SEQ, (SEQ-Bio in french) (3) Hydro-morphological quality : Physical-SEQ, (SEQ-Physique in french)) that allows the assessment of the physico-chemical quality of river water, in order to provide a comprehensive image of river quality, in accordance with the European Water Framework Directive. Thus, Water-SEQ reports the suitability of water to satisfy water uses and to allow aquatic life in the river. The suitability of water is assessed with five suitability classes represented by five colors : blue, green, yellow, orange and red. It also reports the water quality with quality classes and indexes, which can be compared to the quality required, identifying the main impact on water quality. This tool is adaptable thanks to its modular structure and can be adjusted when the regulations are updated. As it is able to answer many different questions concerning water quality, it is in fact a set of tools.

INTRODUCTION

Since 1971, river quality has been assessed in France using a system of 5 quality classes defined by threshold for certain physico-chemical and hydrobiological parameters. This classification system allowed a limited assessment of water suitability for water uses and functions and for decision-making in the monitoring and the planning of the protection of river quality. The French water agencies and the French minister for environment have decided to collaborate and improve the system. It has resulted in the assessment of river quality based on the water component (Water-SEQ), the physical environment (Physical-SEQ) and the biological component (Bio-SEQ). The three components of the river quality assessment system, consistent with the future European Framework Water Directive, are common for all water partners in France. This paper succinctly describes the Water-SEQ. Further details on this system are found in the document published by the French water agencies and the French minister for environment written in English and in French (Anonymous 1999).

Water-SEQ reports the suitability of water to satisfy water uses and aquatic life but also aims at comparing the result of the assessment to the quality required, and identifying specific parameter responsible for a reduced quality rating. Furthermore, Water-SEQ helps to define a restoration objective for each indicator, and reports the efficiency of the different policies to restore water quality through the recording of class and index values. In addition, it allows communication with the decision-makers and the public.

METHODS

Two basic features concerning water quality and indicators

To understand how Water-SEQ has been built, we must recognise that, when people speak of water quality, they can have in mind many different points of view, such as for example:

→ water suitability for aquatic life,
→ water suitability for making drinking water,
→ water suitability for leisure, especially bathing,
→ water suitability for irrigation.

There will be a different answer for each of these points of view and each of them can be correct.

For example nitrates will be a problem for drinking water and aquatic life even though the reason of the problem is different : regulations for drinking water and possible algae growth for aquatic life. There will not be a big problem for the leisure activities, as we are not supposed to drink much when we bath; and there will obviously not be any problem for irrigation. A similar analysis can be made for many other parameters.
So we can assess water quality in many different ways depending on what sort of suitability we are speaking of. And we can also assess water quality, trying to describe contaminating concentrations compared with a reference situation without significant human influence, which is a water quality description behind all these water suitabilities. Water-SEQ calculates all these different suitabilities and the quality description, as shown in Figure 1. It is then a set of tools composed of many different tools.

![Figure 1: Water quality and water suitabilities to uses and to aquatic life](image)

Another characteristic of Water-SEQ is the inclusion of "indicators" which are parameter groups created to be understandable to the layman. We know that many parameters are not easy to understand for the non specialist. And we know that improving water quality is obtained by actions which always concern several groups of parameters, such as organic matter or pesticides. That is why "indicators" have been created.

The Water-SEQ organisation is described in Figure 2 where parameters are first grouped to form indicators; and where it is then possible to calculate either water quality, using quality classes and quality indexes, or water suitabilities using suitability classes. Suitability indexes are to be added to suitability classes, in version 2 of Water-SEQ, but only for suitability to aquatic life.

![Figure 2: Water-SEQ organisation](image)
Indicators

Indicators are groups of parameters as shown on Figure 3.

<table>
<thead>
<tr>
<th>Indicators</th>
<th>Parameters</th>
</tr>
</thead>
<tbody>
<tr>
<td>Organic and oxidizable matter</td>
<td>$O_2d$, $%O_2$, DCO, $DBO_5$, COD, NKJ, $NH_4^+$</td>
</tr>
<tr>
<td>Nitrogen (except nitrates)</td>
<td>$NH_4^+$, NKJ, $NO_2^-$</td>
</tr>
<tr>
<td>Nitrates</td>
<td>$NO_3^-$</td>
</tr>
<tr>
<td>Phosphorus</td>
<td>$PO_4^{3-}$, total P</td>
</tr>
<tr>
<td>Suspended matter</td>
<td>Suspended solids, turbidity, transparency</td>
</tr>
<tr>
<td>Color</td>
<td>Color</td>
</tr>
<tr>
<td>Temperature</td>
<td>Temperature</td>
</tr>
<tr>
<td>Salinity</td>
<td>Conductivity, $Cl^-$, $SO_4^{2-}$, $Ca^{2+}$, $Mg^{2+}$, $K^+$, $Na^+$, TAC, hardness</td>
</tr>
<tr>
<td>Acidity</td>
<td>$pH$, dissolved Al</td>
</tr>
<tr>
<td>Phytoplankton</td>
<td>$%O_2$ and pH, chlorophyll a + pheopigments, algae, $\Delta O_2$ (24 hours)</td>
</tr>
<tr>
<td>Microorganisms</td>
<td>Total coliforms, faecal coliforms, faecal streptococci</td>
</tr>
<tr>
<td>Mineral micropollutants in water</td>
<td>Arsenic, mercury, cadmium, lead, total chromium, zinc, copper, nickel, selenium, baryum, cyanides</td>
</tr>
<tr>
<td>Metal with bryophytes (moss)</td>
<td>Arsenic, mercury, cadmium, lead, total chromium, zinc, copper, nickel</td>
</tr>
<tr>
<td>Pesticides in water</td>
<td>37 substances are concerned</td>
</tr>
<tr>
<td>Organic micropollutants in water</td>
<td>59 substances are concerned</td>
</tr>
</tbody>
</table>

Figure 3: Indicators and parameters included in the 1st version of Water-SEQ

A total number of 15 indicators (calculated with 135 parameters) have been defined. All sorts of problems concerning water suitability to uses or to aquatic life are then described, except radioactivity, which can be easily added if we are able to describe the suitability classes to aquatic life and to uses. Each indicator is described with one or more parameter. In the case of the first two indicators, «organic and oxidisable matters» and «nitrogen except nitrates», we find two parameters in both : $NH_4^+$ and NKJ, as they have different effects. They are included in the first indicator because of their ability to be oxidised, and in the second because they are nutrients and because $NH_4^+$ can be converted to the toxic form : $NH_3$.

Figure 4: How uses are influenced by indicators (water suitability to aquatic life being included)
Water suitability to uses

The applicability of the indicators varies among the water uses (Figure 4). Water suitability to aquatic life have been presented in the same way, even if it is not a human use. For each conjunction of indicator and use, and for each conjunction of indicator and aquatic life, a threshold table has been defined, for all or part of the indicator parameters, to determine suitability class.

Water suitability is assessed with a maximum of 5 suitability classes (blue / green / yellow / orange / red) specifically defined for each use, as shown in Figure 5. The thresholds chosen to determine the change from one suitability class to another, for the considered water uses, were defined according to European Directives (Anonymous 1989; European Commission 1975, 1980, 1981, 1991, 1998), bibliographical sources (Canada Council 1992, 1999) and expert opinions.

<table>
<thead>
<tr>
<th>AQUATIC LIFE</th>
<th>Blue</th>
<th>Green</th>
<th>Yellow</th>
<th>Orange</th>
<th>Red</th>
</tr>
</thead>
<tbody>
<tr>
<td>AQUATIC LIFE</td>
<td>all taxa present</td>
<td>some taxa absent</td>
<td>numerous taxa absent</td>
<td>diversity reduced</td>
<td>diversity very low</td>
</tr>
<tr>
<td>DRINKING WATER</td>
<td>acceptable</td>
<td>simple treatment</td>
<td>normal treatment</td>
<td>full treatment</td>
<td>unsuitable</td>
</tr>
<tr>
<td>LEISURE</td>
<td>optimal</td>
<td>acceptable</td>
<td>acceptable</td>
<td>unsuitable</td>
<td></td>
</tr>
<tr>
<td>IRRIGATION</td>
<td>very sensitive plants</td>
<td>sensitive plants</td>
<td>resistant plants</td>
<td>very resistant plants</td>
<td>unsuitable</td>
</tr>
<tr>
<td>LIVESTOCK</td>
<td>all animals</td>
<td>mature animals</td>
<td>unsuitable</td>
<td></td>
<td></td>
</tr>
<tr>
<td>AQUACULTURE</td>
<td>all types of breeding</td>
<td>all adult fishes</td>
<td>unsuitable</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Figure 5: Suitability classes to uses (water suitability to aquatic life being included)

Reference values and suitability classes to aquatic life

The thresholds selected for parameters describing the limit between high status (blue class) and good status (green class) in water suitability to aquatic life come from two sources, according to the proposed framework directive (European Commission 1999) for community action in the field of water policy:

- concerning natural substances such as nutrients, all the data measured in non human influenced rivers have been collected; the threshold value selected is the upper value of 90% of the observed values. We have then the following blue/green threshold for some nutrients:
  - Nitrates: 2 mgNO₃/l
  - Ammonia: 0,1 mgNH₄/l
  - Phosphates: 0,1 mgPO₄/l
  - Total Phosphorus: 0,05 mgP/l

- concerning micropollutants, the thresholds selected are issued from chronic NOEC (no observed effect concentration), with a safety factor of 10, or from 50-LC, with a safety factor of 1000, as proposed in section 1.2.6 of annex V in the water framework directive (8). We have then the following blue/green threshold for some pesticides and other organic micropollutants:
  - Atrazine: 0,2 mg/l
  - Simazine: 0,02 mg/l
  - Lindane (g-HCH): 0,01 mg/l
  - Benzo(a)pyrene: 0,005 mg/l

The other thresholds concerning aquatic life have been defined, for macropollutants, when they are able to indicate a gradual impoverishment of the biological structure, including the disappearance of the taxa most sensitive to pollution. Each suitability class is defined using the two following criteria:

- potential presence/absence of pollution sensitive taxa, 
- potential diversity of the communities and number of trophic levels present.
In the case of micropollutants, thresholds are issued from ecotoxicity tests, the definition of the suitability classes and thresholds being:

- **Blue**: No measurable risk for species
  - threshold: NOEC/10 or 50-LC/1000

- **Green**: Risk of chronic or sub-lethal effects for the most sensitive species, especially young
  - threshold: NOEC or 50-LC/100

- **Yellow**: Risk of chronic or sub-lethal effects, reduction in numbers, domination of less sensitive species
  - threshold: lowest 50-LC with no security factor

- **Orange**: Risk of lethal effects on sensitive species, reduction of number of species
  - threshold: 50-LC for 3 trophic levels (algae/plants, invertebrates and fishes)

- **Red**: Very high risk of lethal effects on different species, reduction in numbers and variety of species

### Water quality classes and indexes

As we saw in the water suitability section, no water use is connected with all the indicators which are necessary to describe water quality. This situation will still be more obvious when we shall use micropollutant measurements in sediments and in suspended solids, as Water-SEQ version 2 will do. We then have to create a water quality index description. It has been determined by applying to each parameter and each indicator the following conventions, taking into account water suitability to aquatic life, and to uses connected with human health:

- The index value 80 is the limit between blue and green quality classes. It is associated with the first threshold which limits the blue suitability class for «aquatic life», or for «drinking water» or «leisure» uses.
- The index value 20 is the limit between the orange and red quality classes. It is associated with the first threshold which passes into the red suitability class for «aquatic life», or for «drinking water» or «leisure» uses.
- The index values 60 and 40 are the limits between green and yellow, and yellow and orange quality classes. They are associated, whenever it is possible, to change of suitability classes, green to yellow for 60, yellow to orange for 40 for «aquatic life», or for «drinking water» or «leisure» uses.

When there is a possible choice between thresholds for suitability to aquatic life or to drinking water production, preference is given to the first one.

In the case of nitrates, thresholds for suitability classes to biology blue/green/yellow/orange/red are respectively 2, 10, 25 and 50 mg/l. Threshold for suitability classes to drinking water blue/red is 50 mg/l in so far as water quality is acceptable below and unsuitable for treatment above. Thresholds for quality classes are then 2, 10, 25 and 50 mg/l. An example of the index values construction for this indicator is given below in figure 6.

![Quality index](image)

**Figure 6: Index values**
Calculation rules

Calculation rules have been defined for all suitability classes and for quality classes and indexes:

To qualify a sample for assessment:
- Some imperative parameters have been determined for each indicator. If one measure of the imperative parameters is lacking, the indicator cannot be quantified,
- The overall quality (quality classes and quality indexes) and the overall suitability classes, for each indicator, are determined by the parameter that gives the worst result.

To assess the annual or interannual quality:
- A minimum number and distribution of samples during the period are required to qualify each indicator,
- The overall quality (quality classes and quality indexes) and the overall suitability classes, for each indicator are determined by the most downgrading sample observed in at least 10% of the samples analysed during the period.

The last rule has been selected to avoid exceptional situations and ensure a realistic assessment. The following aggregation method is applied. It consists in considering the samples giving the worst suitability or the worst quality observed in at least 10% of the samples. It is called “the 90% rule”. This rule stipulates that only 90% of the results (suitability classes, quality classes or quality indexes) obtained over a period of time are taken into account. Thus, after having classified the suitability and quality classes by decreasing order, and the quality indices by increasing order, the results taken into account are calculated with the HAZEN formula:

\[ F = \frac{(I - 0.5)}{N} \]

where \( I \) = row of the result, \( N \) = total number of results, \( F \) = percentile.

For the 90% percentile, \( F = 0.9 \), the row of the result to be kept is \( I = (0.9 \times N) + 0.5 \). For instance, when \( N = 12 \), \( I = 11.3 \), rounded to 11, the 11th result of 12 is selected.

Figure 7: Example of calculation results with Water-SEQ version 1
RESULTS AND DISCUSSION

Calculation results and quality objectives

Once the suitability classes, the quality classes and the quality indexes are assessed, the results are given by the calculation tool for a period of one year or several years (from 2 to 5 years). An example of the results concerning quality classes and indexes, as calculated now (version 1) is shown in Figure 6. Much of the data are then processed. In this case it concerns a measurement station with 12 samples a year and all the results of 1998 are treated to give water quality and water suitability for the period.

Water quality classes are given for each of the 15 indicators if the results allow calculation. They are used to draw river quality maps, in a similar way as it has been done in France during the last thirty years. Quality indexes are used to draw chronologies. Water suitability classes are to be compared to what is required for uses and for aquatic life. It is then possible to identify the indicators associated with the bad water suitability classes, and to define restoration objectives for each indicator using water quality classes and indexes: class and index objectives. Those objectives, when reached, will then allow uses and aquatic life, as required before.

New indicators

A second version of Water-SEQ is to be produced soon. As shown in Figure 8, it will include micropollutant measurement in sediments and in suspended solids to calculate water quality. Polycyclic aromatic hydrocarbons (HAP) measured on sediment will be used also to calculate water suitability to aquatic life. At the same time, in Water-SEQ version 2, it will be possible to assess water suitability to aquatic life, not only though suitability classes, but also through suitability indexes. It will help calculating quality ratio for ecological quality; suitability index varying from «0» (the worst situation) to «100» (reference conditions) being similar to ecological quality ratio (EQR) defined in water framework directive for the physico-chemical component of ecological quality.

<table>
<thead>
<tr>
<th>Macropollutants in water</th>
<th>Micropollutants</th>
<th>the quality can be lowered by:</th>
</tr>
</thead>
<tbody>
<tr>
<td>Organic and oxidized matter</td>
<td>MOOX&lt;sup&gt;•&lt;/sup&gt;</td>
<td>MPMI - in bryophytes (1)</td>
</tr>
<tr>
<td>Nitrogen (except nitrates)</td>
<td>AZOT&lt;sup&gt;•&lt;/sup&gt;</td>
<td>- in water</td>
</tr>
<tr>
<td>Nitrates</td>
<td>NITR&lt;sup&gt;•&lt;/sup&gt;</td>
<td>- in sédiments</td>
</tr>
<tr>
<td>Phosphorus</td>
<td>PHOS&lt;sup&gt;•&lt;/sup&gt;</td>
<td>- in suspended solids</td>
</tr>
<tr>
<td>Phytoplankton</td>
<td>PHYT&lt;sup&gt;•&lt;/sup&gt;</td>
<td></td>
</tr>
<tr>
<td>Suspended matter</td>
<td>PAES&lt;sup&gt;•&lt;/sup&gt;</td>
<td></td>
</tr>
<tr>
<td>Température</td>
<td>TEMP&lt;sup&gt;•&lt;/sup&gt;</td>
<td></td>
</tr>
<tr>
<td>Acidification</td>
<td>ACID&lt;sup&gt;•&lt;/sup&gt;</td>
<td></td>
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<tr>
<td>Salinity</td>
<td>MINE</td>
<td></td>
</tr>
<tr>
<td>Color</td>
<td>COUL</td>
<td></td>
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<tr>
<td>Micro-organisms</td>
<td>BACT</td>
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</tr>
<tr>
<td>Minéral micropollutants</td>
<td>PEST</td>
<td></td>
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<tr>
<td>- in water</td>
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<tr>
<td>- in sédiments</td>
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<tr>
<td>- in suspended solids</td>
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<tr>
<td>Pesticides</td>
<td></td>
<td></td>
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<tr>
<td>- in water</td>
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<tr>
<td>- in sédiments</td>
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<tr>
<td>- in suspended solids</td>
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<tr>
<td>HAP</td>
<td></td>
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<tr>
<td>- in water</td>
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<tr>
<td>- in sédiments</td>
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<td>- in suspended solids</td>
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<tr>
<td>PCB</td>
<td></td>
<td></td>
</tr>
<tr>
<td>- in water</td>
<td></td>
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<tr>
<td>- in sédiments</td>
<td></td>
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<tr>
<td>- in suspended solids</td>
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<tr>
<td>Other organic micropollutants</td>
<td>MPOR</td>
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</tr>
<tr>
<td>- in water</td>
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<tr>
<td>- in sédiments</td>
<td></td>
<td></td>
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<tr>
<td>- in suspended solids</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Figure 8: New indicators selected in version 2 of Water-SEQ

CONCLUSIONS

Water-SEQ determines the suitability of water to satisfy different uses or its suitability to biology. Furthermore, it defines water quality by means of quality classes that range from blue to red and with quality indexes enabling the monitoring of the river water quality.
Water-SEQ offers the possibility of:
- reporting water suitability for aquatic life and to uses,
- comparing it, for the biota to framework directive requirements, and for the uses with the required quality,
- identifying the indicators which influence water quality,
- defining the restoration objective for each indicator (quality class and index). Water-SEQ tool is able to calculate quality and index class for each indicator and each parameter, when suitability objectives are defined,
- reporting the efficiency of the different policies in restoring water quality with quality classes and indexes.

Such a tool was necessary for decision-makers such as managers, public services, technicians, users and politicians. It therefore had to be in accordance with French and European regulations. It is notable that this set of tools is adaptable thanks to its modular structure. For instance, water suitability classes of new uses can be added, and additional parameters to describe an indicator can be taken into account as long as thresholds are determined in accordance with the suitability classes and the quality classes. It also can be adjusted when the regulations are updated.

REFERENCES
INDICATORS FOR WATER POLICY EVALUATION
FROM A NETWORK MANAGEMENT PERSPECTIVE

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Evaluation is the process of collecting information that is needed to learn from success and failure of policy. Policy-makers need this information to assess the correctness of the implementation phase, to legitimize decisions like the allocation of budgets, and to direct future policy. Information from monitoring and reporting of water system conditions (pressure, state, and impact indicators) are very important but do not suffice. Policy indicators must be able to describe and explain the effectiveness, efficiency, and equity of the policy. A systematic approach to water policy evaluation as an integrated part of the policy cycle can be used to develop policy-process indicators. Indicators of the perceived quality of the policy process are important for the learning function of policy evaluation. The selection of suitable indicators will be addressed from the network management perspective. The case of a water policy aiming at reducing greenhouse emissions will be used to illustrate the theoretical concepts.

Keywords: water policy evaluation, stakeholders, network management, (policy) indicators.

INTRODUCTION

Authorities at regional, provincial, national and even supranational (European) level make water policies to address water management issues. Implementation of these policies generally requires a integrated effort of public and private organizations. As a rule, these policies are evaluated ex post to assess the success of the policy program. Some policy documents are revised regularly, e.g. the Dutch national and regional water policies, and the evaluation is a part of the preparation of the new document. The Water Framework Directive for the European water policy foresees in policy evaluation and stipulates regular reporting, on a river basin scale, of water system conditions and of human activities that influence these conditions. In the Netherlands, regional water system reports are compiled and aggregated to provide a national consistent view of water system conditions. These reports should also enable the evaluation of regional and provincial water management objectives (IPO, 1998).

Evaluation is the process of collecting information that is needed to learn from success and failure of policy. Policy-makers need this information to assess the correctness of the implementation phase, to legitimize decisions like the allocation of budgets, and to direct future policy. Thus, policy evaluation is about all aspects of policy implementation. However, information strategies for water management focus on the monitoring and reporting of water system conditions mainly. This information does not suffice to support policy-making for complex water issues. "Little is known about the achievements of policy implementation and even less about the reasons why it did not succeed. It may be [even] more effective for policy review to devise a system for investigating the achievements of the implementation phase, than to gather more data on e.g. water quality" (Witmer 1994).

The question that arises is how should water policy be evaluated to fulfill the learning function as stated above. A systematic approach to water policy evaluation as an integrated part of the policy cycle should address the following aspects of policy evaluation:
(1) Who should have information on the success of a water policy?
(2) What is their need for information?
(3) What kinds of indicators are needed to yield this information?
(4) How and when in the policy process should these indicators be developed?
These questions will be answered from a theoretical perspective; the practical case of a water policy aiming at reducing emissions from greenhouses will be used to illustrate the theoretical concepts.
THEORETICAL FRAMEWORK

Water management and stakeholder networks

Water management aims at supporting a range of human activities; these activities pose different and often conflicting demands on the water system. In other words, people have different stakes in how water systems are managed in the short and long term. Also, the different stakeholders are often interdependent because of the different ways that they influence and depend on the water system. A well-known example is the interdependency between water-related human activities in upstream and downstream countries of a river basin. Policy-making or planning in water management thus requires that the interdependencies are honored. This is especially relevant when the authority to develop and implement policy is fragmented or shared between public and private parties. The stakeholders need resources or efforts of other stakeholders to attain their goals. The concept 'policy network' or 'stakeholder network' can be used to indicate such patterns of (strategic) relations between interdependent stakeholders involved in policy processes (Kickert et al., 1997).

Figure 1. Stakeholder network: reducing greenhouse-emissions to surface water

A policy network may be depicted as in Figure 1: a network of public and private stakeholders involved in a policy process to reduce those emissions from greenhouse farming which pollute surface waters. The policy aims at setting norms for the use of chemicals and restrictions on discharge of waste water to the surface water. In this particular case the national government initiated the policy process but implementation of the policy depended on other parties. Greenhouse farmers were to implement measures that reduce the emissions of nutrients and pesticides. The farmers would also bear the costs for the necessary investments in accordance with the polluter-pays-principle. It was expected that some farmers would not be able to make the necessary investments and thus forced out of business if the policy would be accepted. The agricultural lobby group was involved to protect the interests of the greenhouse farmers. Regional water boards were involved because they were charged with the enforcement of the national policy. Their relation to the municipalities may be explained by the fact that in many case farmers were not able to get access to a sewer system. Municipalities and the water board have to cooperate to expand sewer system access. This means taxes may have to be raised. Households are expected to benefit from the water quality improvements in the local area. They contribute to the policy by sharing costs for expansion of the municipal sewer systems and wastewater treatment infrastructure.

The role of networks in policy making

By definition, stakeholders in a network are interdependent. The nature and extent of these interdependencies can be modified during the policy process because of internal or external changes. Stakeholders show strategic behaviour in policy networks to change the policy process and the content of the policy document to their advantage. The management of the relationships in the network, or network management, aims at bringing the policy process to
meet its objectives. Stakeholder behaviour may be influenced by provision of information or organization of stakeholder meetings with different aims and working methods.

Stakeholders may improve, reduce, or even undo the effectiveness of the policy. If and how they affect policy depends on the following stakeholder characteristics (Allison, 1971; Bressers, 1983):

1. **Stakeholder goals and priorities may comply or conflict with policy goals.** The (lack of) agreement between goals determines whether a stakeholder supports or hampers policy. The chance of successful policy increases if the involved stakeholders give a high priority to policy goals;

2. **Information about policy and policy instruments.** Knowledge, education, experience is used to influence policy. The success of policy increases with the availability of usable information and the ability to collect it for involved stakeholders;

3. **Power and resources to actually influence policy.** This can be formal power, expressed in authority, or informal power that can be expressed in a strong lobby or a high degree of organization. Stakeholders with a strong position of power and priorities in line with policy are expected to be more successful in the policy process.

Stakeholder influence is likely to occur in several moments of the policy cycle, or the policy process. To make this visible, we consider the policy process as a process consisting of several sub-processes as planning, decision making, implementation, and evaluation. These processes transform the policy elements, from policy issues in the beginning to a policy evaluation document in the end (figure 2). Note that the iterative processes and feedback loops that occur during a policy-process are not depicted.

Stakeholders participate in planning processes, trying to get their interests looked after and put on the political agenda. The interaction between the organization of the policy process (open or semi-closed) and stakeholder characteristics determine stakeholder behaviour and need for information. In the actual decision making process, they will try to influence the policy goals and priorities. Beyond that stage, stakeholders exert influence by exploiting the interdependencies that result from their role in the implementation of policy. Information plays an important role throughout the process: stakeholders may bring new information into the policy-process in order to influence the outcome of the process or alter their strategy based on new knowledge or insights. Information can also be used to manage the network of stakeholders. Carrying out extra research during a policy process is a typical example of how a policy-maker may try to keep all stakeholders, believers and non-believers, involved in a policy-process and prevent the process from stalling or going nowhere. Policy evaluation may be used to generate information that can serve new policies or confirm the success of past policies. Evaluation is thus important for the continued cooperation among stakeholders.

![Figure 2. The policy process influenced by stakeholders (adapted from Bressers, 1983)](image-url)
The role of policy evaluation in networks

Figure 2 shows the process of evaluation as an integral part of the policy cycle. Depending on the study design, an evaluation generates knowledge about policy results and the policy process. The arrows of influence in the graph indicate that an evaluation is subject to stakeholders' influence and that the evaluation outcome can influence the future participation and attitude of stakeholders. Policy evaluation can play an important role in network management. Policy makers can design better policies and request participation of stakeholders based on the evaluation outcome. Stakeholders may use the evaluation to better understand their interdependencies and to determine their actions in following policy processes. Furthermore, the evaluation outcome may stimulate stakeholders to participate (again) in policy planning and prevent undesirable behaviour (hampering) during policy implementation. Thus, the usefulness of the evaluation outcome depends on the extent it meets the need of information of all stakeholders.

The information yielded in a policy evaluation can be used by the network in three ways (Rossi & Freeman, 1993):

(1) Information can be used to develop a rationale for action. The policy outcome may be unsatisfactory or reveal new problems that need to be addressed in new policies.

(2) Information can be used to prepare go/no-go decisions. Knowledge of the effectiveness of the policy process and the policy itself serves as input for decisions on planning methods and tactics that should or should not be part of new policy programs.

(3) Information can be needed for legitimization and accountability. Information on how well interventions are implemented, the extent to which they reach targets, their impacts and their costs may help to ward off adversaries and to increase acceptance.

Information also enables stakeholders to learn from success and failure of policy, which can be used to plan strategies for participation in new policy cycles. (Figure 2). This learning function of evaluation is of high importance in policy cycles like the frequent revision of national and regional water management plans. What do policy makers, and especially politicians, want to learn? They have especially interest in the effectiveness, efficiency, and equity of the implemented policy (De Bock et al., 1996). Effectiveness focuses on the achievement of policy objectives: have we reached our goals? Efficiency focuses on financing and the instruments used to achieve policy goals. Equity, on the other hand, focuses on the distribution of gains and losses (costs and benefits) among the stakeholders as a consequence of the chosen policy and instruments. Because of the importance of the learning function of evaluation in the policy cycle, the policy outcome should not only be described but also be explained to the extent possible. This places requirements on the type of indicators that ought to be used in an evaluation as well as on the methods to collect the data.

INDICATORS FOR WATER MANAGEMENT

Present indicators for water management

The present indicators for water management focus on cause-effect relations between the water system and human activities. They are developed according to the so-called pressure-state-impact-response framework (P-S-I-R) (Lorenz, 1999; van Harten et al., 1995). Pressure refers to human activities that influence the water system. State indicators describe the actual change of state, or

<table>
<thead>
<tr>
<th>dimension</th>
<th>description</th>
<th>indicator</th>
</tr>
</thead>
<tbody>
<tr>
<td>pressure</td>
<td>greenhouses discharge water containing nutrients and pesticides to surface water.</td>
<td>emission (in kg) or load (kg/m³) of nutrients and pesticides.</td>
</tr>
<tr>
<td>state</td>
<td>the emission of these substances deteriorates the water quality</td>
<td>concentrations of the nutrients and pesticides in water; toxicity of water.</td>
</tr>
<tr>
<td>impact</td>
<td>the state of the receiving waters impedes other functions such as nature, agriculture and recreation (sport fishing)</td>
<td>biodiversity; volume of extractable water (considering water quality norms) for watering cattle; sport fish abundance.</td>
</tr>
<tr>
<td>response</td>
<td>greenhouse emissions are placed on the political agenda and plans are made for emission reduction</td>
<td>need for response (based on P-S-I); planned and achieved emission reduction</td>
</tr>
</tbody>
</table>

Table 1. An example of a set of indicators of the P-S-I-R framework.
ecological functioning of the water system. Impact indicators describe the effect of a (change of) state to the supply of water system goods and services. Response describes societal response to environmental changes. An excellent presentation of this framework of indicators for environmental management is given by Lorenz (1999). Table 1 illustrates this framework in relation to the case of reduction of greenhouse emissions.

In theory, P-S-I-R can be used to evaluate water policy. In practice, policy evaluation is hampered by a lack of appropriate indicators. It is, for example, difficult to select appropriate pressure indicators. The nutrient and pesticide emissions (in kg) depend on factors that vary widely among greenhouse companies, namely the type of culture and best available management practices. Waste load, on the other hand, is not an appropriate pressure indicator either because drainage water flow rates change with the growing season. In other words, the monitoring efforts to quantify these pressure indicators are costly compared to the value of information that can be yielded. In this particular case, policy evaluation requires other pressure indicators, e.g. company data about the quantity of nutrients and pesticides used. Such indicators require a different type of monitoring effort like company surveys and interviews. In fact, a new environmental policy for greenhouse operations in the Netherlands requires that companies hold detailed records of water discharge rates and of the amounts of chemicals applied to a crop. These data will be used to monitor progress in reduction of greenhouse emissions.

**Indicator selection: meeting the information need**

Development of P-S-I-R indicators for policy evaluation must be subject to the information need (Lorenz 1999). Information need, however, is not unambiguous: it depends on the composition of the stakeholder network, the role of the different stakeholders in the policy cycle, the and stakeholder characteristics. Four questions must be answered to be able to define an information need:

- Who need(s) information during a policy cycle, and from a policy evaluation in particular?
- What information do they need?
- When do they need this information?
- How can this information be collected?

We have already discussed the role of policy evaluation in networks. The stakeholders in a network will have questions that concern effectiveness, efficiency, and equity of a policy. In theory, the P-S-I-R indicators can give limited information about policy effectiveness; additional indicators are needed to explain (rather than describe) the policy outcome. Policy efficiency and equity cannot be described with P-S-I-R indicators. Still, P-S-I-R indicators are the focus of national and regional monitoring efforts for water policy evaluation (IPO 1998, Elbersen and van Tilborg, 2000). Perhaps, this can be explained by the observation that policy evaluation typically focuses on what is most easily measured rather than on information needed for learning in a policy-making (Patton & Sawicki, 1986).

Water policy evaluation for network management requires an approach that goes beyond the typical monitoring efforts. The evaluation will monitor the policy process quality rather than water quality. For instance, in the case of reducing greenhouse emissions, the regional water authority charged with policy enforcement may raise the following questions:

- Were sufficient means (personnel, expertise, and finances) made available to enforce the policy?
- What aspects of the policy content formed obstacles in policy enforcement?
- In what cases was enforcement of the policy judged wrong by a court of justice, and why?
- What is the return on policy enforcement with regard to the water quality in the area?

These questions arise from problems with the feasibility of one aspect of policy implementation, enforcement of the policy measures according to the law. The water authority needs the answers to the above questions to determine the strategy they need to follow in future policy processes. More information about the obstacles they have encountered may be used to try and change the content of the new policy, to seek alliances with other stakeholders to expand their power base, etcetera. Of course, different stakeholders in a network (see figure 1) will raise different questions. The environmental lobby may be interested to know if the goals for water quality improvements were set high enough to actually improve the quality of the aquatic environment. The agricultural lobby, on the other hand, may want to know the reasons for the high number of company closures resulting from this policy. The answers to all of these questions can have consequences for stakeholder strategies. Information about the quality of the policy process is important for the management of the stakeholder network and future policy processes.
Evaluation of policy-process quality

The success or failure of policy implementation often can be explained by stakeholder behaviour in the planning and decision-making phases. Therefore, we define the quality of the policy process by the (inter)actions of stakeholders within the network. But what information is needed to assess the quality of the policy process? Bressers (1983) distinguishes four important aspects of process quality in policymaking (Table 2). These quality aspects should be assessed in a policy evaluation.

<table>
<thead>
<tr>
<th>items for evaluation of process quality</th>
<th>aspects of process quality</th>
<th>information with strategic value</th>
</tr>
</thead>
</table>
| course of the policy process: changes over time | correctness  
acceptance | desired information  
actual information |
| correctness of policy content: coupling policy – practice | goals – policy  
ambition level | information – policy |
| acceptance of policy content | consensus on goals  
beneficial prospects | discrepancy vs. corresponde |
| strategic behaviour: (inter) actions in the process | obstruction versus co-operation | communication lines |

Table 2. Policy indicators (adapted from Bressers, 1983)

A policy process evaluation must obtain information about stakeholder behaviour and about their perceptions of the process quality. This can be done by monitoring strategic behaviour and interviewing of stakeholders during the process. First, the course of the policy process itself must be investigated. This means that each phase of the policy process must be reviewed, rethinking what should have been done and comparing with what actually had been done (correctness) and on the acceptance of the course of policy events. This review must include an assessment of the (im)possibilities of participation by stakeholders in the different stages in the policy cycle.

Policy evaluation requires information on the correctness of policy content also. Policy issues change over time and some issues may be solved more easily than others. The relevance of the current policy, and information on the needs for reformulation or deletion of policy items, is important input for the next policy cycle. Correctness of policy content refers to the relationship between policy and practice. This quality aspect reflects if policy goals addressed the policy problem, if stakeholders had sufficient information regarding the intended policy, and if the allocation of power and resources supported policy implementation. Satisfaction with the level of ambition also contributes to the perception of correctness. Stakeholders who agree on the correctness of policy goals still do not necessarily agree on ambition levels or targets.

Information on the acceptance of policy content can be useful to explain the success of policy implementation. Acceptation and commitment determine stakeholder behaviour. Note: acceptance changes over time as policy issues become more or less important for stakeholders. The acceptance of policy content must be seen in relation to the distribution of stakeholder characteristics. Do stakeholders agree with policy goals and do they experience beneficial prospects? Do they judge the present allocation of power and resources suitable and equitable for policy implementation?

Policy is influenced by the strategic behaviour, the factual actions and interactions during the policy process, of the stakeholders in the network (figure 2). Strategic behaviour in a network can be described in terms of the information that is exchanged or withheld, the means that have been used to implement policy, and the power that has been used. An evaluation of strategic behaviour focuses on situations of obstruction or co-operation especially, and on the dynamics of activity within the network. Information on strategic behaviour can be used to explain if and why policy turned out more or less effective, efficient or equitable as intended.
Indicators for the policy-process quality

The development of policy indicators is based on the stakeholder characteristics on the one hand and the information need on the other. The information has to be obtained in a dialogue with stakeholders, using surveys and interviews. The big difference with monitoring a water system is that now perceptions are monitored instead of actual situations. This can be illustrated with the case of greenhouse emission reduction. Tables 3 and 4 present policy indicators for correctness and acceptance of policy content together with examples of interview questions. The answers to these questions are subject to bias as each stakeholder gives information based on his own goals, knowledge base and resources. The answers may change according to the phase in the policy process the interviewee refers to. The questions in tables 3 and 4 were formulated for evaluation of the entire policy cycle.

<table>
<thead>
<tr>
<th>indicators</th>
<th>description of information need</th>
<th>applied to emission reduction policy for the greenhouse sector</th>
</tr>
</thead>
<tbody>
<tr>
<td>coupling goals – policy</td>
<td>Did we solve the right problem?</td>
<td>Has water quality improved in response to greenhouse emission reduction?</td>
</tr>
<tr>
<td>ambition level</td>
<td>Was this policy feasible? Did we set the right targets?</td>
<td>Was the target for water quality improvement sufficient to restore the ecosystem?</td>
</tr>
<tr>
<td>coupling information – policy</td>
<td>Was relevant information about the policy available?</td>
<td>Were stakeholders aware of limitations in access to municipal sewer systems during decision-making phase?</td>
</tr>
<tr>
<td>coupling balance of power – policy</td>
<td>Is the present allocation of power adequate to implement policy?</td>
<td>Do water authorities have enough power and resources to influence the behaviour of greenhouse farmers?</td>
</tr>
</tbody>
</table>

Table 3. ‘Correctness of policy content’ indicators for policy evaluation

The use of policy indicators requires a rather different approach than traditional water system indicators. Policy indicators refer to the interdependent relations in the network. The evaluator should therefore pay attention to the following issues:

1. Policy indicators have ‘soft’ standards; they are not used to judge or disqualify. ‘Soft’ and negotiable standards protect the interdependent relations in the stakeholder network from damage or conflict. Since policy indicators deal with stakeholder perceptions, the evaluation results cannot be used to criticize or accuse stakeholders for shortcomings and disturbances of the policy process. It is possible however, to indicate the ‘bottlenecks’ in the policy process (Bressers, 1983). Instead of judgement and accusation it is preferable that

<table>
<thead>
<tr>
<th>indicators</th>
<th>description of information need</th>
<th>applied to emission reduction policy for the greenhouse sector</th>
</tr>
</thead>
<tbody>
<tr>
<td>consensus on goals</td>
<td>Is policy in line with stakeholder interests?</td>
<td>Did greenhouse farmers commit to the water quality goals?</td>
</tr>
<tr>
<td>beneficial prospects</td>
<td>Does policy yield benefits to stakeholders?</td>
<td>What benefits do greenhouse farmers experience from this policy? Do other stakeholders benefit? How?</td>
</tr>
<tr>
<td>discrepancy vs. correspondence</td>
<td>Which stakeholders disposed of information in line with the policy?</td>
<td>Did information about the policy that greenhouse farmers had access to (e.g. news media) correspond with the policy?</td>
</tr>
<tr>
<td>balance of power, ascendancy</td>
<td>Did powerful stakeholders accept the policy?</td>
<td>How did the greenhouse sector and water authorities receive the policy?</td>
</tr>
<tr>
<td>use of instruments</td>
<td>Which instruments and measures were put into practice, how and by whom?</td>
<td>What instruments were used to execute policy? What measures were used to either hamper or support policy?</td>
</tr>
</tbody>
</table>

Table 4. ‘Acceptance of policy content’ indicators for policy evaluation
stakeholders negotiate or discuss policy results, policy implementation and stakeholder behaviour.

(2) Policy indicators refer to qualitative information. This is less exact than quantitative information but it suits different points of view and is based upon stakeholder experience (Bressers, 1983). Quantitative expression and ‘hard’ judgements stated by the evaluator are not appropriate, as the multitude of opinions within a stakeholder network makes it difficult to set an unambiguous standard for policy indicators.

REFLECTION

Policy evaluation aims at gathering information on effectiveness, efficiency, and equity of the policy process. A policy evaluation must be set up following a systematic or structured approach. Preparation of the evaluation starts, in fact, at the beginning of the policy cycle. Stakeholder participation in the design of the evaluation is important to be able to supply the information that is needed for network management. The following conditions are important for a successful evaluation:

Policy goals have to be well formulated; they are the basis for the evaluation of success. Stakeholders must be informed about the course of the policy process and about the implementation phase, especially. Policy indicators must be chosen in accordance with policy goals. The indicators are designed to yield information about the effectiveness, efficiency, and equity of the policy. Additional indicators are selected to be used to explain policy implementation and results.

To fulfill these conditions, we suggest the following good practices:

(1) Decide during the decision-making phase upon the purpose of an evaluation and according information need.

(2) Keep in mind that stakeholders influence the policy process, either through strategic behaviour or by chance, throughout all phases of the policy cycle. Monitor their behaviour during the process and use this information to help explain policy success.

(3) Consensus or commitment among stakeholders about (policy) indicators is important for value of can the policy evaluation. This can be achieved by inviting stakeholders to participate in the indicator selection process.

Selection of policy indicators at an early stage in the policy process can be beneficial for the policy process as a whole: it enables stakeholders to exchange ideas and assumptions about the policy issue, negotiations, implementation, and evaluation. In that regard, evaluation of the process quality can be seen as an instrument for network management. For water policy, this is especially interesting. Discussions about the water quality monitoring effort required to support policy enforcement, can now be expanded with dialogues on effectiveness, efficiency, and equity of the water policy. Such discussions will strengthen the network of stakeholders, support co-operation among stakeholders, and so contribute to improving water quality.

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UN/ECE GUIDELINES ON MONITORING AND ASSESSMENT OF INTERNATIONAL LAKES

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A preliminary version of the Technical Guidelines on monitoring and assessment of lakes prepared by the Core Group of UN/ECE Working Group on Monitoring and Assessment (WGMA) of Transboundary Watercourses and International Lakes, chaired by the Finnish Environment Institute, will be presented. The Technical Guidelines on Lakes are a continuation to the series of guidelines on the monitoring of the different types of transboundary water bodies prepared by the Working Group. Two of these guidelines have been published earlier: one for rivers in 1996 (and the revised version 2000) and the other for ground waters (in 2000). The general content of the guidelines has been approved in the first meeting of the Core Group on International Lakes in May 2000 in Helsinki, and discussed in the first meeting of UN/ECE Working Group on Monitoring and Assessment in Makkum, the Netherlands, in September 2000. The Technical Guidelines on International Lakes will be finalised during the year 2001, discussed in the second meeting of WGMA in September 2001 in Finland and published at the end of 2001.

INTRODUCTION

The Convention on the Protection and Use of Transboundary Watercourses and International Lakes (accepted in Helsinki, 1992) and its Protocol on Water and Health (accepted in London, 1999) include provisions on monitoring and assessment. The Convention contains also an obligation for riparian parties to harmonise their rules when setting up and operating the monitoring programmes. This obligation include also measurement systems and devices, analytical techniques, data processing and evaluation procedures. To support the implementation of this obligation UN/ECE Working Group (earlier Task Force) on Monitoring and Assessment has successfully produced two guidelines on the monitoring of transboundary water bodies, one on rivers (in 1996 and the revised version in 2000) and one on ground waters (2000).

The work to produce special guidelines on the monitoring of International lakes was started one year ago, in the Task Force meeting held in Bled in November 1999. A Core Group on Lakes was established in Helsinki-meeting at the end of May, 2000. The Finnish Environment Institute has been the responsible organisation in preparing draft papers for a continued handling. The high-grade experts from twelve countries have participated in the development of the guidelines. The first draft papers were discussed in the first meeting of the UN/ECE Working Group on Monitoring and Assessment (WGMA) in Makkum in September 21, 2000. The preparation of the guidelines will follow the decisions of the meeting and worthy advises of the delegates from different countries.

THE STRUCTURE OF THE GUIDELINES

During the preliminary discussions about the structure of the guidelines on lakes it was proposed to clarify the hierarchy of different guidelines and other publications. The guidelines on rivers and groundwaters contain both strategic aspects and technical details. The main part of these guidelines deals with strategic aspects of monitoring, and only some important topics concerning the detailed implementation of the practical monitoring programmes are discussed.
To clarify the functions of different publications of the UN/ECE Working Group on Monitoring and Assessment, it was proposed to separate the strategic guidelines from the technical ones (Figure 1). The highest level of publications (1) will be the Strategy Document, which is meant for politicians and persons, who are responsible for the general approval and implementation of the monitoring activity of transboundary Watercourses and International Lakes.

**Level 1:** The Strategy of Monitoring and Assessment of Transboundary and International Lakes

**Level 2:** The Technical Guidelines of Monitoring and Assessment of Transboundary and International Lakes

**Level 3:** The Background Papers
   - Paper 3.1: The Inventory of Transboundary and International Lakes
   - Paper 3.2: Experiences of the use of Guidelines (will be published later)

**Figure 1. Three levels of monitoring guidelines**

The next level (2) is made up of the different technical guidelines for the implementation of the monitoring programmes in the field and in the laboratories and finally in reporting. The technical guidelines consist of expert instructions and are useful information on different methods and practices. These guidelines shall be revised and renewed every now and then, e.g. when international demands for monitoring have changed.

The third level (3) consists of different background papers, which will be prepared either for revising the technical guidelines or for testing the new guidelines.

The Strategy Document and the Technical Guidelines can be published in one volume or separately. Further discussions in the UN/ECE Working Group on Monitoring and Assessment will clarify the final format of the guidelines on lakes.

**CONTENT OF THE STRATEGY DOCUMENT**

The strategy document is proposed to contain e.g. the following Chapters: Identification of river basin management issues, Information needs, General aspects for monitoring and assessment, General aspects for monitoring programmes, Data management and reporting, Quality management and Joint or co-ordinated action and institutional aspects. As conclusion the General recommendations will be presented.

**THE TECHNICAL GUIDELINES**

UN/ECE Working Group on Monitoring and Assessment of Transboundary Watercourses and International Lakes has already prepared and published the following two guidelines:

* Guidelines on water-quality monitoring and assessment of transboundary rivers
* Guidelines on monitoring and assessment of transboundary ground waters.

From the earlier background papers, the following two could also be evaluated as technical guidelines:

* Volume 3: Biological Assessment Methods for Watercourses

A proposal has been made to add a special volume Guidelines on Assessment and Presentation of Monitoring Results to the set of technical guidelines.

**BACKGROUND PAPERS**

UN/ECE Working Group on Monitoring and Assessment of Transboundary Watercourses and International Lakes has earlier prepared and published e.g. following background papers:
CONTENT OF THE TECHNICAL GUIDELINES ON LAKES

The Technical Guidelines on Lakes starts with Introduction, where topics as Lakes as a part of the water reserves in Europe, International demands for lake monitoring and assessment and Background information for lake monitoring are discussed. The guidelines will include separate Chapters concerning Hydrological and morphological features, Limnological features, Monitoring programme and The Assessment and presentation of the monitoring results.

In the Chapter Hydrological and morphological features, the Definition and classification of lakes and reservoirs, Morphology and hydrodynamics and Hydrological features are briefly discussed.

In the Chapter Limnological features the lake ecosystem is described. The ecosystem consists theoretically of two different parts; the biotope and the biocoenosis. The biotope is the abiotic part of the ecosystem. The primary quality characteristics of the biotope are determined by the properties of the drainage basin and the hydrological conditions.

The biotope can be described by a relatively small number of physical and chemical variables. In the sub-chapter Physical and chemical properties of lake waters the most important groups of the biotope are described: Optical properties (Secchi depth, Colour number), Thermal conditions (thermal seasons, stratification), The oxygen status (oxygen, organic substances, redox), The nutrient status (phosphorus, nitrogen), The salt status and The harmful substances (relevant heavy metals, organic compounds).

The living organisms of a lake form the biocoenosis, which can be characterised by the observations of the different groups of plants, animals and microbes. The algae and macrophytes are the major primary producers of the lake ecosystem. The sub-chapter Biocoenoses of the lakes gives a short summary of the central groups of the biocoenosis: Phytoplankton, Macrophytes, Periphyton, Zooplankton, Benthos and Fishes.

Sediment is a very important part of the lake ecosystem. In the sub-chapter Sediment as a sink and source of elements, topics such as Sedimentation and sediment rate, Internal loading, Toxic elements in sediment and Redox potential are discussed.

In the Chapter Monitoring programme all the relevant topics, which should be included in the monitoring programmes of international lakes, are discussed: Hydrological monitoring and modelling, Water quality monitoring, Monitoring of the biocoenoses of the lake, Monitoring of sediments and Monitoring methods.

In the sub-chapter Hydrological monitoring and modelling the main topics are Water balance components (precipitation, inflow, water level,...), Other observations (ice, erosion, sedimentation etc.), Hydrodynamics and Modelling and forecasting (hydrological cycle, inflow, water level, hydrodynamics).

In the Water quality monitoring programme of the International lakes the following objects should be handled: Selection of sampling sites, Frequency of sampling, Sampling strategy of the inflows/outflows, Vertical and horizontal stratification of the temperature, The physical variables, The oxygen status, The nutrient status, The salt status, Organic substances, The harmful substances and Radiological measurements.

The Monitoring programmes of the biocoenosis of the lakes consist of the following topics: Phytoplankton, Macrophytes, Periphytic growth, Zooplankton, Bottom fauna, Fishes and Bacteriological investigations.

The objects Sedimentation rate and the quantity of sediment, Internal loading (nutrients and metals), Measuring toxicity, and Redox-potential are the main topics discussed in the sub-chapter Monitoring of sediments.
The Guidelines also contains a chapter The Assessment and presentation of the monitoring results. In this Chapter the following items are discussed: Statistical methods, Classification systems, Indicators, Original data, data registers and maps, and Monitoring reports.

The general content of the technical guidelines has been approved in the first meeting of the Core Group on International Lakes, held in Helsinki in May 2000 and discussed in the first meeting of UN/ECE Working Group on Monitoring and Assessment in Makkum, the Netherlands, in September 2000. The Technical Guidelines on International Lakes will be finalized during the year 2001, and published at the end of 2001.

The preparations will be continued according the guidelines discussed in Makkum meeting. More attention will be paid to the variables, which are important especially in full filling the monitoring obligations presented in the EU Water Framework Directive. The directive came into force on December 22, 2000, and the implementation of the monitoring programmes according this directive shall be in complete operation in 2006.

REFERENCES

MONITORING FOR THE PLANNED BROWN COAL MINING
GARZWEILER II

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Garzweiler II is a brown coal mining project at the left bank of the Lower Rhine near Mönchengladbach – about 15 km from the Dutch border. From the year 2006 a total of 1.3 billion tonnes of brown coal will be extracted over a period of 40 years. Brown coal mining means a serious encroachment upon the water and ecological balance of the surroundings. To restrict the effects of brown coal mining an extensive catalogue of objectives and measures to protect water and nature balance has been drawn up. Monitoring means that all hydrological and ecological changes in and around Garzweiler II will be strictly observed. As a kind of early-warning system this should help to recognise any negative developments and reduce the risk of damage to the nature to be protected. Experts from around 50 various institutions are currently working on this monitoring system. In the course of monitoring a whole host of information is accumulating and ever increasing. This information will be entered into a well-structured and timely intranet for the parties involved. In the case of unforeseen and unacceptable developments in the water and ecological balance this information will form the decision-making basis for any need for action.

BROWN COAL MINING

From the year 2006 an brown coal mining project, Garzweiler II, is planned for the left bank of the Lower Rhine near Mönchengladbach – about 15 km from the Dutch border. The plan is to extract a total of 1.3 billion tonnes of brown coal over a period of approx. 40 years. Using

![Figure 1: Spatial dimension of the brown coal mining areas in the Rhine region](image-url)
mechanical excavation, brown coal will be mined over a total area of 48 km² and to a maximum depth of 210 m. Garzweiler II is one of five open mines in the brown coal mining area in the Rhine region. In future only the Inden, Hambach and Garzweiler mines will be of any significance. Figure 1 shows the spatial dimension of the brown coal mining areas.

Hydrogeology

The brown coal mining locations in the Rhine region are situated in the southern lower Rhine basin. Here during the tertiary and quaternary period gravel, sand, silt, clay and brown coal were deposited in alternating layers. The entire region is run through with vertical faults. Geologically speaking, we divide the region into the following blocks: the Rur block in the West, the central Erft block, the Cologne and Ville block to the East and the Venlo block in the North. Garzweiler II is situated in the Venlo block i.e. in the northern part of the mining area. The hydrogeological structure of this region is characterised by a succession of aquifers and aquicludes. The vertical faults at the block edges reduce the groundwater flow between the blocks. The normal depth to watertable, i.e. not influenced by brown coal mining, lies between a few decimetres and metres in the entire region. This means that the coal normally lies deep down in the groundwater. In the lower depths the groundwater is often confined.

Surface Mining Drainage

Brown coal mining in the Rhine region is carried out using mechanical excavation in open mines. To guarantee safe operating conditions on the bottom and stabilise the excavation slopes it is necessary to lower the groundwater below the mine bottom, which, in Garzweiler, means to around 210 m. This pumping is done by a group of several hundred wells around the mines. The distance between the wells varies between 60 and 120 m. The Garzweiler pumping water quantity is currently around 80 million m³/a and will be increased to 150 million m³/a. The groundwater surface is consequently lowered not only in the mining locations but also in the wide surrounding area.

Refilling the surface mines

During the extraction of brown coal a vast amount of overburden has to be moved. The overburden / coal ratio is 5:1, i.e. to achieve an average annual yield of approx. 40 million t at Garzweiler II, an average of approx. 200 million t overburden has to be shifted per annum. The overburden mass is dumped directly after mining, i.e. the mine is refilled apart from a remaining pit. This pit will then be developed into a lake with a surface area of 23 km², a maximum depth of 180 m and a water volume of approx. 2000 million m³. The refilling of this pit with diverted Rhine water is planned for the year 2030, directly after the exhaustion of the mine and over a period of 40 to 60 years.

Effects of brown coal mining on water and ecological balance

Brown coal mining means a serious encroachment upon the water and ecological balance of the direct and wider surroundings. The lowering of groundwater obviously influences the streams and their wetlands whose flora and fauna are dependent on water. In this case, the Schwalm-Nette area on both sides of the Dutch / German border is affected.

A further effect of brown coal mining is the impairment of groundwater quality by the dump material's hydro-chemical processes of acidification and their concomitant and consequential effects. The dump material contains geogenic sulphur compounds. Upon contact with the air, these oxidise releasing acids, sulphate and iron. The low pH-values mean that other substances such as heavy metal can also be mobilised. As the groundwater re-rises there is a danger of these substances being transmitted from the dump into the neighbourhood groundwater.

OBJECTIVES AND MEASURES TO PROTECT THE WATER AND NATURE BALANCE

To restrict the effects of brown coal mining and the measures involved an extensive catalogue of objectives has been drawn up (‘Braunkohlenplan’, Brown Coal Plan).

For instance, as far as lowering the groundwater is concerned, it has been determined that the hydrological characteristics of certain wetlands, dependent on groundwater, may not be altered. To protect wetlands and water supplies the lowering of the groundwater level around the mines will be compensated by infiltration and artificial re-filling measures. This will be
realised by infiltrating purified pumping water into the underground via seepage slits or wells. Water will also be channelled directly into some streams. The infiltration and channeling quantities currently amount to 40 million m\(^3\)/a and will be increased to approx. 80 million m\(^3\)/a for Garzweiler II. The amount of water to be purified will of course increase accordingly. In order to support the infiltration measures and the later re-filling of the remaining pit it is planned to divert Rhine water as of the year 2030.

With regard to the groundwater quality and the protection of local water supplies, sulphur compounds should, as far as possible, be prevented from oxidising in the dump, and the acids that are unavoidably released should be neutralised on the dump. To this end, lime is already being added to the dump substances as an acid-buffer. If need be, the release of substances will in future be further reduced by hydraulic measures, e.g. wells in the direct environs of the dump. The time aspect of all these measures can be seen in Figure 2. The measures already underway and those still in planning have a planning horizon of more than 100 years.

![Figure 2: Timetable of hydrologic measures for the Garzweiler II project](image)

**MONITORING GARZWEILER II**

**Definition and Tasks**

Monitoring is the systematic programming of spatial surveillance, control and assessment of all relevant hydrological and ecological parameters in and around the mining location Garzweiler II.

Monitoring Garzweiler II is divided into two phases: Concept and Execution. The transition from one phase to the other is smooth with constant feedback.

- In the Concept Phase the planning of the monitoring system, i.e. methods, environmental standards, surveillance routines and systems, are in the fore.
- In the Execution Phase the main focus is on surveillance, analysis, assessment and evaluation of all information.

Monitoring means that all hydrological and ecological changes in and around Garzweiler II will be strictly observed. The surveillance of the measures and technical installations is to control their effectiveness. As a kind of early-warning system this should help to recognise any negative developments and reduce the risk of damage to the nature to be protected.

The tasks and general objectives of the monitoring are:

- To quantify and determine the objectives listed in the Brown Coal Plan "Water and Ecological Balance".
- To examine the effectiveness of the compensatory measures and the achievement of the (quantified and determined) objectives of the Brown Coal Plan.
- The early recognition and/or prognosis of possible deviations from the objectives due to mining.
The compilation of contemporary and comprehensible information on the hydro/ ecological development both as a whole and in detail. This information forms the basis for the state authorities to assess the compliance with the Brown Coal Plan and the laws pertaining to water and waterways. This information is also available to the mine operating company for use in controlling the infiltration and refilling installations they are responsible for.

Organisation

Experts from around 50 various institutions are currently working on this monitoring system: among them ministries, local authorities, specialist state departments, the Water Board, universities, consultants and of course the operating company. The Province of Limburg, Hoofdgroep Milieu en Water Maastricht is also involved.

The basic technical knowledge is worked out in 6 smaller working groups dealing with: groundwater, wetlands / nature and landscape, surface water, water supply, overburden dump and remaining pit. All tasks are assigned to the respective field or working group. The working programmes and the evaluation of results are drawn up jointly in a Monitoring Com-mittee with representatives from all parties involved.

All agreements on working methods or essential working results are documented in a project manual. This is updated on a regular basis and forms the common basis for information and business. The Project Internet Homepage makes the manual available to all parties involved. In future monitoring results will also be included.

Concept and execution

The concept of the monitoring is basically to elaborate and administer, for each field of work, the respective monitoring strategy, environmental standards, demands on the measuring network and surveillance routines, methods of research and assessment and the evaluation of results, in accordance with the objectives set out in the Brown Coal Plan. The basic modus operandi is similar in all working groups. The findings from the individual groups are brought together to ensure integrated and comprehensive monitoring.

For monitoring Garzweiler II the existing infrastructure of measuring equipment will be used and extended. Approx. 3200 groundwater measuring stations from the various institutions, such as the Land (Land Groundwater Service), the Water Board (Erftverband) and the operating company (Rheinbraun) are currently available for the monitoring.

The Brown Coal Plan makes particularly high demands on the conservation of groundwater-dependent wetlands. Here an exemplary description of the respective monitoring concept:

The objectives for the wetlands according to the Brown Coal Plan are as follows:

- "In order to maintain the groundwater levels of the wetlands to be protected in the northern area, groundwater recharging is to be carried out ... The infiltration equipment is to be situated in such a way that the amount of infiltrating water flowing into the wetlands is as low as possible, but the water level nonetheless maintained ..." (Brown Coal Plan Chap. 2.1, Objective 3)
- "In the case of groundwater lowering caused by pumping, surface waters important for water supply or the ecological balance must be conserved. Outflow and water levels are, for instance, to be guaranteed by the direct channelling of pumped groundwater or diverted Rhine water, in-filtration measures and local surface water retention. A deterioration of the water quality must be avoided. The use of surface waters must continue to be possible without upsetting the eco-logical balance." (Brown Coal Plan Chap. 2.4)
- "The groundwater-dependent wetlands in the Schwalm-Nette region and along the streams draining to the Rur (Rothenbach, Schaagbach and Boschbeek) are to be conserved in their species-rich variety and character of groundwater-dependent symbiosis." (Brown Coal Plan Chap. 3.2, Objective 1)

To comply with these objectives, monitoring here comprises various components: large-scale regional groundwater monitoring and small-scale groundwater, surface water and landscape ecological monitoring for the wetlands.
The large-scale groundwater monitoring functions as an early warning system for changes in the groundwater levels in comparison to the calculated prognosis. The early warning system should ensure that any groundwater lowering is recognised in good time. There should be enough time to introduce countermeasures before the lowering reaches the wetlands under protection. For the early warning system the differences between the measured and the calculated groundwater levels are illustrated two-dimensionally every month, with a maximum delay of three months. The calculation is based on a combination of statistical processes and groundwater model calculations. (Bucher 1999).

The small-scale groundwater monitoring monitors the compliance with the threshold values defined for groundwater levels in wetlands. Here the wetlands are divided into compartments. For these an average annual groundwater level is calculated (threshold value), as would be the case in wetlands unaffected by mining. The calculation is based on approx. 20 to 30 measuring stations per compartment, as slight fluctuations at individual measuring stations cannot be unambiguously interpreted. A variation (as a rule a lowering) of more than 10 cm from the annual average signifies a critical change and thus a deviation from the objective. A lowering of more than 5 cm must be ascertained and documented.

The surface water monitoring provides for the quantitative and qualitative surveillance of waters important for the water supply or the ecological balance. The intensity of surveillance is graded according to criteria composed, on the one hand, of the water supply-ecological significance of the waters and floodplains and, on the other hand, of the expected influence and the possibility of alternative surveillance (e.g. of the groundwater level).

The small-scale landscape ecological monitoring comprises examinations of vegetation, fauna and limnology. The monitoring findings are then compared with the developments in reference wetlands outside the potential influencing area of the mines. The vegetation examinations are carried out along trans-sections and permanent observation areas. The trans-sections run at right angles to stream valleys and show the entire spectrum of the various local conditions in wetlands. The permanent observation areas describe the conditions at selected locations, e.g. in highly sensitive and valuable wetlands. A record is kept of plant species combinations and any changes to them, e.g. the respective coverage rate of the plants, increase/decrease in humidity, indicators for nutrients and disturbances which show external influences. The cause can, but need not be mining. From this observation of water plants in selected sections of stream water with a higher proportion of infiltrated water, predictions can be made on the long-term influence of a slow change in the water quality by an increased proportion of infiltrated water. These effects are otherwise hardly discernible. However, comparative data can only be had once further examination campaigns have been completed.

Ground beetles (Carabidae) are being examined in 10 selected trans-sections. The objective is the supplementary description of the overall ecological state of the wetlands, e.g. combination of species, proportion of humidity friendly species, predator / prey ratio etc.

RESUMEE AND OUTLOOK

Based on a clear catalogue of objectives (‘Braunkohlenplan’, Brown Coal Plan), a multitude of hydrogeological, hydrological and landscape ecological findings, the monitoring concept for Garzweiler II was almost completed in 1999. The execution phase has already been started. For the concept and execution of the monitoring the project manual, Monitoring Garzweiler II, has proved to be invaluable. Processing has always been transparent and orientated towards the common objectives. All parties involved have accepted the project manual as the basis for business, information and decision-making.

The parties involved are currently busy with questions of detail and above all with the draft of a detailed method documentation describing the methods and tools used for monitoring.

In the course of monitoring a whole host of information is accumulating and ever increasing. This information will be entered into a well-structured and timely intranet for the parties involved. In the case of unforeseen and unacceptable developments in the water and ecological balance this information will form the decision-making basis for any need for action.
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INTEGRATED MONITORING STRATEGIES FOR GLOBAL AND REGIONAL ASSESSMENTS

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Evaluation and assessment of freshwater quality at the regional and global scales requires a comprehensive integrated approach if meaningful relationships and results are to be derived. GEMS/Water has operated a comprehensive freshwater quality monitoring and assessment programme for over 20 years. A consistent problem in water quality assessment is the lack of coordination between monitoring programmes designed to study and evaluate different aspects of natural aquatic systems. Although individual activities may be successful, studies that require large-scale watershed approaches where the geographic area is shared by multiple nations becomes problematic. The integration of programme activities becomes critical where timing of sampling and analytical procedures is required for appropriate computations to be made. Computation of flux to the oceans from large rivers is of particular concern. GEMS/Water has implemented a programme element to co-ordinate station locations and sampling protocols on major rivers of the world to enable more accurate and precise flux computations to be obtained on a wide variety of parameters.

INTRODUCTION

For over 20 years the Global Environment Monitoring System for water (GEMS/Water) has operated as the water quality monitoring and assessment arm of the United Nations Environment Programme (UNEP). The primary means by which GEMS/Water has been able to achieve its international position has been, and continues to be, the direct interaction with key agencies and individuals in each participating country worldwide. By establishing a network of countries contributing data from national and in some cases, state, monitoring programmes, GEMS/Water has built a global water quality database for rivers, lakes (including reservoirs), and groundwater. Since 1998 the number of participating countries within GEMS/Water has increased to 101. Participation activity is variable between countries, but a significant increase in activity has occurred with the growing recognition that freshwater resources are approaching a crisis at the local, regional and global scales. New countries are joining the GEMS/Water programme and some countries that have not participated for several years are re-activating their membership and expanding their participation. Figure 1 depicts the current global national coverage of GEMS/Water.

National environmental and water resource agencies throughout the world carry out programme activities of GEMS/Water. Global distribution of participation is broad with one area

Figure 1: Participating GEMS/Water countries (shaded)
of major deficiency. The continent of Africa is under represented and therefore the broad picture of freshwater quality is not clear for this important region. Recognition of the deficit in global coverage has prompted development of an African initiative within the United Nations Environment Programme (UNEP). Part of the initiative will emphasize the development of increased participation in GEMS/Water throughout Africa to provide information on the status and trends of freshwater resources on the continent.

GEMS/Water participates in global water assessments and carries out research into regional and global freshwater quality issues (Fraser et al, 1995). Elements of the programme also address the development of technical capacity for participating countries to improve their national monitoring networks and capabilities through UN sponsored training courses in the field of water resources management, including technical courses for practical implementation of skills (WHO, 1991).

Currently the GEMS/Water database consists of approximately 1.4 million data points covering a parameter suite of over 100. Classification of the data on hand breaks into several broad classes including: indicators, major ions, nutrients, metals, microbiology, organics, and river flow. Geographic distribution of the data contained in the GEMS/Water database (GLOWDAT) is widespread with a higher concentration of stations in European countries, India, and Japan.

Monitoring programmes in participating countries contribute data to GEMS/Water for approximately 700 stations world wide (Figure 2). The station compliment addresses the three main target sectors for freshwater resources, 1) Rivers, 2) Lakes, 3) Groundwater. Station data stored in the GLOWDAT database is dominated by river data followed by lakes and then groundwater data. GEMS/Water does not actively undertake sampling programmes in countries but relies upon co-operative agreements with member states to provide data from their on-going water quality monitoring programmes. Stations are characterized into four classes: baseline, impact, trend, and flux. Flux stations are particularly important as they are used to gather data and information on transport rates between ecological regions; from the terrestrial environment to the marine, and trans-boundary between countries.

Comprehensive assessments of water resources require an integrated approach to monitoring where the local national scale becomes linked and amalgamated with information and data from other riparian or continental countries. This process sometimes necessitates the use of disparate data sets that reflect differing analytical and data management capabilities. When computations are required to produce information such as nutrient or organic pollutants flux from the terrestrial sphere to near-shore marine environments, further integration is required at the programme level to insure that data collection protocols are compatible. The overall integration strategy for water quality data requires extreme care and consideration. Normal statistical estimates using disparate data will obscure results. Examination of selected data sets from the GEMS/Water global database covering trans-national watersheds can be used to demonstrate the principles. GEMS/Water provides expertise and training to participating countries to assist in the development of water quality monitoring programmes at the national scale. Capacity building activities are designed to make data and information available to the United Nations more compatible and of higher quality.
FINDINGS AND DISCUSSION

The importance of the transport of freshwater and its accompanying suspended and dissolved constituents from the terrestrial regime to the marine environment has been recognized for many years (Milliman and Meade, 1983; Walling and Webb, 1983). Significant activities under the auspices of UNEP have specifically targeted this important component of integrated water resources management. It is necessary to increase the collaborative and strategic aspects of data gathering between the quality and quantity components of river monitoring.

GEMS/Water maintains close ties with the Global Runoff Data Centre (GRDC) in Germany, which operates a comprehensive database for the runoff of major rivers worldwide. Quality and quantity aspects of the two programmes have been linked to provide computation and assessment integration. In many instances, the flow characteristics of a riverine system have a dominating influence over the health and integrity of an ecosystem. It is for this reason that accurate and continuing measurements at gauging stations must be a priority in water resources management programmes and that siting strategies for optimal computation of hydrographs be designed to yield the best representation of flow dynamics.

Long-term monthly averages of the relationship between flow and suspended sediment concentrations in selected rivers at joint GEMS/GRDC stations indicate the potential benefit of enhanced programme integration. Box-plot distributions display the minimum, 10th, 25th, 50th, 75th, 90th percentiles and maximum of the frequency distribution. The Mekong River, which flows through China, Laos, Thailand, Cambodia, and Vietnam, exhibits highly significant wet/dry seasonality with a tight correlation to suspended sediment transport. The stretch of the river at Pakse, Laos shows the maximum suspended sediment concentration of 1526 mg/L (mean = 246 mg/L) corresponding with the maximum average monthly flow of 14290 m³/sec in August (Figure 3). Water quality data and information on the Mekong River has been assessed from the China border to the delta region (Hodgson and Fraser, 1997).

Figure 3: Flow and suspended sediment, Mekong River, Laos
In the case of the Neva River at Novosaratovka, Russia the constructed long-term hydrograph on a monthly basis shows the typical discharge pattern expected for a northern temperate river. Low flows in the winter followed by strong flow during the spring freshette that begins in March. The majority of the flow variance observed occurs in the second quarter of the year. Mean concentrations of suspended solids in the spring are approximately 7.0 mg/L falling with the summer flow and then increasing again as the autumn rains commence (Figure 4).

![Neva River hydrograph](image)

**Figure 4: Flow and suspended sediment, Neva River, Russia**

A particular problem in identifying water quality relationships arises when different laboratories operating under different programmes in neighbouring countries analyze data for the same parameter. Time series data were assessed for nutrients and physical parameters for major GEMS/Water rivers. Variances associated with spatial and temporal components of the distributions depict uncertainties in the data records and data patterns related to both geographic location and seasonal characteristics.

The River Rhone as it flows from Switzerland to France through Lake Leman shows the effects of both temporal and spatial variance (Figure 5). Conductivity was chosen in this case to eliminate the influence of biological activity. Long-term sampling shows annual cycling with differing variances in the data distributions. The region of low variance for Lake Leman (Station: 200006) shows the dampening effect on variance that occurs with lake systems. There were no significant differences in the analytical methods and procedures used therefore variances are dependant upon sampling location, seasonality and are dominated by the flow regime.

A more complex situation is faced when productivity processes are included due to the additional variance associated with cycling of nutrients. Total phosphorus concentrations in the Mekong River from Laos to Vietnam show high non-seasonal fluctuations. Variances are mainly attributed to agricultural practices and periodic flooding (Figure 6). There is a shift in the data record of approximately +0.05 mg/L P that is observable at the jurisdictional boundary between Cambodia and Vietnam. Step functions of this type are indicative of differing sampling and analytical practices between riparian countries. The Mekong study coordinated sampling and data handling activities in an integrated programme for all riparian countries. With this common base it is possible to identify differences in the data distributions that may not be related to actual changes in water quality.
Global and regional assessment programmes for water quality require integration at several levels. It is currently not possible to have all necessary activities operating under one strict protocol when working at the global level with multiple countries. This is particularly so when chemical, physical and biological components with associated sample meristics are required for the assessment. In bringing data together from multiple laboratories and jurisdictions the nature of the data and the capacity level of a contributing agency must be included in the assessment. This becomes more imperative where environmental and societal pressures unduly influence laboratory procedures and data management. It is not possible to demand that all agencies globally adhere to the highest tolerances for inclusion and integration of data. This being so, integrated monitoring and assessment activities must develop specific protocols and tolerances for data acceptance. The GEMS/Water protocol for data acceptance follows the following schema.

1. Receipt of data followed by acknowledgement notification to data originator.
2. Format conversion to GEMS/Water standard, if necessary.
3. Parameter code verification.
5. Database entry.

Figure 5: Time series for electrical conductance, River Rhone

Figure 6: Time series for total phosphorus, Mekong River
6. Application of parameter code and range filters to identify anomalies.
7. Manual code and parameter value revision if possible.
8. Statistical analysis for consistency of received data set.
9. Comparison of new data set with data previously received.
10. Preparation of data report.
11. Data report and queries sent to data originator for confirmation and clarification.

The feedback mechanism with data originators continues until all anomalies are clarified. This does not remove unusual outliers from the data set but rather insures that the data held are accurate or are flagged to identify anomalous characteristics. The process is similar for existing data in the database. Suspect data that is identified due to non-standard techniques or broad tolerances is flagged and is not used in standard statistical analyses.

As an adjunct to the development of monitoring capability at the national scale, quality assurance and quality control programmes operating at the global level are required. GEMS/Water has operated a QA/QC programme for participating country laboratories for many years. The results of this effort have been variable but overall the quality of data received improves with the efforts to increase country capacity and capability.

In large-scale comprehensive assessments there is a tendency to overlook the limitations of data and proceed with technological constructs that aggregate data or represent analogues of environmental conditions. Such constructs may overestimate or underestimate the true situation leading to positions and recommendations that are not firmly based. Data aggregation techniques are appropriate where the importance of individual empirical values is not significant. In cases where threshold values are important such as toxics and drinking water health criteria it is the excursion value, duration and frequency of occurrence that is significant. Aggregation techniques and the use of analogues mask the detail necessary to identify possible water quality problems. Sufficient data must be taken to identify variances so that accurate estimates of conditions can be made without undue aggregation. To avoid the pitfalls of unwarranted data aggregation it is imperative that sufficient physical, chemical, and biological monitoring activities required for integrated studies receive support. Programmes that address the requirements for large-scale integrated assessments require international funding sufficient to allow for effective and efficient operations. There is no substitute for real data.

Future directions of the water quality and quantity monitoring activities of the GEMS/Water programme will emphasize the joint requirements for data acquisition needed for the Global Runoff Data Centre and the GEMS/Water Collaborating Centre to contribute to water quality assessments carried out by United Nations agencies (UNEP, WMO, UNESCO and WHO). Strong international co-operation to provide good geographical coverage and reliable and timeous data at the global scale is needed to meet the requirements to improve our knowledge of the state of the world’s inland waters and their impacts on the marine environment. A major thrust will be made to provide detailed estimates of material fluxes from primary continental watersheds to the marine environment. As the meshing of the monitoring activities accelerates, the overall importance of the databases as sources for regional and global assessments will increase.

REFERENCES


Milliman, J. D. and Meade, R. H. "World-wide delivery of river sediment to the oceans". In *J. Geology*, 91, 1, pp. 1 - 21, 1983.


MONITORING OF THE SURFACE WATERS IN REPUBLIC OF MOLDOVA AND INTEGRATION IN EUROWATERNET

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Water quality monitoring is a crucial part of environmental management. Though monitoring is expensive, it is accepted now that a regional approach is the most suitable solution in dealing with the multifactor and interrelated ecological and economic management of the environment. Monitoring and information-collecting activities are continuously being developed in order to respond to increasing demands for objective, reliable and comparable data on the conditions of waters that could serve as a basis for decision-making and management. Description of the present structure of surface waters monitoring in the Republic of Moldova is given.

This paper focuses on how to build up the national monitoring networks using the EUROWATERNET techniques and guidelines, in order to fulfill the information needs (quality of surface water, ecological quality, reduction and control of pollutant discharges, eutrophication and acidification), to provide up-to-date and comparable information.

INTRODUCTION

Actually many Global Institutions and National Governments have declared water as a top priority with regard to the natural resources of the future. In many parts of the world the water scarcity threatens national and international security, as well as economic, environmental and social stability. One other important point of reference here is the fact that investments in providing clean water have very high economic, social and environmental returns everywhere. Environmental pollution, taking into account its social-economic consequences and the wide spreading sphere, is a problem with both national and international implications, especially when the pollution sources are close to other country boundaries, bringing about undesired effects upon the neighbouring country.

Performance in areas such as water pollution and soil degradation is difficult to appraise since there are no quantitative targets, nor the necessary data. The progress toward targets is slow by the reason that problems are dealt with separately and the interrelation between environmental problems and their causes are not addressed to full extent.

More comprehensive or integrated approaches are therefore required toward the management and assessment of these problems.

Monitoring progress related to the integration of such systems into the management of environmental problems is difficult. However, some indicators of progress already exist. The main barriers to further progress with systems integration are the lack of scientific knowledge and information about the interrelation of environmental problems, the lack of targets to measure policy performance, as well as the gap between the scientific disciplines and the political institutions that deal with different environmental impacts.

At present, some of the systems used to monitor and collect information about the environment are inefficient and wasteful. They generate excessive amounts of data and information related to other subjects. So, there is an urgent need for policies to ensure better focused information, consistent environmental assessment and reporting. It is also worth to mention the need for concerted efforts in order to streamline environmental monitoring and practices; to focus information collected on key issues; to develop indicators, which should be widely agreed and recognized; to underline the significance of environmental change and the progress of sustainability.
METHODS

Current practices

In the Republic of Moldova, water resources monitoring is made in the framework of the National System for Water Quality Monitoring. This system comprises about 40 stations of first grade of importance (Figure 1). By this term we mean important control section located usually downstream of the large pollutant sources. There are also a large number of control stations of the second degree of importance.

Actually in our country monitoring activities cover only a relatively modest part of the total hydrographical network. The network also covers several lakes (reservoirs) (Figure 2). The main characteristics of Moldavian rivers and reservoirs are shown in Tables 1 and 2 respectively.

Figure 1. River Monitoring Network
Figure 2. Lakes (reservoirs) Monitoring Network
**Table 1  General characteristics of the main rivers of the Republic of Moldova**

<table>
<thead>
<tr>
<th>River</th>
<th>Length, km</th>
<th>Area, km²</th>
<th>Mean water flow, m³/sec</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Transboundary rivers</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Dniester*</td>
<td>630</td>
<td>19070</td>
<td>339</td>
</tr>
<tr>
<td>Prut*</td>
<td>695</td>
<td>7990</td>
<td>92</td>
</tr>
<tr>
<td>Danube*</td>
<td>0,9</td>
<td>3</td>
<td>5500</td>
</tr>
<tr>
<td>Ialpug</td>
<td>142</td>
<td>3180</td>
<td>2,9</td>
</tr>
<tr>
<td>Cogalnic</td>
<td>125</td>
<td>1030</td>
<td>9,8</td>
</tr>
<tr>
<td><strong>Internal rivers</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Reut</td>
<td>286</td>
<td>7760</td>
<td>9,9</td>
</tr>
<tr>
<td>Bac</td>
<td>155</td>
<td>2150</td>
<td>2,9</td>
</tr>
<tr>
<td>Botna</td>
<td>152</td>
<td>1540</td>
<td>1,1</td>
</tr>
<tr>
<td>Cainari</td>
<td>113</td>
<td>835</td>
<td>1,4</td>
</tr>
<tr>
<td>Ichel</td>
<td>101</td>
<td>814</td>
<td>0,7</td>
</tr>
<tr>
<td>Cubolta</td>
<td>100</td>
<td>947</td>
<td>1,6</td>
</tr>
<tr>
<td>Ciuhur</td>
<td>97</td>
<td>724</td>
<td>0,7</td>
</tr>
</tbody>
</table>

* - only for the Republic of Moldova territory

**Table 2  General characteristics of the main reservoirs of the Republic of Moldova**

<table>
<thead>
<tr>
<th>Reservoir</th>
<th>River associated</th>
<th>Area, km²</th>
<th>Volume, mln m³</th>
<th>Length, km</th>
<th>Maximal depth, m</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dubasari</td>
<td>Dniester</td>
<td>67,5</td>
<td>485</td>
<td>128</td>
<td></td>
</tr>
<tr>
<td>Costesti-Stanca</td>
<td>Prut</td>
<td>92,0</td>
<td>1085</td>
<td>90</td>
<td>34,2</td>
</tr>
<tr>
<td>Cuciurgan</td>
<td>Cuciurgan</td>
<td>27,3</td>
<td>88</td>
<td></td>
<td>7,0</td>
</tr>
<tr>
<td>Taraclia</td>
<td>Ialpug</td>
<td>15,1</td>
<td>62,0</td>
<td>8,5</td>
<td></td>
</tr>
<tr>
<td>Congaz</td>
<td>Ialpug</td>
<td>4,9</td>
<td>9,9</td>
<td>5,0</td>
<td>3,4</td>
</tr>
<tr>
<td>Comrat</td>
<td>Ialpug</td>
<td>1,7</td>
<td>2,60</td>
<td>2,85</td>
<td>2,6</td>
</tr>
<tr>
<td>Mingir</td>
<td>Lapusna</td>
<td>2,6</td>
<td>12,2</td>
<td></td>
<td>5,0</td>
</tr>
<tr>
<td>Ghidighici</td>
<td>Bac</td>
<td>8,0</td>
<td>40</td>
<td></td>
<td>7,0</td>
</tr>
<tr>
<td>Ialoveni</td>
<td>Isnovat</td>
<td>4,4</td>
<td>21,7</td>
<td></td>
<td>12,0</td>
</tr>
<tr>
<td>Ceaga</td>
<td>Ceaga</td>
<td>2,8</td>
<td>4,1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Costesti</td>
<td>Botna</td>
<td>1,8</td>
<td>3,3</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rezeni</td>
<td>Botna</td>
<td>1,7</td>
<td>4,0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cheazevca</td>
<td>Sarata</td>
<td>1,0</td>
<td>2,8</td>
<td>2,5</td>
<td>5,9</td>
</tr>
<tr>
<td>Sarata-Noua</td>
<td>Sarata</td>
<td>1,54</td>
<td>2,28</td>
<td>2,5</td>
<td>2,6</td>
</tr>
<tr>
<td>Zgurita</td>
<td>Cainari</td>
<td>0,95</td>
<td>1,7</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Caplani</td>
<td>Caplani</td>
<td>1,5</td>
<td>8,3</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Tarnovo</td>
<td>Racovat</td>
<td>1,02</td>
<td>1,87</td>
<td>2,6</td>
<td>4,0</td>
</tr>
<tr>
<td>Badrajii-Vechi</td>
<td>Racovat</td>
<td>1,0</td>
<td>4,9</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cupcin</td>
<td>Ciuhur</td>
<td>0,82</td>
<td>1,27</td>
<td>2,4</td>
<td>2,6</td>
</tr>
</tbody>
</table>
The existing water quality monitoring network consists of 4 monitoring subsystems. The institutions responsible for monitoring of water resources are:

**Ministry of Environment and Territory Development**
- Service Hydrometeo – surface water quality, quantity;
- State Inspectorate of Environment Quality (Districts Ecological Agencies) – surface water quality, waste water;
- National Institute of Ecology – collection and processing of water monitoring data, elaboration of monitoring concepts, strategies;

**Ministry of Health**
- Sanitary-Hygienic National Centre (Districts sanitary-hygienic services) – surface waters and shallow groundwater quality (quality control of drinking water by chemical and bacteriological parameters)

**Ministry of Agriculture**
- State Water Management Consortium "Apele Moldovei" – surface water quality (characterisation and modality of using in national economy)
- Agency of Geology of Moldova "AGeoM" – groundwater quantity and quality

**Academy of Sciences**
- Surface water quality, groundwater, fundamental studies in the frame of State Programmes

Each organization follows its specific goals and operates according to specific monitoring programmes concerning sampling sites, sampling frequency, analytical equipment and methods. Pollutant identification is achieved by analysing water samples in the laboratories. The quality indicators to be analysed are established depending on the specific activities in the respective zone. The water quality measurements include about 30 hydrochemical (Table 3) and 5 biological (macroinvertebrates, phytoplankton, zooplankton, bacterioplankton, fish) determinands. These indicators are divided in two categories. The first category includes a restrictive number of the very important indicators, which give a rapid evaluation of water quality by daily analysis. The second category comprises indicators that are monthly or weekly analysed.

There are several types of quality control practices such as internal, preventive and statistical control of measurements. Recently, laboratory management and analyses are made in conformity with the ISO standards. Laboratories also participate in relevant external interlaboratory quality control schemes involving the distribution of check samples. We can mention in this respect EQUATE, Qualco Danube and Aquacheck.

**RESULTS AND DISCUSSION**

The sites for monitoring surface water quality have been selected close to main wastewater discharge sources (Figure 1). The national river monitoring programmes include stations on the major rivers, while the number of stations on small rivers and reference sites is rather low. Many of the small rivers can not be monitored. Small rivers are, however, ecologically important. Their relative size might be more at risk from human activities than reaches of larger rivers. Therefore, it is necessary to establish stations covering small river catchments and reference areas. There is a shortage in terms of spatial location of reference stations.

Great efforts are also being made to upgrade the monitoring system. But we still need to optimise the monitoring network in terms of gauging sites and programmes. At the present moment several projects are under way. They aim at re-evaluating the existing system of surface water monitoring and building capacity for the monitoring of sediment quality. Action-oriented monitoring is also envisaged in order to obtain more information about causes and effects in water bodies affected by eutrophication, acidification and micro-pollutants. Irrigation water quality monitoring will be reorganized to take into consideration the measuring sites of the food-quality measuring system.

There was no systematic information exchange between Ministries and data information network. At present all data from monitoring programmes will be included into a common database managed by Centre of Ecological Monitoring.
Various efforts in research and development have been undertaken to achieve comparability of data, and to provide a comprehensive picture of the complex interactions and interlinkages within and between environmental media within catchment areas or parts thereof. There is growing evidence that the availability of water-quality and water-quantity data in Europe is improving, and progress is being made in the introduction and use of harmonised data collection, reporting and assessment procedures. The extent of the economic and environmental relevance of transboundary watercourses and international lakes is clearly demonstrated by ongoing activities.

EUROWATERNET is a network, which will provide information on water quantity as well as water quality issues. The network is designed to give a representative assessment of water types and variations in human, industrial and agricultural pressures across the European Environment Agency area. This will provide up-to-date and comparable information and will ensure that similar types of water body are compared. The need to compare like-with-like is achieved with 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 will help to ensure that valid status comparisons will be obtained.

### Table 3 Monitored hydrochemical parameters

<table>
<thead>
<tr>
<th>Monitored parameter</th>
<th>Method of determination</th>
<th>Sensitivity</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temperature</td>
<td>Thermometry</td>
<td>0.1°C</td>
</tr>
<tr>
<td>pH</td>
<td>pH-meter</td>
<td></td>
</tr>
<tr>
<td>Mineralization</td>
<td>calculating, gravimetrically</td>
<td></td>
</tr>
<tr>
<td>Suspended solids</td>
<td>Gravimetrically</td>
<td></td>
</tr>
<tr>
<td>CO₂</td>
<td>Volumetrically</td>
<td>1 mg/l</td>
</tr>
<tr>
<td>O₂</td>
<td>volumetrically (Winkler)</td>
<td>0.05 mg/l</td>
</tr>
<tr>
<td>BOD</td>
<td>volumetrically (Winkler)</td>
<td>0.05 mg/l</td>
</tr>
<tr>
<td>COD</td>
<td>volumetrically (K₂Cr₂O₇)</td>
<td>5 - 50 mg/l</td>
</tr>
<tr>
<td>% of saturation</td>
<td>Calculating</td>
<td></td>
</tr>
<tr>
<td>HCO₃</td>
<td>Potentiometrically</td>
<td>0.5 mg/l</td>
</tr>
<tr>
<td>SO₄</td>
<td>volumetrically (Pb(NO₃)₂)</td>
<td>1 mg/l</td>
</tr>
<tr>
<td>Cl</td>
<td>(more reaction)</td>
<td>1 mg/l</td>
</tr>
<tr>
<td>Ca</td>
<td>volumetrically (EDTA)</td>
<td>0.5 mg/l</td>
</tr>
<tr>
<td>Mg</td>
<td>Calculating</td>
<td></td>
</tr>
<tr>
<td>Na</td>
<td>Photometrically</td>
<td>1 mg/l</td>
</tr>
<tr>
<td>K</td>
<td>Photometrically</td>
<td>1 mg/l</td>
</tr>
<tr>
<td>Hardness</td>
<td>Compenometrically (edta)</td>
<td>0.5 mmol/l</td>
</tr>
<tr>
<td>Phenol</td>
<td>(amidopirinum)</td>
<td>0.005 mg/l</td>
</tr>
<tr>
<td>Oil products</td>
<td>Fluorometrically</td>
<td>0.02 mg/l</td>
</tr>
<tr>
<td>Detergents</td>
<td>(methelene blue)</td>
<td>0.015 mg/l</td>
</tr>
<tr>
<td>Organoclorine pesticides</td>
<td>Gas cromatography</td>
<td></td>
</tr>
<tr>
<td>(DDT, DDD, DDE)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>N-NH₄</td>
<td>Nestler reaction</td>
<td>0.002 mg/l</td>
</tr>
<tr>
<td>N-NO₂</td>
<td>Gris reaction</td>
<td>0.007 mg/l</td>
</tr>
<tr>
<td>N-NO₃</td>
<td>Photometrically</td>
<td>0.01 mg/l</td>
</tr>
<tr>
<td>P-PO₄</td>
<td>Photometrically</td>
<td>0.005 mg/l</td>
</tr>
<tr>
<td>P-total</td>
<td>Photometrically</td>
<td>0.005 mg/l</td>
</tr>
<tr>
<td>Copper</td>
<td>Photometrically</td>
<td>0.002 mg/l</td>
</tr>
<tr>
<td>Silicon</td>
<td>Photometrically</td>
<td>0.5 mg/l</td>
</tr>
<tr>
<td>Zinc</td>
<td>Photometrically</td>
<td>0.002 mg/l</td>
</tr>
</tbody>
</table>
One of the key concepts of EUROWATERNET is to use the existing national monitoring and information databases. Countries should not develop two incompatible monitoring and assessment systems. On the contrary, they should be enabled to have a common source of information for different needs.

Moldavian monitoring system could serve as a part of EUROWATERNET which will incorporate information fully compatible with the reporting needs of national programs. The key questions to be answered by Eurowaternet, such as: quality of surface water, ecological quality, reduction and control of pollutant discharges, eutrophication and acidification, are mentioned as high priority in the national programmes as well.

However, it is likely that a flexible approach will also be required for the selection of different types of monitoring stations (reference, representative and impact) included in national networks, especially for the lakes monitoring network, in order to be able to answer some specific questions. It will, therefore, be expected that some stations will fulfil more than one task. River stations and lakes should be selected from the total station population using the stratification criteria in accordance with Eurowaternet design guidelines.

In order to make the information fully statistically representative, as part of the development of Eurowaternet, the precision and confidence obtained from different numbers of stations are being assessed. An optimal number of stations could be established to characterise the river catchment (1 station per 1000 km²).

The number of stations in the national lakes monitoring program in Moldova is in general lower (6 representatives) than the proposed number (17 representatives and 2 references). Small lakes also are not monitored.

Some useful stages in creating a platform for development and implementation of EUROWATERNET have been completed. We can mention here the efforts of our country to establish a statutory and regulatory framework and the creation of procedures for preventing, reducing and managing accidental pollution of transboundary waters, the bilateral and multilateral agreements and other steps.

The environmental policy of Moldova aims at more close relationship with relevant international activities by improving international co-operation on environmental protection at the regional and European levels; implementing bilateral agreements and participating in regional programmes. Moldova participates in several international agreements in relation with water management and management of transboundary watercourses such as the Convention on Sustainable Use and Protection of the Danube River Basin. Since 1993 Moldova is a Party to the ECE Convention on Environmental Impact.

The pollution of transboundary watercourses and international lakes has become widespread in some European countries. For decades, these waters have played an important economic role without particular thought being given to the notion of preventing, controlling and reducing transboundary impact.

Especially for the downstream countries, it is very important to receive information about all kind of transboundary pollution having an effect on the water quality. Due to the fact that Moldova is a downstream country, it was important for us to establish bilateral agreements with the upstream countries to effectively cope with transboundary pollution. These bilateral agreements contain, in particular, provisions on appropriate mutual exchange of all pertinent information, joint contingency plans. Moldova has ratified the Convention on the Protection and Use of Transboundary Watercourses and International Lakes.

Our country is located in the drainage basin of the Black Sea. Moldova is thus interested and involved in the Black Sea protection initiatives. In particular, Moldova ratified the Agreement among the Governments of the Participating States of the Black Sea Economic Co-operation on collaboration in Emergency Assistance and Emergency Response to natural and man-made Disasters. This will enable us to elaborate relevant intergovernmental investment programmes. To enhance the protection of waters and reduce transboundary impact requires reliable data.

We are sure that EUROWATERNET will contribute to enhance effectiveness of joint measures to protect and manage transboundary waters. This will be in line with the objectives of the Convention on the Protection and Use of Transboundary Waters and International Lakes and others international agreement.
Moldova is in the course of drawing up new strategies and performing a continuous activity for harmonisation and approximation of its legislation to the European Union standards, as well as to the provisions of international law on water and environment protection.

WORK STILL TO BE DONE:

- to optimise the monitoring network in terms of reference station;
- to reorganise the lakes monitoring network;
- to develop the contingency planning and appropriate methodologies and mechanisms;
- to modify the current analytical procedures so that comparable data can be obtained;
- to implement the proposed (EUROWATERNET) monitoring network in a step-by-step process, further develop and test in Moldova as a volunteer country;
- to organize training courses under the aegis of EUROWATERNET.

REFERENCES

**ADMINISTRATIVE CAPACITY TO PERCEIVE LONG-TERM TRENDS IN WATER QUALITY. CONSEQUENCES FOR TRANSBOUNDARY RIVER MANAGEMENT**

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**ABSTRACT**

The administrative capacity of a transboundary river commission to perceive long-term trends in water quality was studied by focussing on inter-annual variability in the concentrations of the major nutrients $N_{tot}$ and $P_{tot}$. Time series of annual data from a number of sites in the Vechte catchment in the Dutch/German border area, were processed by applying linear regression of $10\log$-transformed annual concentrations, for periods where such a model seemed appropriate. The standard deviation in the residuals (s) was used as a parameter for inter-annual variability in detrended time series of 14-23 years. European environmental water management, as formulated in the EU-Water Framework Directive(2000), prescribes the submission of progress reports, at 6 years intervals, categorising water quality in the river catchment into five quality classes. The 95% confidence intervals around nutrient concentrations, averaged for these 6 year intervals, were compared with the class ranges, both on a $10\log$-scale. For $N_{tot}$ and $P_{tot}$ these intervals corresponded with circa $1/4$ and $1/2$ of the class ranges. With the same measure for inter-annual variability, the statistical power to detect a step-trend from one reporting period to the next was calculated. Modal values for $s$ in the catchment were 0.05 and 0.10 for $N_{tot}$ and $P_{tot}$ respectively, resulting in step-trends of $x^{-1.19}$ and $x^{-1.41}$ respectively. The consequences of the crude categorisation of water quality for transboundary management are discussed.

**INTRODUCTION**

Environmental monitoring is the process of repetitive observing, for defined purposes on one or more elements of the environment according to prearranged schedules in space and time and using comparable methodologies for environmental sensing and data collection (Chapman 1992). To communicate monitoring data on water quality to others, for reasons of comparison and evaluation, categorisation is used to express in a most accessible way how good or bad our waters are. Once classes have been defined, a major question is how definitively a water body can be categorised into one of these classes, given the data available and the inter-annual variances around possible long-term trends in water quality. It is to be expected that inter-annual variabilities in for instance nutrient concentrations diminish once data are aggregated for larger units of management, such as river basins. River basins, in many cases transboundary river basins, are the management units of the new European Water Framework Directive (WFD).

Mid 2000 the European Parliament and the Council of the European Union adopted the WFD (EU 2000). The five major purposes of the WFD are: (i) to enhance the quality of aquatic ecosystems and of terrestrial ecosystems and wetlands depending on them, (ii) to promote sustainable water use, (iii) to establish progressive reduction of emissions, (iv) to reduce the pollution of groundwater and (v) to contribute to mitigating the effects of floods and droughts. The environmental objective to achieve within 15 years, is "good (surface) water" status for all water bodies. The qualification "good" can only be reached when both the ecological status and chemical status of the waterbody are at least "good". For rivers good chemical status is still loosely defined on the basis of the physico-chemical quality elements, e.g. "nutrient conditions do not exceed the level established so as to ensure the functioning of the ecosystem and the achievement of values specified for the biological elements". The WFD is strongly ecosystem-oriented and applies to four categories of water: inland surface waters, transitional waters like estuaries, coastal waters and groundwater bodies. Its purposes...
should be reached by a great number of activities to be organized and co-ordinated at (international) river basin level. One of them is the establishment of a monitoring programme in each Member State by the year 2006 to provide coherent and comprehensive overviews of water status in each river basin. In 2009 management plans for each basin must have been drafted and these plans will be reviewed and updated every 6 years. These plans must describe the river basin, its sources of pollution, its water quality problems and the measures taken to solve these problems, including their financing. In each plan maps have to be drawn indicating the status of the water bodies monitored, into five categories, each with its own colour code. In Europe most river basins larger than 10,000 km² cover territories, which belong to at least two countries (Verhallen et al. in press). The significance of accomplishing integrated river basin management, exceeding one or more administrative borders within a river basin, is that co-ordination must prevent numerous discontinuities in e.g. monitoring networks, data collection, sample treatment and analysis etc. The WFD prescribes surveillance monitoring for significant bodies of water crossing a Member State boundary.

The Vechte basin is a downstream sub-basin (3800 km²) of the River Rhine, the latter which in the terminology of the WFD is an "international river basin district". The river has its source in Germany in the Federal State of Nordrhein-Westfalen, flows into the Federal State of Niedersachsen and then to the Netherlands where it empties in the 200,000 ha Lake IJssel. Since 1977 water authorities from the two German States and from the Netherlands co-operate in the monitoring of the water quality of the River Vechte and one of its main tributaries the River Dinkel in the border area of the two countries (van Dijk et al. 1992, Petry & Schimmer 2000). The results of the monitoring are published under the shared responsibility of a Dutch/German Vechte Commission in their progressive management plans (Anonymous 1992, van Hezewijk et al. 1997).

For the communication on trends in water quality and for the decision-making on water quality management by the Vechte Commission, as being the highest level in the administration, usually only highly aggregated data are used. As according to the WFD, "good chemical status" should be reached for the whole river basin within 15 years, it is most important to be able to follow progress made as good as possible from long-term downward trends in nutrient concentrations. Strong inter-annual variability weakens the statistical power for detecting such trends.

The aim of this paper is to evaluate the annual data on \( N_{\text{tot}} \) and \( P_{\text{tot}} \), collected in the Vechte basin at the German-Dutch borders for their inter-annual variability (noise) and so for the capacity of the Vechte Commission to perceive possible trends (signal). Inter-annual variabilities, as estimated throughout the border area, are then used to calculate 95% confidence intervals around annual figures for nutrient concentrations. These confidence intervals are then compared with the ranges in nutrient concentration per quality class as applied in water quality management in the Netherlands. In this way we could assess the statistical power of the administration to follow trends in water quality (Lettenmaier 1976, Peterman 1990), and the resolution and usefulness of their categorisation into quality classes as prescribed by the EU-WFD.

**STUDY AREA**

The River Vechte is a medium-sized rain-fed, lowland river with a catchment area of 3800 km², a length of 177 km and a total elevation difference of 106 m (Grabs, 1997). The basin is inhabited by about 1.5 million people. Most of the land (75%) is used for agricultural purposes (maize and grass mainly), followed by forest and nature (20%) and urban area (5%). About 1800 km² of the basin is located in Germany. The river rises in the Baumgebirge in Nordrhein-Westfalen with a characteristic steep gradient of two-thirds of the total elevation along 25% of its length. The main upstream tributaries are the Dinkel and the Steinfurter Aa. The Dinkel is a typical sandy, lowland stream, rising in Nordrhein-Westfalen. After 38 km it transborders the Netherlands, flows 43 km over Dutch territory, then back into Germany in the federal State of Niedersachsen. This tributary with a length of 103 km has an agricultural dominated catchment of 642 km².

In the Dutch part of the river basin two-third of the loading with \( N_{\text{tot}} \) originates from agricultural sources, via leaching and runoff, and one-third from waste water treatment plants with industrial point sources being negligible (Anonymous 1997). Although such estimates were not made for the German part of the basin, the national figures for Germany on point and non-point
sources refer to the same relative contribution of approximately 2:1. The loading with $P_{tot}$ originates from agricultural sources (50%), effluent (47%) and from point sources. Water quality classes in the Netherlands, as based on concentrations of $N_{tot}$ and $P_{tot}$, are given in Table 1.

Table 1 Reference values for water quality in Dutch inland waters as based on nutrient concentrations (Witmer 1998, Miedema & van Dijk 2000)

<table>
<thead>
<tr>
<th>Quality</th>
<th>$Range_{P_{tot}}$ (mg/l)</th>
<th>Log class width $P_{tot}$</th>
<th>$Range_{N_{tot}}$ (mg/l)</th>
<th>Log class width $N_{tot}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Very bad</td>
<td>$&gt; 0.75$</td>
<td></td>
<td>$&gt; 11.0$</td>
<td></td>
</tr>
<tr>
<td>Bad</td>
<td>0.45-0.75</td>
<td>0.22</td>
<td>6.6-11.0</td>
<td>0.26</td>
</tr>
<tr>
<td>Poor</td>
<td>0.30-0.45</td>
<td>0.18</td>
<td>4.4-6.6</td>
<td>0.18</td>
</tr>
<tr>
<td>Moderate</td>
<td>0.15-0.30</td>
<td>0.30</td>
<td>2.2-4.4</td>
<td>0.30</td>
</tr>
<tr>
<td>Good</td>
<td>0.05-0.15</td>
<td>0.48</td>
<td>1.0-2.2</td>
<td>0.34</td>
</tr>
<tr>
<td>High</td>
<td>$&lt; 0.05$</td>
<td></td>
<td>$&lt; 1.0$</td>
<td></td>
</tr>
</tbody>
</table>

MATERIAL AND METHODS

The monitoring programme in the border area is maintained via German-Dutch co-operation and dates back to 1977. The programme consists of the frequent monitoring of nutrient concentrations at 27 sites throughout the border area and the annual monitoring, since 1982, of macrofauna in the same streams and river reaches. The 11 major sampling sites in the border area, selected here for assessing inter-annual variability in the major nutrients $N_{tot}$ and $P_{tot}$, are located in watercourses, with stream order 1 to 3 (Figure 1). The sampling frequency at most sites was at least monthly, but at sites 3, 6 and 8 frequency was considerably lower. This may have enlarged inter-annual variability as due to pronounced seasonality in nutrient concentrations (Figure 2).

Figure 1 Catchment area of the River Vechte and of its major tributaries Dinkel and Radewijkerbeek, together with the locations of the 11 sampling sites. For the names of the sites see Table 2.
Figure 2 Monthly concentrations of $N_{tot}$ and $P_{tot}$ at site 1 (averaged over the years 1981-1990) and site 10 (averaged over the years 1980-1991).

The annual averages for concentrations of $N_{tot}$ and $P_{tot}$ were 10log-transformed before linear regression on time, for a series of years, which showed clear non-linear developments of these concentrations over time. The distributions of the residuals after log-transformation did not depart from normal. Inter-annual variability in nutrient concentrations ($C_t$) was parameterised with the standard deviation ($s$) in the residuals around the regression:

$$s = \sqrt{\frac{\sum (10\log C_i - 10\log C_i)^2}{n-1}}$$

A change in water quality, as a step trend ($\Delta$) in average annual nutrient concentrations, can be parameterised, after log-transformation with:

$$\Delta = |\log C_2 - \log C_1| = \log a$$

where $\log C_{av}$ is the average of 10log-transformed annual nutrient concentrations during a 6-years reporting period for the EU-WFD, and $a$ is the multiplier in average concentrations between the two periods. The statistical power (1 - $\beta$) to detect a true step trend ($\Delta \neq 0$) in 10log-transformed annual concentrations, so correctly rejecting a false $H_0$ ($\Delta = 0$), can be inferred from the test statistic (modified after Zar 1984):

$$t_{p,v} = \frac{|\log C_2 - \log C_1|}{se} - t_{a,v}$$

where $\alpha$ is the critical value for making a Type I error and $\beta$ the resultant probability for making a Type II error, both with $n$ degrees of freedom. The standard error (se) in the difference between the two averages, where the variance ($s^2$) and the number of years ($n = 6$) in the reference period and in the period with a new average are the same, is estimated with:

$$se = s \sqrt{\frac{2}{n}} = s \sqrt{\frac{1}{3}}$$

The statistical power for perceiving a change in average annual concentrations is than obtained from the t-value test with $v = 2(n - 1)$ degrees of freedom then becomes:

$$t_{p,v} = \frac{\Delta}{s} \sqrt{\frac{3}{\beta}}$$
The difference in annual concentrations, indicated with the multiplier a (= 10D), which can be perceived with statistical power (1 - b) = 0.90, under conditions a = 0.10 and n = 6 years, can be graphed as a function of inter-annual variability s (ta = tb = 1.372): a = 101.58s. Calculations were made for one-tailed test, so for a instead of a/2, because one expects improvements in water quality.

RESULTS

Water quality management in the catchment of the River Vechte since 1977 has been most successful in reducing Ptot-concentrations (Figure 3). Around 1977 the major watercourses still fell in quality classes "Very bad" or "Bad", but at the end of the 1990s they had reached "Moderate" or "Good" status, although with sometimes large inter-annual variability. In the River Vechte, site 1 had reached almost "Good" status by 1999. The sudden increase in Ptot concentration in 1995 at sites 4 and 10 in the major tributary Dinkel is attributed to the renovation of a waste water treatment plant upstream of site 10 around that time. The quality at sites 2, 5, 6 and 9 in the smaller tributaries was "Moderate" in the 1980s already, and has improved further since then.

![Figure 3 Developments in Ntot and Ptot at sites 1, 4 and 10 in the catchment of the River Vechte. The horizontal lines indicate the lower boundary of quality class "Very bad" and the upper boundary of quality class "High", dashed lines for Ntot and straight lines for Ptot.](image)

Inter-annual variability (s) per site was estimated for time series of 13 to 23 years. The variabilities in Ptot and Ntot per site seem to be grossly correlated and variabilities were always higher for Ptot than for Ntot (Table 2, Figure 4). Lowest variabilities were estimated for the River Vechte (site 1), the highest ones for a site (6) in a 1st order stream, which was sampled with low frequency. Stream order seems to be the most relevant here, as there were no significant relationships between inter-annual variabilities and the average concentrations during the selected period (R² Ntot = 0.004, R² Ptot = 0.002). The residual variance in nutrient concentrations at site 1, the only site for which annual figures on river runoff were available, could not be explained from the inter-annual variability in river runoff (R² = 0.0064 for Ntot; R² = 0.018 for Ptot), and thus stands as it is.
**Table 2** Geometric mean (GM) concentrations (mg/l) and inter-annual variabilities (s) in nutrient concentrations, as estimated from the residuals around linear regression of 10log-transformed annual averages per sampling site. Periods were selected after visual inspection of the plots for linearity.

<table>
<thead>
<tr>
<th>Site</th>
<th>( N_{\text{tot}} )</th>
<th>( P_{\text{tot}} )</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>GM</td>
<td>s</td>
</tr>
<tr>
<td><strong>Vechte catchment</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 Vechte*</td>
<td>7.9</td>
<td>0.023</td>
</tr>
<tr>
<td>2 Radewijkerbeek*</td>
<td>6.2</td>
<td>0.048</td>
</tr>
<tr>
<td>3 Wielener Moorgraber</td>
<td>28.0</td>
<td>0.081</td>
</tr>
<tr>
<td>4 Dinkel-down*</td>
<td>9.2</td>
<td>0.028</td>
</tr>
<tr>
<td>5 Geele beek*</td>
<td>4.6</td>
<td>0.047</td>
</tr>
<tr>
<td>6 Rammelbeek</td>
<td>5.4</td>
<td>0.152</td>
</tr>
<tr>
<td>7 Punteek*</td>
<td>6.2</td>
<td>0.056</td>
</tr>
<tr>
<td>8 Goorbeek</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>9 Ruengerbeek*</td>
<td>8.0</td>
<td>0.048</td>
</tr>
<tr>
<td>10 Dinkel-up*</td>
<td>12.2</td>
<td>0.030</td>
</tr>
<tr>
<td>11 Glanerbeek**</td>
<td>14.6</td>
<td>0.109</td>
</tr>
<tr>
<td><strong>Annual report on water quality in the Netherlands</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Large rivers</td>
<td>4.86</td>
<td>0.018</td>
</tr>
<tr>
<td>Lake IJssel</td>
<td>3.38</td>
<td>0.027</td>
</tr>
<tr>
<td>North Sea – coastal</td>
<td>0.61</td>
<td>0.077</td>
</tr>
<tr>
<td>North Sea – offshore</td>
<td>0.12</td>
<td>0.097</td>
</tr>
</tbody>
</table>

*At least monthly monitoring, **Monthly until 1992, ***1976 and 1996 had extremely low values.

Figure 4 Inter-annual variabilities in \( P_{\text{tot}} \)-concentrations plotted on those for \( N_{\text{tot}} \) for a series of waterbodies. Watercourses in the Vechte catchment are marked by their stream order. "Large rivers" have stream order 4 and higher.
For comparison, we estimated inter-annual variabilities in time series of annual nutrient concentrations in Dutch waterbodies as communicated to the general public. Graphs of these time series can be found in the freely available annual progress report, with basic data on an accompanying CD (CIW 1999). The nutrient concentrations in this report are averaged over individual rivers, each of stream order 4 or higher, into one administrative category "Large rivers". Inter-annual variabilities in the de-trended series for this broad category were only slightly lower than for site 1 in the River Vechte, and also here variability was higher for P\text{tot} than for N\text{tot} (Table 2, Figure 4). Inter-annual variability estimated for the 2000 km\textsuperscript{2} Lake IJssel, comprising circa half of the total surface of Dutch inland waters, was as low. Inter-annual variability in nutrient concentrations averaged for a limited number of sites in the Dutch part of the North Sea, was particularly large for N\text{tot}, which is the limiting nutrient in the marine environment.

If during a 6-years reporting period for the EU-WFD, water quality would neither increase nor decrease, the 95% confidence interval (CI) around the 6-years average would on a log-scale correspond with:

\[
95\% CI = \frac{4s}{\sqrt{6}} \text{ or } 1.63 \, s
\]

Referring to Figure 4, and taking inter-annual variabilities \(s = 0.05\) for N\text{tot} and \(s = 0.10\) for P\text{tot} as representing a moderate situation in the Vechte catchment, the 95% confidence intervals then would be 0.08 and 0.16 respectively. These intervals correspond with circa 1/4 and 1/2 of the average class intervals (log-scale) in Table 1. The conclusion is that categorisation of water quality on the basis of P\text{tot} concentrations could only be crude, for lower order streams especially.

With the same moderate values for inter-annual variability (s), step trends in N\text{tot} as large as \(^{*}:a = 1.19\) and in P\text{tot} as large as \(^{*}:a = 1.44\) and can be detected with a warranted amount of statistical power \((1 - \beta) = 0.90\) (Figure 5). On a 10\textsuperscript{log}-scale, these step trends correspond with 1.58 \(*:s\), so again with around 1/4 and 1/2 of the interval size of an average water quality class for N\text{tot} and P\text{tot}. It implies that the statistical power for categorising water quality is significantly larger for N\text{tot} than for P\text{tot} (Figure 6).
DISCUSSION

Every two year the Dutch-German Transboundary River Commission reports on the progress made in the water quality of the border streams and of the main River Vechte. Time series of annual concentrations per sampling site are depicted along a linear scale. These series date back to 1977, so they can be evaluated in a progressively larger time window. Monthly concentrations over the last two years are also given. Comments on the graphs are made on the basis of visual interpretation of the graphs, without referring to statistical characteristics. Comments on the short monthly series are made by scoring the frequency with which limit values are exceeded. In an executive summary remarks are made referring to two different - Dutch and German - quality standards and to the bilaterally agreed common limit values.

For the 2 year progress report to be effective, not only in reporting on streams and sites violating critical values in the recent past, but also in showing general trends in the long-term, more attention should be paid to inter-annual variability in water quality. The progressively longer time series of annual data, which now enable evaluation in a time window of more than 20 years already, are an indispensable element of the progress reports. But for proper comparisons to make throughout the area, and for building awareness on characteristic differences in accuracy of the nutrient concentrations per site, it needs standardisation of inter-annual variability on a log-scale. Such awareness is particularly relevant for those at the higher levels in the administration and certainly for those in the Transboundary River Commission, who have a responsibility to evaluate developments in water quality, now still referring to two national standards and with the EU-standard coming up.

A major question left is whether inter-annual variability can be explained, in part, from inter-annual variabilities in river runoff and in nutrient supply, originating from agriculture and from waste water treatment plants. If so, this would reduce unexplained variability and would thus enlarge the accuracy in categorising water quality and reduce the minimum size of a detectable step trend. The renovation of the waste water treatment plant upstream of site 10 in 1995 could be traced from a strong increase in P\text{tot} in the tributary Dinkel (Figure 3). It is worth trying to evaluate a possible relationship between P\text{tot}-concentrations recorded in the catchment and the...
annual output of the sewage plants. Agriculture is certainly the major source of N_{tot} (Cook 1998). Co-variance in nitrate levels in rivers and in fertiliser applied in the area, however, is yet hard to prove (Home & Goldman 1994, their Table 16-3). The scope for reducing unexplained variability in the annual concentration of N_{tot} seems therefore limited. Further studies in the Vechte basin should possibly focus more on inter-annual variabilities at sites, which are sampled with low frequency, and where intra-annual variability thus carries over on inter-annual variability.

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PESTICIDE MONITORING

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Pesticide monitoring cannot be done in a universal way. Different monitoring goals may require different monitoring program designs. Examples of different goals are: the monitoring of emissions, the relationship between agricultural activities and water quality, or the suitability of river water quality for specific uses. Pesticide monitoring also is expensive and financial constraints often cause a pesticide-monitoring program to be a compromise. The most expensive monitoring program however, is the one that is not well considered beforehand! The "accordion method" is a new monitoring concept for water quality. It can cope with rapid changes in the occurrence of polluting substances in surface water (including pesticides). The method consists of a combination of a broad survey and specific monitoring. The viability of this approach was demonstrated in 1999. A total of 106 (including 81 pesticides) out of 374 contaminants were found in the main waterways in the Netherlands. Among them were several "new" pollutants, whose occurrence was not known before. Based on this survey, "new" substances should be added to the specific monitoring program, while others, which are no longer relevant, could better be left out. For pesticide monitoring in surface waters, the "accordion method" can give more relevant information at lower costs.

INTRODUCTION

Monitoring itself does not improve water-quality. If performed in a proper way, it can be the source of valuable information. However, monitoring of pesticides in surface water is not so simple. The chemical analysis of modern pesticides is complicated and expensive. An additional complication is that the application of agricultural pesticides, (the most important source of pesticide pollution in surface waters), is characterised by frequent changes in both the quantity and type of chemicals used.

Monitoring objectives, i.e. the anticipated use of monitoring results, can vary. The purpose of a monitoring project can be, for example, to get insight into the relationship between a certain activity (e.g. the application of certain pesticides on a specific area of land) and the resulting occurrence in surface water. Another example is to get insight in the relationship between a certain activity and surface-water quality. Although these two objectives seem very similar, their differences will lead to a different set-up of these two monitoring programs. Of course there are many other possible objectives for monitoring, e.g. getting more general information about surface water-quality and the levels of pollutants, or the verification of the suitability of surface water for specific uses (e.g. drinking water production). Another objective may be the collection of data that can be used by politicians for the enforcement of new legislation or which could be used to convince farmers and agricultural organisations of the necessity to take measures to reduce pesticide pollution. In this situation, not only the reliability, but also the force and persuasiveness of the results of the monitoring are paramount.

Although monitoring can even have more purposes than the ones mentioned here, it is obvious that differences in the purpose of monitoring have effect on the set-up of the monitoring program. Figure 1 presents a chain of relations between a certain activity (e.g. pesticide use on a specific area of land) and the downstream river water quality.

Depending on the monitoring objectives, (i.e. the questions that need to be answered) it has to be decided which part of the chain has to be monitored and which pesticides should be monitored. Within the European Union, there are over 800 different substances marketed as active ingredients of so-called plant protection products. From an analytical point of view, it is (almost) impossible to analyse all these substances and the costs would be prohibitive. It is therefore unavoidable to make a selection. Knowledge about the types, quantity and periods of application of pesticides actually used by farmers in a certain region would be of great help when making this selection, but often this information is not available or is even considered confidential.
So we see that the set-up of a monitoring project is depending on the local situation and dictated by the goals of the project. Bearing this in mind, several choices have to be made for the set-up of a pesticide monitoring program like the choice of parameters to be analysed, sampling frequency, locations, precision and accuracy needed, etc. Different monitoring purposes may lead to a different set-up of a monitoring program. Once this set-up has been made and the sampling has been carried out accordingly, it is not unusual if the outcome is of little use for other purposes. Or in other words: there is not one universal system of pesticide monitoring.

THE ACCORDION-PROJECT

With the objective to collect information about the quality of surface waters in general and to assess whether water quality criteria are met, RIZA has performed pesticide monitoring in the main waterways in the Netherlands for a long time. For this purpose, a fixed selection of about 50 individual pesticides has been measured year after year at regular intervals and at a number of locations. This selection was made several years ago, based on the knowledge of pesticides and the analytical state-of-the-art at that time. As a result of the static character of the program, shifts in the use of individual pesticides over the years and changes in pesticide use within a growing season (e.g. due to unfavourable weather conditions or sudden plagues), were not reflected in the outcome. More and more it was felt that this program had become a static routine, that no longer gave sufficient information about the actual surface water quality. In addition, there were strong indications that there were rapid changes in the actual use of pesticides by the agricultural sector, both in quantity and in the chemical composition of the pesticides used. These changes were partly caused by ongoing developments in the chemical industry, but – very significant – also as a result of the national and EU regulations for the placing of plant protection products on the market (91/414/EEC). Detailed information about these changes in agricultural pesticide use was scarce or even not available at all. Sales-figures of individual pesticides were considered as confidential information by the Dutch Ministry of Agriculture and were not available to the water-authorities.

In this situation, a new monitoring method was developed: the so-called "accordion-method". The objective of this method was to gain up-to-date water-quality information about a variety of substances, including the occurrence of a rapid changing group of pesticides. The main objectives of the project were to improve flexibility, to save costs and to achieve actual and relevant information about the occurrence of pesticides in surface waters. The accordion-method consists of two stages. The first stage is a broad survey, in which the presence of as many substances as possible was examined (the accordion wide, pulled out). Analytical developments during the last few years in GC-MS and LC-MS technology made it possible to analyse a great number of substances in one analytical run, while still maintaining a reasonably low detection limit. (Often 0,01 µg/l or lower). The second stage is the actual monitoring phase, in which during a certain period the analytical efforts are focused on a limited number of substances, whose relevance was shown during the survey (accordion closed, folded in). The broad survey phase of this two-stage concept was new. It explains why the most attention was given to this part during the project.
In 1999, the viability of the concept was tested in practice in the main waterways in the Netherlands. Samples of water and suspended solids were taken at seven carefully selected locations (figure 2). At each location, the sampling took place in three different periods during the agricultural season (i.e. in April, June and August), which resulted in a total of 21 samples. In the laboratory, the waters were chemically analysed for 374 different organic compounds (345 of them were pesticides) whereas the suspended solids were analysed for 244 compounds (of which 232 were pesticides).

Fig 2 Locations selected for the surveys

RESULTS OF THE SURVEYS

In total, 106 different compounds were found in water; 81 pesticides, 8 pesticide-metabolites, and 17 industrial contaminants. In the suspended solids, where detection limits are higher than in water, 28 compounds were detected; 20 pesticides and 8 industrial contaminants. Among these compounds not only pesticides registered in the Netherlands (NL) were found, but also pesticides of which the use was allowed only in the upstream neighbouring countries Germany (D) and Belgium (B). Very remarkable however, was the demonstrated occurrence of 5 pesticides that are not registered in any of these three countries. (See figure 3)

The analytical results of the project were compared to water-quality criteria. This showed that pesticides were responsible for exceeding the Dutch water-quality criteria (MPC, Maximum Permissible Concentration) in 57 % of the 21 surveys. The pesticide-criterion of max. 0.1 µg/l,
for drinking water, was exceeded in 75% of the surveys. Due to the large number of pesticides investigated, for the first time it was also possible to compare the outcome of the measurements with the (drinking water-) criterion that the sum of all pesticides in total should not exceed 0.5 µg/l. This criterion was exceeded in 40% of the surveys.

In the samples of suspended solids, in all cases the Dutch quality criteria (MPC) were exceeded, due to too high concentrations of PCB’s and hexachlorobenzene (industrial contaminants). In 20% of these samples, the concentration of pesticides exceeded the criteria as well.

When comparing the transboundary pollution at the border-crossings of the river Rhine (Lobith), the river Meuse (Eijsden) and the Gent-Terneuzen canal (Sas van Gent), it was clear that the largest number of contaminants (80) were found in the Gent-Terneuzen canal and the lowest in the Rhine (35). See figure 4 to 6.
The survey results from the seven locations were compared with the water quality criteria and the drinking water criteria of 0.1 µg/l. The clearest demonstration of the differences in pesticide-contamination of the different waters/locations is shown by the total concentration of pesticides (figure 7). The highest level of pollution is found in the Gent-Terneuzen canal (Sas van Gent) and the Scheldt-river (Schaar v. Oudendoel). Comparing the two locations Keizersveer and Eijsden (both situated along the river Meuse, in which Keizersveer is located some 200 kilometres downstream of Eijsden), it is also visible that the agricultural activities in the Netherlands cause an increase of the total pesticides concentration, in spite of dissipation processes like sorption, breakdown and evaporation.

Based on the outcome of the survey, substances were selected for inclusion in the future monitoring program. Three criteria were used for this selection: 1) demonstrated excedance of the water-quality criteria (MPC), 2) excedance of the drinking-water criterion, and 3) substances found at least five times (out of the 21 samples). 46 out of the 106 different substances...
measured in water, met one or more of these three criteria and were recommended for future monitoring. When comparing these 46 recommended substances with the current monitoring program, it turned out that 26 out of these 46 substances are not incorporated in the current monitoring program. On the other hand, 20 substances out of the current monitoring programme are not relevant anymore and could better be left out.

CONCLUSIONS

There is no universal system of pesticide monitoring. A program for pesticide monitoring starts with a clear definition of its purposes. Different purposes of monitoring will lead to a different set-up of a monitoring program. Choices have to be made about parameters to be analysed, frequency of sampling, locations, amount of persuasiveness needed, etc.

Pesticide monitoring is expensive. Financial constraints cause the set-up of a monitoring program to be more or less a compromise. The most expensive monitoring program however, is the one not well considered beforehand!

With the purpose of monitoring water-quality information about a rapid changing variety of substances (pesticides in particular), RIZA developed the concept of the "accordion-method." This is a combination of a broad survey and the specific monitoring of a limit number of substances.

The survey demonstrated the validity of the concept and showed the occurrence of no less than 106 contaminants, including 81 pesticides, in the main waterways in the Netherlands. Several "new" pollutants, whose occurrence was not known before, were demonstrated.

Five of the pesticides were not registered in the Netherlands or in the neighbouring countries Belgium and Germany. It is unlikely that conventional monitoring methods should have revealed their existence. At 71 % of the locations, either the water quality criteria (MPC) or the drinking water criterion (0.1 µg/l) or both were exceeded due to too high concentrations of one or more substances.

The use of a broad survey gives a nearly complete view of the state of pollution. For the first time it was also possible to check if the drinking water criterion for the sum of all pesticides (0,5 µg/l total pesticides) was met, because of the large number of pesticides that were investigated.

The outcome of the survey proved to be a useful tool as a selection criterion for specific monitoring. The use of a survey can even save money, as substances can be left out from the specific monitoring program, which are no longer relevant. In particular for pesticide monitoring, the use of the accordion method may result in a monitoring program that goes along with rapid changes in agricultural pesticide use. With this method, monitoring surface-water quality can give more and more relevant information at lower costs.

REFERENCES

THE COMBINED USE OF GROUNDWATER MONITORING AND MODELING TO ENSURE A RELIABLE LONGTERM WATER SUPPLY IN HEAVILY INDUSTRIALIZED AREAS

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The groundwater resources in heavily industrialized regions are influenced by numerous sources of contamination. Long term supply of drinking water from groundwater resources is impossible without protective actions and specific water treatment. For planning and optimizing those activities, information on the state, the processes and the trends of the groundwater system are required. Model based studies are also necessary to predict system reaction to possible encroachments like sand-mining or an increase of water abstraction.

As shown by the example of Ludwigshafen on Rhine, the combined use of modeling and monitoring evolved into a useful toolbox for system analysis as well as planning, optimizing and control of actions. This toolbox was continuously improved by adjusting it to a complex multilayered ground-water flow system. Using modern data base technology, an integrated groundwater information system has been formed to handle future problems in groundwater management.

INTRODUCTION

Ludwigshafen on Rhine has become one of the worlds major locations of chemical production sites (e.g. BASF, KNOLL, RASCHIG, GIULINI, ALCOA). With the growths of industry and township there was an increase in public and industrial water demand. Water supply has been developed using local groundwater resources.

Figure 1: Location of drinking water wells on the "Parkinsel"
Concerning water quality there is potentially a great conflict between groundwater abstraction and sources of contamination resulting from urban, industrial and agricultural landuse. Specific for Ludwigshafen are
- several chemical waste dumps,
- expanded deposits of chemical waste, partially used to raise the land-surface out of groundwater influence,
- spills due to handling of chemicals and
- spills due to the influence of war.

The drinking water wells are situated on an island in the Rhine river between the cities of Ludwigshafen and Mannheim, the center of the densely populated and industrialized "Rhine-Neckar-region" (Figure 1).

The aquifer system, part of the "Rhine-Graben-System", is some 100 m thick and consists of alluvial deposits. They were deposited in a sequence of highly permeable sand and gravel deposits and less permeable layers of silt and clay resulting in a highly complex aquifer system (Figure 2).

![Figure 2: Generalized hydrogeological structures in the Rhine-Neckar-Region](image)

From the beginning of the 20th century until the 1950s drinking water was pumped out of the upper aquifer, which is about 20 m deep. As a result of groundwater contamination with phenolic sub-stances, later on pumping was relocated to deeper aquifers. At that time the first main aquitard was assumed to be a safe barrier against contaminant transport. Now about 7 million m³/a are pumped out of 20 wells, mainly screened between 40 m and 100 m below ground.

**HISTORICAL DEVELOPMENT OF MONITORING AND MODELING**

**Realizing the necessity of monitoring**

The increasing sensibility for environmental problems in the 1970s and memories of the problems in the 1950s gave an idea of the possible threat to the supply of clean water. The need for monitoring the catchment area of the drinking water wells has been realized early by the water supply company. Up to that time monitoring consisted of:
- extensive monitoring of piezometric heads in the upper aquifer by state and urban authorities,
- analysis of drinking water quality according to the regulations of the public health department. (Since 1976 regulated by German "Trinkwasserverordnung"; later on by EC drinking water regulation).
First steps of monitoring and modeling

In the following "monitoring a catchment area" - in an expanded but useful definition - covers collecting all information necessary to understand, describe, quantify and evaluate the state of and the processes in the groundwater system. In Ludwigshafen specific monitoring started with registration of possible sources of contaminations such as dumping sites and chemical industry sites by the water supply company.

![Figure 3: Areas that were suspected of being contaminated (in 1982)](image)

As shown in figure 3, several such sites were found in the vicinity of the wells, but having little knowledge of the groundwater system, assessing or designing an appropriate system of monitoring wells was not possible.

At that time the first model-based investigations of the groundwater-flow-regime in the "Rhine-Neckar-region" were performed. According to the low level of information and the limited technical possibilities, a simplified model concept with two (upper and lower) aquifers was used (Figure 9). This gave a first idea of long term groundwater flow and the impact of the groundwater abstraction in lower aquifers to the flow in the upper aquifer. As early as 1982 a numerical transport model was coupled to that flow model (Zipfel (1983)). It was used to evaluate the possible risks for the drinking water wells, posed by sites suspected of groundwater contamination (Figure 4).

Based on that first concept of groundwater flow and transport a system of monitoring wells was installed and a basic measuring programme designed. This was done by the cooperation of several concerned institutions, authorities and consultants. With a few shallow observation wells down gradient of possible contamination areas, first information about the emissions from the contaminated sites were gathered. At one of those sites nearby the drinking water wells, heavy pollution with aromatic and chlorinated hydrocarbons was detected.

**Further work – an iterative process of investigation, monitoring and modeling**

The next tasks for groundwater management were risk assessment (including investigation of contaminant spreading), and the design, efficiency control and optimization of protective actions. They were achieved in iterative steps of continuously improved modeling and investigation/monitoring activities (Figure 5). At first, contaminant spreading and pathways were investigated. A dense location of multilevelled observation wells covering the whole region of interest would have been too expensive due to the complex groundwater system. Again a stepwise procedure proved to be the best way. Model based interpretation of measures of...
water quality and piezometric head helped to detect pathways with changing flow directions in different aquifers and weak zones in the aquitards.

As contaminations proved to endanger drinking water wells seriously, protection wells were installed in a median aquifer (Figure 7). They should capture contaminants before reaching the drinking water wells. The abstraction rates and the configuration of the protective groundwater withdrawal had to be optimized. Drinking water abstraction should be safe on the one hand, but flow of clean water to the wells not be hindered on the other.

This optimization required a more detailed description of the median aquifer in the model. As result of a model system with one model layer for all deeper aquifers, abstraction rates of about 2 to 2.5 million m$^3$/a at the protection wells would have been necessary. Using a more detailed model, finally the shield withdrawal was dimensioned to 0.8 million m$^3$/a (about 12% of drinking water abstraction).

Figure 4: Model-calculated spreading of possible groundwater contaminations in the upper aquifer (1982/1983)

Figure 5: Continuous improvement of monitoring and modeling
Operating the protective pumping, there is a demand for
- controlling the efficiency of the protection measures permanently
- optimizing the pumping step by step with the aim of best protection for minimum costs.

Near the contamination source in the upper aquifer, remedial actions with pump and treat measures were installed by the liable institutions. There the water supply company is an advisory member of the management committee.
THE CURRENT SITUATION

Current monitoring network

Now there exists a sophisticated monitoring system with a mixed approach of:
- widespread monitoring of catchment area and surroundings
- emission monitoring down gradient of contaminated sites (usually done by the liable persons or institutions) and
- quality control in the vicinity of the pumping wells.

Corresponding to the multilayered groundwater system, monitoring wells are screened in 7 different levels (Figure 7). At present, the monitoring network of the water supplier consists of 250 monitoring wells (including emergency water supply wells).

![Figure 8: Monitoring wells in the vicinity of the "Parkinsel"](image)

Current monitoring program

There is a sophisticated hydraulic monitoring programme with
- short intervall measures of hydraulic head with dataloggers in regions with short-term changes (near wells or river Rhine)
- long intervall (usually monthly) manual measures in other areas.

The water quality monitoring programme consists of
- catchment area covering measures on a set of indicator parameters (like AOX, DOC, EC) in annual intervals
- supervision of the vicinity of the pumping wells with monthly to half year measures on indicator and region specific contamination parameters (like aromatic and chloroaromatic hydrocarbons)

Groundwater information system

A client-server database system is used for data storage and evaluation. It is based on ORACLE as database engine using different standard software as front-ends, like EXCEL, ACCESS and ARCVIEW. Standard Data evaluations are:
- Hydrographs, especially for supervision of groundwater stages in the different aquifers
- concentration development graphs for evaluation of trends
- concentration distribution maps

There is data exchange with other institutions with monitoring activities in Lud-wigs-hafen, e.g. the water authorities. At present there are activities to enlarge the information system to a service for several institutions in Ludwigsafen like urban authorities and companies of the chemical industry. In the information system not only measured data are stored, but also information about geology, land use and precipitation.
Current groundwater model system

The additional detailed information about the hydrogeological structures resulted in a further development of the model reproducing the real groundwater system with its flow characteristics.

Now the regional groundwater model consists of 13 different layers (Figure 9). Model information is held in a specific information system. Model mesh size is adapted to small scale problems by a telescopic mesh refinement with – up to now - a smallest applied mesh size of about 30 m.

![Figure 9: Adjusting the modeling-system to the hydrogeologic system](image_url)

Topical problems in groundwater management

Currently the modeling and monitoring-toolbox is used to deal with the following problems:

- The future development of water quality has to be predicted to get basic information for the design of a new water treatment plant. Just now a suitable transport model has been designed and monitoring data are being evaluated. One difficult question is, whether protective actions and natural attenuation already have stopped progression of contamination spreading also from other contaminated sites.

- The concept of drinking water supply is reviewed fundamentally. Alternative concepts like re-location of pumping activities to shallow wells near river Rhine with the purpose of river bank infiltration or to other regions are checked on technical, economic and safety criteria.

CONCLUSIONS

The long term supply of drinking water from groundwater in industrialized areas is often endangered by contamination. Decision making in groundwater management by water supply companies is impossible without sufficient information on the state, the processes and the trends of the groundwater system. In Ludwigshafen on Rhine, a sophisticated information system using a combination of modeling and monitoring has evolved in the past 25 years.

While monitoring gives raw data and information, modeling produces a better understanding of system procedures. This understanding of the system is necessary for making monitoring tailor made, which also means to be reduced to the required minimum.
In planning processes, modeling is a standard tool for simulation of system reactions to different influencing actions, like groundwater withdrawal. Optimization of actions in complex groundwater systems is possible only with well adjusted models and good monitoring requires a well adjusted monitoring system. An iteration of modeling and monitoring proved to be the best way for the adjusting process.

Monitoring is more than just having a set of certain system parameters like water level or pH on the screen. Monitoring must give all information on the properties of and the influences on the ground-water system. For example geology, geochemistry and land use are all aspects of monitoring.

REFERENCES


DESIGN OF A REGIONAL GROUNDWATER MONITORING NETWORK:
THE PRISMAS PROJECT EXPERIENCE

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The main purpose of the PRISMAS National project is the definition of standard procedures for designing and exploiting a groundwater-monitoring network over a large region. ARPA-Umbria, within the PRISMAS project, is performing the quantitative and qualitative monitoring of the main alluvial and carbonate aquifers of the Umbria region (Central Italy). The monitoring network is composed by 12 remote stations and 220 wells and springs which are surveyed by means of four seasonal campaigns per year. The data collected both by the punctual monitoring and by the remote monitoring flow into a relational database, which manages their acquisition and processing allowing the continuous update of the network. All the data stored are easily accessible and can be extracted for further specific elaborations. The combination of the punctual monitoring data with the data of the continuous remote system allows the study of the chemical and hydrogeological variations (natural or man-induced) occurring in the studied aquifers in order to differentiate the seasonal changes from the long-period trends (more than one year) and the diffuse contamination processes from the local pollution episodes.

INTRODUCTION

In the Umbria Region the protection of groundwater resources is a high priority environmental concern. Many groundwater-quality problems are areally dispersed and may be widespread and frequent in occurrence. Examples include problems associated with the application of agricultural chemicals, leaks in sewers, septic tanks, animal feedlots, the aggregate effects of many different point-source pollution events in urban areas, and natural, geologically related water-quality problems.

The PRISMAS National Project, funded by the Ministry of Environment and by the Region of Umbria, started in 1997 and its main purpose was the definition of standard procedures for designing and exploiting a groundwater-monitoring network at the regional scale. These procedures would be adopted as a reference for countrywide applications. Within this context ARPA-Umbria started from February 1998 the monitoring of the principal aquifers of the Umbria region with the following specific objectives:

– the organisation of a permanent network for the monitoring of groundwater at a regional scale in Umbria. The presence of a permanent monitoring network is asked for by the recent Italian policy (DL 152/99) which receive the urban waste water treatment EU Directive (91/271/EEC) and the EU Directive concerning the pollution from nitrates (91/676/EEC);
– the definition of groundwater chemistry of each aquifer;
– the study of the spatial and temporal variations of the chemical composition of groundwater related to diffuse and local contamination processes;
– the individuation of the quantitative seasonal and long-trend variations caused by changes in the quantity of recharge (long term changes and/or seasonal variations in infiltration), variations in the recent pumping history and climatic changes.

The geological setting of the study area is the result of two main tectonic phases. A primary compressive phase produced, during Oligocene-Miocene, overthrusting of the Tuscan Nappe onto the Umbria-Marches Mesozoic carbonate-evaporite units, and the uplift of the Apennine chain (Miocene-Plio-Pleistocene). A subsequent tensional tectonic phase (Upper Miocene-Upper Pleistocene) overprinted the compressive structures and resulted in several NW-SE graben systems bounded by westward-dipping master faults and filled by Neogene sediments (Lavecchia, 1990).

From an hydrogeological point of view, the NW-SE graben systems are the seat of five large alluvial aquifers with thickness ranging from few meters to more than a hundred meters,
composed by sand, gravel and clay, whereas the Mesozoic carbonate-evaporite formations of the Apennine chain host some vast regional aquifers characterised by large infiltration areas and few springs, discharging most of the circulating water, located at the margins of the hydrogeological structures.

A general description of the hydrogeological setting of Umbria is reported in Boni et al. (1991) and Marchetti and Martinelli (1991).

METHODS

Definition of the monitoring network

Starting from a database of about 1400 points sampled during previous studies (Boni et al., 1991; Marchetti and Martinelli, 1991; Marchetti, 1995; Martinelli and Passeri, 1998; Ficiarà et al., 1998), we selected a monitoring network made of 250 wells and springs covering the five larger alluvial aquifers of the region and the regional carbonate and volcanic aquifers. For the carbonate and volcanic regional aquifers, where most of the circulating water is discharged by few, highly representative springs, the monitoring points have been selected on the basis of their discharge rate and considering their position with respect to the main geological structures. For the alluvial aquifers, the selection of the monitoring points has been done considering their statistical significance, the land use, the local geological heterogeneity and the well construction characteristics. We selected preferably domestic wells and public-supply wells rather than agricultural wells because they are not in use during the autumn and the winter. After the first campaign (February 1998) the monitoring network has been refined eliminating or substituting the wells with the worst constructive characteristics. The resulting monitoring network (Table 1) is composed by 220 points (206 wells and 14 springs) and is characterised by a variable sampling density, mainly depending on the geological heterogeneity and on the local land use. The location of the monitoring points is shown in Figure 1.

Table 1. Distribution of the monitoring points and sampling density

<table>
<thead>
<tr>
<th>ALLUVIAL AQUIFERS</th>
<th>area of the hydrogeological basin (km²)</th>
<th>monitoring wells</th>
<th>Density (number of points - km⁻²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Conca Ternana</td>
<td>60</td>
<td>30</td>
<td>0,50</td>
</tr>
<tr>
<td>Valle Umbra</td>
<td>330</td>
<td>86</td>
<td>0,26</td>
</tr>
<tr>
<td>Alta Valle del Tevere</td>
<td>110</td>
<td>29</td>
<td>0,26</td>
</tr>
<tr>
<td>Media Valle del Tevere</td>
<td>200</td>
<td>41</td>
<td>0,20</td>
</tr>
<tr>
<td>Conca Eugubina</td>
<td>50</td>
<td>20</td>
<td>0,40</td>
</tr>
<tr>
<td>TOTAL</td>
<td>750</td>
<td>206</td>
<td>0,27</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>SPRINGS</th>
<th>area of the hydrogeological basin (km²)</th>
<th>monitoring springs</th>
<th>Total discharge of the aquifers (m³ - s⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>East-Umbria carbonate aquifers</td>
<td>650</td>
<td>9</td>
<td>11.6</td>
</tr>
<tr>
<td>Valnerina carbonate aquifers</td>
<td>1025</td>
<td>4</td>
<td>23.3</td>
</tr>
<tr>
<td>Volsini volcanic aquifer</td>
<td>200</td>
<td>1</td>
<td>0.8</td>
</tr>
</tbody>
</table>

The monitoring strategy includes two different approaches:

- the punctual monitoring, consisting of periodical sampling and measurements of the hydrogeological and chemical-physical parameters in the selected points. In order to obtain correct conditions for field measurements, sampling and conservation of water samples, a mobile-laboratory for the preparation of samples for the analysis of pesticides, hydrocarbons, PAH and phenols at the home-laboratory. In order to correctly describe the seasonal and long-term variations we decided to use a sampling frequency of four campaigns per year.
a remote monitoring system, consisting of 12 automatic stations that continuously measure the flow rate, the specific electrical conductivity and the temperature of selected springs of the carbonate aquifers. In the next months the remote monitoring system will be extended also to the volcanic and the alluvial aquifers.

**Figure 1. The principal aquifers of Umbria and the PRISMAS-Umbria monitoring network**

**Water level measurements, sampling and analysis**

The depth to water level is measured using the electric-tape method both in static conditions and during pumping. The static and the maximum pumping water levels are determined by subtracting the distance between the measuring point (usually the wellhead) and the water table from the altitude of the measuring point (previously determined using a differential GPS system). Groundwater sampling is done as close to the wellhead as possible, before tanks, distribution systems, or treatment systems. Directly in the field are performed the measurements of chemical-physical parameters (pH, redox potential, dissolved oxygen, temperature and specific electrical conductivity) the determination of the alkalinity (measured by titration with 0.01 N HCl) and the preparation of the samples for the successive laboratory analysis.

For each monitoring point the preparation procedures include:
- filtration using a 0.45-µm membrane and acidification with HNO₃ of a 100 ml sample for the heavy metals laboratory analysis;
- pre-concentration of 2 litres of solution, using SPE columns, for the analysis of pesticides, hydrocarbons, PAH and phenols;
- preparation into 100 ml glass bottles of the samples for the total organic carbon determination and for the analysis of organic micropollutants;
- preparation into 250 ml polyethylene bottles of the samples for the determination of major cations and anions;
- storage of the samples at 4°C into the refrigerator of the mobile-laboratory.
The anions Cl\(^{-}\), SO\(_4\)\(^{2-}\), F\(^{-}\) and NO\(_3\)\(^{-}\) and the cations Na\(^{+}\), K\(^{+}\), Ca\(^{2+}\) and Mg\(^{2+}\) are analysed by ion chromatography, ammonium by photometry, heavy metals by atomic absorption spectrometry with a graphite furnace. The determinations of pesticides, hydrocarbons, PAH phenols and organic micropollutants are performed by gas-chromatography and gas-cromatography-mass spectrometry.

The field procedures are performed by two operators which collect approximately 10 samples per day; all the laboratory determinations are performed within 24 hours after sampling.

Data flow and processing

The hydrogeological and chemical data of the punctual monitoring network, together with the data of the remote monitoring system, flow into a relational database (DB-PRISMAS) that manages their acquisition and processing allowing the continuous update of the network. The flow chart of Figure 2 show the data processing conceptual scheme.

![Flow chart showing the data processing conceptual scheme](image)

DB-PRISMAS has a user-friendly interactive environment and covers a wide range of functions frequently used for the interpretation of hydrochemical and hydrogeological data going from simple unit transformation and charge balance to descriptive statistics. The stored data can be printed, exported in various file formats (txt, xls, dbf) and subdivided into subset using the appropriate queries. Furthermore, DB-PRISMAS is characterised by a direct extraction procedure for the geochemical program AQUACHEM\(^\text{©}\), for the geostatistical programs GEOEAS and SURFER\(^\text{©}\), and for numerous hydrogeological modelling programs. Finally, the files produced by DB-PRISMAS and SURFER\(^\text{©}\) can be linked to the PRISMAS geographic information system based of the ArcView\(^\text{©}\) GIS software, allowing the representation, the editing and the continuous update of the spatial data. An example of the graphical layouts produced by the PRISMAS geographic information system is shown in Figure 3.

Costs of the Project

The PRISMAS National Project has been funded by the Italian Ministry of Environment and by four Regions (Umbria, Piemonte, Liguria, Basilicata) for 2.250.000 €. The costs for the design and implementation of the PRISMAS Umbria monitoring network amount to about 750.000 € (33% of the total cost of the National Project). 13% of the costs are for the revision of the existing data and the selection of the monitoring network, 8% for the design and assembly of the mobile-laboratory (including the prototype concentrator), 36% for the remote automatic stations and 43% for the discrete monitoring activities (including the laboratory analysis).

RESULTS

Alluvial aquifers

Groundwaters from the alluvial aquifers show a variable composition, generally Ca(Mg)-HCO\(_3\), with a salinity ranging from 0.4 to 1.6 g/l and diffuse NO\(_3\) contamination processes. The average NO\(_3\) concentrations ranges from 20 mg/l in the Conca Ternana aquifer to more than 50
mg/l in the Valle Umbra aquifer. Some public supply wells have NO3 concentrations close to the European and Italian water pollution limit value (50 mg/l). The principal sources of groundwater contamination are irrigation practices, fertilizer application, subsurface percolation of septic tanks, animal feeding operation. Local contamination events with pesticides, hydrocarbons, phenols and heavy metals have been recorded in the last two years. In particular pesticides contamination concern 15% of the monitoring points of the Alta Valle del Tevere alluvial aquifer and about 5% of the monitoring points of the Valle Umbra and Conca Eugubina aquifers. A low level, diffuse contamination with chlorinated hydrocarbons has been detected in the Conca Ternana, Conca Eugubina and Valle Umbra aquifers during the June 2000 campaign.

Carbonate Aquifers

The data of the regional carbonate aquifers show that groundwater chemistry depends mainly on three processes: (1) calcite dissolution close to equilibrium conditions under an externally fixed CO2 pressure producing Ca-HCO3 waters; (2) dedolomitization driven by the irreversible CaSO4 dissolution and generating Ca(Mg)-SO4 groundwater compositions; (3) mixing of the Ca-HCO3 groundwater with the Ca(Mg)SO4 groundwater. The main observed chemical variations are clearly related to natural trends.

For example the comparison of the continuous and the discrete monitoring data of the Capovena spring show that the variations of the SO4 content are strictly correlated to the specific electrical conductivity recorded by the remote station, but inversely proportional to the spring flow (Q). This behaviour can be explained considering that the Capovena water is a mix of a shallow Ca-HCO3 component and a deeper Ca(Mg)SO4 component. The total variations of the spring flow are due essentially to the seasonal variations of the shallower Ca-HCO3 component, while the Ca(Mg)SO4 deep component is less influenced by seasonal trends. The decrease of the Ca-HCO3 component occurring during the dry season produce a relative increase of the Ca(Mg)SO4 component.

In all the surveyed springs NO3, ammonia, organic pollutants and heavy metals are practically absent: only Fe and Mn show detectable contents compatible with the natural background. The Apennine springs are a strategic virtually unpolluted resource, which protection should be a primary objective of the future water management policies.
CONCLUSIONS

During the 1997-2000 period the main problems affecting the studied aquifers have been the diffuse nitrate contamination, some pesticide contamination events and some point source contaminations with phenols, hydrocarbons and heavy metals. During the project period more than 2000 chemical analyses have been performed and two operators spent more than 250 days for the field work. The use of a mobile laboratory and the direct connection of all the components of the project (field operators, laboratories, GIS operators) to the same relational database, greatly improved our productivity allowing the achievement of our objectives. Besides the analytical and scientific problems, other important aspects of the PRISMAS experience are (i) the organization of all the components involved in the project (field operators, laboratories and computer operators), (ii) the development of software applications ensuring the correct flux of data from the field campaigns to the final reports and (iii) the estimation of the costs. The correct estimation of the costs and of the time necessary for the execution of the project is one of the main results of the PRISMAS Project. Finally, another key element of the PRISMAS National project is the diffusion of the results. It will be obtained in three different ways: 1) presentation of the main results to the scientific community during congresses and seminars; 2) publication of technical manuals concerning the technical, hydrogeological, social and economic aspect of the monitoring network design; 3) a WWW site at the address http://www.arpa.umbria.it/prismas.htm.

REFERENCES


MONITORING AND ASSESSMENT OF TRANSBOUNDARY GROUNDWATERS
State of the art on monitoring and assessment of groundwaters and the set up of a transboundary data information system on groundwater

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In this paper the link between groundwater monitoring and assessment and groundwater management issues will be dealt with. The definition of the management objectives and tasks results in a need for information which has to be provided by appropriate groundwater monitoring and assessment. Groundwater management issues are related to the actual and potential uses and functions of the groundwater resources, to the problems that might occur when certain threats becomes effective and to the measures that have to be implemented. To arrive at the specification of the data to be collected in the initial monitoring design stage successively the management objectives and tasks, the information needs and the technical objectives have to be worked out. The characterisation of the relevant aquifer systems and the actual state of the groundwater resources is a prerequisite for considering monitoring and assessment of groundwater in general and of transboundary groundwater bodies in particular. The step-wise approach of the general set up of the design and establishment of a groundwater monitoring and assessment system is presented. The main categories or types of monitoring are given.

Sustainable data management (collection, storage, exchange and presentation of information) is an essential part of the monitoring and assessment process in order to realise adequate and sustainable water management. This is illustrated by the Euregio information system for transboundary water management REGIS-Vechte, which is designed and developed for the cross-border basin of the Vechte River between Germany and The Netherlands.

PREFACE

This paper addresses the contents and outcomes of the report “State of the art in Monitoring and Assessment of Groundwaters”. (Uil et al, September 1999). The report is one of the four background documents used for the drafting of Guidelines on monitoring and assessment of transboundary groundwaters. The study was carried out for the ECE Task Force on Monitoring and Assessment under the Convention on the Protection and Use of Transboundary Watercourses and International Lakes.

PROBLEM DEFINITION

Because the borders between riparian states do not necessarily coincide with the natural boundaries of groundwater flow systems, groundwater may flow from one state to another. Moreover, groundwater management actions such as abstractions or other activities on one side of the border may adversely affect groundwater functions on the other side. To be able to distinguish natural characteristics from anthropogenic effects, information will be required about the aquifer and groundwater flow systems on both sides of the border. However, in practice, it is often difficult to obtain consistent pictures of the subsoil and groundwater characteristics. Based on the information originating from different states, the interpreted pictures often show abrupt and unrealistic changes in geology and groundwater characteristics. The information originating from different states, the interpreted pictures often show abrupt and unrealistic changes in geology and groundwater characteristics. Based on the information originating from different states, the interpreted pictures often show abrupt and unrealistic changes in geology and groundwater characteristics. Based on the information originating from different states, the interpreted pictures often show abrupt and unrealistic changes in geology and groundwater characteristics. Based on the information originating from different states, the interpreted pictures often show abrupt and unrealistic changes in geology and groundwater characteristics. Based on the information originating from different states, the interpreted pictures often show abrupt and unrealistic changes in geology and groundwater characteristics. Based on the information originating from different states, the interpreted pictures often show abrupt and unrealistic changes in geology and groundwater characteristics. 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Based on the information originating from different states, the interpreted pictures often show abrupt and unrealistic changes in geology and groundwater characteristics. Based on the information originating from different states, the interpreted pictures often show abrupt and unrealistic changes in geology and groundwater characteristics. Based on the information originating from different states, the interpreted pictures often show abrupt and unrealistic changes in geology and groundwater characteristics.

Until recently the interests in groundwater were mostly focused on exploitation of the available resources and on counteracting the possible spread of contaminants in those aquifers with moderate to high permeability, allowing reasonable flow rates and large discharges. However in
the last decades the possibility of having (nuclear) waste sites in low permeability zones induced also an interest in the control and monitoring of the groundwater quality in those areas. Therefore groundwater hydraulics in low permeability zones (including salt domes and hard rock areas) becomes more and more important, in particular when transboundary aspects are involved.

Without proper establishment of cross-border groundwater monitoring and assessment, errors may occur in the assessment of the groundwater resources and in the prediction and evaluation of the effects of management actions on the groundwater functions.

Present practices of transboundary monitoring and assessment in ECE countries

In October 1996 questionnaires were sent to the 27 ECE countries on the initiative of the UN/ECE Task Force on Monitoring & Assessment, with the aim of collecting information on the actual practices with regard to monitoring and assessment of transboundary groundwaters. Twenty two questionnaires containing different levels of detail were returned (Uil et al., Sept. 1999). Most of the ECE countries stated that information about groundwater is collected either in a systematic way or on an ad hoc basis. Based on the questionnaires returned, it is concluded that apparently in most responding countries some attention is paid to transboundary monitoring and assessment of groundwater. However, the impression is gained that purposely designed and operated transboundary monitoring and assessment of groundwater hardly exist.

The national monitoring networks are mostly operated and maintained with application of national standards and quality control procedures. Computerised data processing are applied in most responding ECE member states. Different data base systems are used. Data formats developed will be different which complicates the exchange of information.

DEFINITIONS

The two terms monitoring and assessment of groundwater are frequently confused and used synonymously. Monitoring is just one of the instruments used to obtain information for adequate assessment of groundwater quality and quantity.

- **Groundwater monitoring** is the collection of data, generally at set locations and depths and at regular intervals in order to provide information which may be used (i) to determine the state of groundwater both in quantitative and qualitative sense, and (ii) to provide the basis for detecting trends in space and time and (iii) to enable the establishment of cause-effect relationships.

- The process of **groundwater assessment** is an evaluation of the physical and chemical state of groundwater in relation to (i) natural situation, (ii) human intervention and (iii) actual and intended functions and uses. Particularly in relation to uses which may adversely affect human health and the environment.

Groundwater monitoring network

It is assumed that the technical basis of a network for groundwater monitoring generally consists of a network of observation points which are either existing wells, boreholes and springs or are purposely designed and installed observation wells or observation piezometers tapping the groundwater body which has to be monitored. A network provides quantitative information on the measured or analysed variables. Other possible monitoring techniques such as geophysical techniques and remote sensing which can provide qualitative information on groundwater have not been considered.

CHARACTERISATION OF THE GROUNDWATER SYSTEMS

A prerequisite for monitoring and assessment of groundwater resources in general and for transboundary groundwater bodies in particular is the preliminary characterisation of the relevant aquifer systems and the actual groundwater water flow systems. The first aim is to define the limits of all the groundwater flow systems which may have transboundary flow components. At the border between two states different flow systems might be superimposed and even opposite flow directions might occur (Figure 1).
Figure 1: Transboundary groundwater flow systems

Transboundary systems might have recharge areas on one side of the border and discharge areas on the other side. In such cases it will be important to know the transboundary subsurface flow in terms of quantity and quality. Activities within the recharge areas at one side of the border might adversely affect the groundwater quality on the other side of the border. Further transboundary water transmitting layers, zones or structures have to be delineated in order to be able to produce a consistent picture of the geometry of these layers, zones or structures (Figure 2.). This will be needed for a proper assessment of possible transboundary phenomena. Therefore an integrated interpretation of transboundary information on geology, geophysics and geohydrology will be necessary.

Figure 2: Effect of a transboundary aquitard on groundwater flow

Attention should be paid to the effect of the different groundwater transmitting rock types. Generally groundwater flow in porous media is much easier understood than the transport of groundwater in rock types with secondary porosity such as fractured and fissured consolidated rock formations or in limestone with karst features.
GROUNDWATER MANAGEMENT OBJECTIVES AND ISSUES

Generally, groundwater management is aimed at an environmentally and economically sound and sustainable development of the groundwater resources, which is socially accepted. The groundwater management objectives should be defined for the different management issues which are linked to the (i) functions and uses of the groundwater resources, the (ii) threats and problems with respect to groundwater quality and quantity and (iii) management measures. Then the information needs that should be provided by monitoring can be specified. The general groundwater management issues are comparable with the core elements in water management as presented in UN/ECE (1996). These elements in water management and their interactions are shown in the diagram of Figure 3.

Figure 3: Core elements in water management

The groundwater issues ascertained are as follows:
(-) The actual and potential uses and/or functions which are or can be assigned to the inherent quantity and quality features of the delineated groundwater system,
(-) The external threats (pressures) from pollution sources and other human activities,
(-) The actual and potential groundwater quantity and quality problems, which occur or may occur when a threat is or becomes effective,
(-) The inherent vulnerability of the groundwater system, and
(-) The identification of measures and their impact on the overall functioning of the groundwater system.

In Table 1 a summary is given of possible functions/uses, threats and problems.

Table 1: Possible functions/uses, threats and problems

<table>
<thead>
<tr>
<th>Functions/uses</th>
<th>Threats</th>
<th>Problems</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ecological function</td>
<td>- landuse (diffuse pollution: agriculture, (geo-) infrastructure, industry, urban areas), - airborne pollution - abstraction - point/line pollution sources - potential pollution sources</td>
<td>- desiccation, desertification - acidification - excessive nutrients loads - salinization - pollution (organic, heavy metals) - spreading of pollutants - public health - land subsidence - foundation problems - over-abstraction</td>
</tr>
<tr>
<td>Water supply</td>
<td>- drinking water - agriculture - industry</td>
<td></td>
</tr>
<tr>
<td>Storage</td>
<td>- waste - geothermal energy</td>
<td></td>
</tr>
<tr>
<td>Transport</td>
<td>- soil remediation - confinement of pollution</td>
<td></td>
</tr>
<tr>
<td>Miscellaneous</td>
<td>- prevention of landsubsidence - protection of foundations</td>
<td></td>
</tr>
</tbody>
</table>

Functions/uses

After the characterisation of the groundwater systems, one of the first issues to be addressed will be the assessment of the actual and possible future functions and uses of the groundwater resources. This will be based on the quantity and quality features of the groundwater.

Vulnerability

An aquifer system is considered vulnerable when problems will occur relatively fast after a threat becomes effective and no or only little time will be available to implement adequate counteracting measures. Therefore the assessment of the vulnerability of a transboundary
groundwater system provides important information for the groundwater manager to assess possible transboundary effects as consequence of certain groundwater uses and functions or other human activities. The vulnerability of the system depends mainly on the groundwater flow situation (confined or unconfined, recharge, direction, flux and discharge), the thickness of the unsaturated zone, the hydraulic and (bio)chemical soil properties (e.g. organic content, hydraulic properties and composition of the top layer) and geology.

**Threats**

A threat is an activity or situation which might cause groundwater quality or quantity problems. In particular, when the threat and the connected problem are separated by a national boundary a transboundary approach to groundwater management is required. Threats may include potential pollution sources such as agriculture areas, urban areas, industrial areas, infrastructure and also subsoil use and airborne pollution. Besides adverse quality effects, threats may cause undesired quantity effects due to abstractions, improved drainage, construction of dams etc. Hence, potential threats can generally be assessed from the actual and historical land and subsoil use.

**Problems**

Groundwater problems are defined as undesired situations with respect to human health and the environment, which are related to the groundwater uses and/or functions. Abstractions may result in increasing salinization, over abstraction may give rise to mining of the groundwater resources and falling groundwater levels may give rise to problems like desiccation, land subsidence and foundation problems. Intensive agricultural practices and emission from other pollution sources have resulted in problems like acidification and excess nutrients.

**Measures**

After identification of the functions, uses, problems and threats of the groundwater resources, management measures can be elaborated to protect and/or re-establish and/or guarantee the functions and uses of the groundwater. Target scenario’s which have to be obtained by the proposed measures will be defined. Monitoring will be needed to check the effectiveness of the measures. In some cases early warning systems will be needed to warn off the possible occurrence of undesired situations in order to undertake counteracting measures. The elaboration of management measures has not been included in the study.

**INFORMATION NEEDS AND TECHNICAL OBJECTIVES**

Technical objectives, which should be quantified descriptions of the information needs, have to be specified to determine the data to be collected. These data has to provide the information that is needed to fulfil the management objectives. The subsequent steps are depicted in Figure 4

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![Figure 4: The initial steps of designing a groundwater monitoring system](image-url)
Technical objectives can be divided into two groups. The first group consists of "characteristics". The information needed is the final product of data analysis, where the data are interpreted, aggregated or integrated to provide information on a higher level. For many purposes the information needed is in terms of general characteristics with respect to the status of the groundwater resources in both qualitative and quantitative senses, like spatial or annual averages, extremes, natural variability, etc. The second group consists of "continuous representations". Continuous representations are quantitative or intuitive interpolations or extrapolations describing the target variable as a function of space and time. Determination of the spatial and temporal trends is often one of the most important objectives of a monitoring system. They may indicate undesired or desired changes in the groundwater conditions.

For each of the groundwater management issues the information needs and technical objectives have to be defined. In Table 2 a summary is given of possible information needs, technical objectives and the data to be collected for the management task to define the possible functions and uses of groundwater.

### Table 2: Linkage between management objectives and technical objectives for groundwater monitoring

<table>
<thead>
<tr>
<th>Management objective, task</th>
<th>Information needs</th>
<th>Technical objectives</th>
<th>Data type (or specific technical objectives). Some examples with respect to groundwater monitoring</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 Characterisation of the groundwater flow system, threats and problems, needed for defining of potential functions and uses</td>
<td>1 Characterisation of groundwater flow system (three dimensional picture)</td>
<td>- surface water system</td>
<td>- groundwater levels</td>
</tr>
<tr>
<td></td>
<td>2 Aquifers, aquitards and aquicludes,</td>
<td>- groundwater level contours</td>
<td>- macro parameters</td>
</tr>
<tr>
<td></td>
<td>3 Identification of groundwater potential (groundwater availability)</td>
<td>- horizontal and vertical flow components</td>
<td>- annual / monthly abstractions</td>
</tr>
<tr>
<td></td>
<td>4 Vulnerability of groundwater</td>
<td>- groundwater quality distribution</td>
<td>- ground water monitoring</td>
</tr>
<tr>
<td></td>
<td>5 Threats on groundwater</td>
<td>- groundwater level abstractions</td>
<td>- - discharge/recharge</td>
</tr>
<tr>
<td></td>
<td>6 Problems (quantity)</td>
<td>- soil properties</td>
<td>- fertilisers, pesticides</td>
</tr>
<tr>
<td></td>
<td>7 Problems (quality)</td>
<td>- geology</td>
<td>- groundwater levels, trends</td>
</tr>
<tr>
<td></td>
<td>8 Reference values for natural situation or background situation</td>
<td>- soil properties, diffuse pollution</td>
<td>- groundwater balances</td>
</tr>
<tr>
<td></td>
<td>- acidification</td>
<td>- acidification</td>
<td>- specific parameters</td>
</tr>
<tr>
<td></td>
<td>- excess nutrients</td>
<td>- excess nutrients</td>
<td></td>
</tr>
<tr>
<td></td>
<td>- salinization</td>
<td>- salinization</td>
<td></td>
</tr>
<tr>
<td></td>
<td>- pollution (spreading)</td>
<td>- pollution (spreading)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>- local phenomena to be avoided</td>
<td>- local phenomena to be avoided</td>
<td></td>
</tr>
<tr>
<td></td>
<td>- natural situation</td>
<td>- natural situation</td>
<td></td>
</tr>
<tr>
<td></td>
<td>- diffuse anthropogenic effects</td>
<td>- diffuse anthropogenic effects</td>
<td></td>
</tr>
</tbody>
</table>
DESIGN OF GROUNDWATER MONITORING AND ASSESSMENT SYSTEMS

The different steps and aspects of the general design sequence are depicted in the scheme of Figure 5. The design procedure follows the different components of the "monitoring cycle", it begins with the definition of the information needs and ends with the communication to the water managers and the general public. The design of the network distribution and the measurement frequencies and the specification of the parameters to be measured is the basic step for designing a network.

![Diagram of monitoring network system design](image)

Figure 5: Design and implementation process of a monitoring network for groundwater
Monitoring network system design

The principal design components are (i) the selection of the type and density of the network stations, (ii) the parameters to be measured and (iii) the frequency of measurements. Factors that determine the main components in network design for groundwater quality are summarised by Chilton et al (1994), see Table 3. An appropriate design also considers the necessary activities, methods and/or conditions for the establishment of the network stations, the technical design of the measurement points and the materials used. Further the set up of the operation procedures for groundwater monitoring has to be worked out (measurement and sample campaigns, laboratory analysis and data handling). Appropriate storage, processing and presentation of the data have to be guaranteed by an easy accessible data base management and information system.

Table 3: Factors that determine the main components in network design (Chilton et al., 1994)

<table>
<thead>
<tr>
<th>Sampling point</th>
<th>Sampling frequency</th>
<th>Choice of parameters</th>
</tr>
</thead>
<tbody>
<tr>
<td>Type</td>
<td>Location/Density</td>
<td></td>
</tr>
<tr>
<td>Primary assessment objectives Hydrogeology (complexity)</td>
<td>Primary assessment Objectives Hydrogeology (complexity) Geology (aquifer distribution) Landuse Statistical considerations (Costs)</td>
<td>Primary assessment Objectives Hydrogeology (residence time)</td>
</tr>
<tr>
<td>(Costs)</td>
<td>(Costs)</td>
<td>(Costs)</td>
</tr>
</tbody>
</table>

Statistical versus hydrogeologic approach

A statistical approach can be applied if observations are already available. In case of the design of the network distribution the approach is based on a basic property of a monitoring network namely the existence of a correlation of variables among different observation points. The distribution of the observation points of the network should enable the determination of the target variable anywhere in the system by interpolation of the measurements at the observation points with sufficient accuracy. Consequently the derived value for the target variable is referred to as the estimated value. The selection of the density and also of the frequency should be made in such a way that the estimated value of the target variable is sufficiently accurate.

Loaiciga et al. (1992) identify the main approaches to groundwater monitoring network design as (1) the hydrogeologic and (2) the statistical approaches. If the estimation error is explicitly calculated in one form or another, the quantification will always involve some sort of statistical criterion, e.g. certain accuracy. The hydrogeologic approach is the basis of the procedure most commonly used in practice. In areas where relevant hydrologic data are limited or even absent – in so called scarce data areas – this approach may be the only possible technique. In the hydrogeologic approach, no explicit quantification of the uncertainty is given. Instead, the network design follows from a deterministic, hydrogeologic, area description based on expert judgement.

Target variables

The objectives for monitoring are preferably defined in terms of target variables. In some cases the target variable can be a property of the groundwater, e.g. concentration of nitrate, however, in most cases the target variable is some function of the measured properties. It is important that, in principle, target variables are quantifiable in terms of scalar characteristics (mean value, indicator), a surface in space, a trend in time, levels of excess etc.

Design-information-costs relationship

The basic idea of monitoring network design is that there should be a relation between the monitoring effort (cost) and the order of magnitude of the estimation error. Increasing the monitoring effort will decrease estimation error. After having selected the appropriate target variable to meet the management objective, the optimal monitoring effort should be determined.
on the basis of this relationship and the subsequent benefits for the groundwater management. However, in most cases, it is difficult or even impossible to define this relationship and often the "information content" is used as a substitute objective. The monitoring network is designed by balancing the information content and the monitoring effort. The fact that monitoring network design is based on the relation between a estimation error and the monitoring effort, does not necessarily mean that the estimation error is quantified in statistical terms. In some cases the error can also be taken into account intuitively by professional judgement.

Main types of monitoring

Various types of groundwater monitoring categories and networks are distinguished. In Table 4, a distinction is made between monitoring for strategic management and policy, monitoring for operational water management and surveillance purposes. Operational monitoring and surveillance will be linked to respectively functions/uses and problems/threats, the so-called core elements in water management. Other monitoring types and networks can often be linked to one of these three categories. Some general characteristics and objectives of the different types of monitoring have been summarized in this table.

Table 4: Main types of monitoring

<table>
<thead>
<tr>
<th>Types of groundwater monitoring</th>
<th>Characteristics</th>
<th>Information</th>
</tr>
</thead>
<tbody>
<tr>
<td>Strategic</td>
<td></td>
<td></td>
</tr>
<tr>
<td>State-of-the art</td>
<td>Transboundary context</td>
<td>Comparable types</td>
</tr>
<tr>
<td>Basic / reference for status assessment and compliance</td>
<td>background stations</td>
<td>beyond local anthropogenic influence.</td>
</tr>
<tr>
<td></td>
<td>reference system</td>
<td>- relation to (diffuse) anthropogenic or natural causes</td>
</tr>
<tr>
<td></td>
<td>statutory monitoring</td>
<td>- international directives and conventions</td>
</tr>
<tr>
<td></td>
<td></td>
<td>- natural situation</td>
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<tr>
<td></td>
<td></td>
<td>- trends (natural, diffuse pollution, hydraulic regime)</td>
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<tr>
<td></td>
<td></td>
<td>- baseline (to detect human impact)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>- background levels</td>
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<tr>
<td></td>
<td></td>
<td>- spatial distribution</td>
</tr>
<tr>
<td></td>
<td></td>
<td>- (early warning)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>- compliance</td>
</tr>
<tr>
<td>Operational</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Monitoring, linked to function/use monitoring for specific purposes compliance special protection areas remediation and restoration</td>
<td>- user related monitoring</td>
<td>- linked to uses and functions, regulations, laws, directives, acts etc.</td>
</tr>
<tr>
<td></td>
<td>- compliance monitoring</td>
<td>- protection of functions and uses models</td>
</tr>
<tr>
<td></td>
<td>- implementation monitoring</td>
<td>- implementation and effectiveness</td>
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<tr>
<td></td>
<td>- effectiveness monitoring</td>
<td>- validation</td>
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<td>- validation</td>
<td>- quality standards</td>
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<tr>
<td></td>
<td></td>
<td>- criteria, thresholds</td>
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<tr>
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<td>- health risk</td>
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<td></td>
<td>- environmental risk</td>
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<td></td>
<td></td>
<td>- validation</td>
</tr>
<tr>
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<td></td>
<td>- forecasting</td>
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<td></td>
<td></td>
<td>- effectiveness of measures</td>
</tr>
<tr>
<td>Surveillance</td>
<td></td>
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<tr>
<td>Early warning and surveillance emergency response</td>
<td>- early warning monitoring</td>
<td>- tresholds</td>
</tr>
<tr>
<td></td>
<td>- impact monitoring</td>
<td>- early warning</td>
</tr>
<tr>
<td></td>
<td></td>
<td>- risks</td>
</tr>
<tr>
<td></td>
<td></td>
<td>- effectiveness of measures</td>
</tr>
</tbody>
</table>

REGIS-VECHTE: EUREGIO Information system for Transboundary Water Management

REGIS-Vechte is the application of the Regional Geohydrological Information System REGIS for the cross-border basin of the Vechte River between the Netherlands and Germany (Uil, Sept. 2000). The work is carried out within the project "Digitaler Wasserweg Vechte" in the framework of the INTERREG II program of the EU. The EUREGIO and the responsible water management authorities from both Germany and the Netherlands are participating in the project. REGIS is an interactive and open geohydrological application system based on an Oracle Database and ArcView GIS with specific REGIS extensions for data management, manipulation, evaluation and presentation. In the Netherlands REGIS includes also standard geohydrological mapping. Water management authorities, water boards and water companies are using the system.

The set up of this information system illustrates the importance of an information system as part of the monitoring and assessment process. The transboundary context of the project complicates proper data management and makes the use of a suitable and easy accessible information system even more useful.
The project’s objective is to set up an information system with uniform and geographically consistent digital information on the water resources of the Vechte River cross-border basin. Within the framework of the project the information related to groundwater and subsurface available in the central databases of the Netherlands and the two federal states of Lower Saxony and North-Rhine Westphalia in Germany is collected, converted, processed and stored in the database of REGIS-Vechte. Moreover some water companies in Germany have provided groundwater information. Figure 6 illustrates the flow of data.

![Diagram of data flow](image)

**Figure 6 Illustration of the flow of data for the set up of REGIS-Vechte, Euregio information system for transboundary water management.**

The figure illustrates also the most important transboundary challenges of the project, namely integration of the data and information received from the different databases into the REGIS system. Besides administrative problems e.g. to get permission to use certain data the following major problems that are linked to transboundary aspects have been tackled:

- to enable the cross-border geohydrological characterisation and modelling of the subsurface the relevant data (e.g. bore hole descriptions) had to be processed and interpreted in a uniform and comparable manner. The three geological surveys involved agreed upon the geohydrological layer schematisation to be applied. Accordingly about five thousand bore holes have been interpreted and entered into the REGIS system.
- Standards, definitions, symbols and units used for geological and geohydrological terms are often different.
- All the geographical information (points, lines, polygons and grids) with respect to the geohydrological units and groundwater had to be stored and presented in the co-ordinate systems of both countries and elevations had to be given with respect to the reference levels of both countries. Therefore it was necessary to create two databases. For the German users an information system has been established with the Gauss-Krüger co-ordinate system and with NN (Normalnull) as reference level. For the Dutch users an information system with the Dutch RD co-ordinate system and NAP (Nieuw Amsterdams Peil) as reference level has been established. Hundreds of files have been converted from one system to the other.
- The databases and data models used by the data and information providers are different, consequently the data have been received in different formats. The data have been converted to the REGIS data format. For the data from central databases it is worthwhile to set up standard protocols for data conversion however, for the data received from all kind of local data sources like e.g. water companies this is impossible.
Different databases and data models are used. Loss of data occurs when data are converted from one database to the other.

Presentation of cross-border groundwater information

As example of presentation of cross-border groundwater information with the REGIS system a grid map of the phreatic surface has been created with the data from both countries (Figure 7). It illustrates the occurrence of cross-border groundwater flow and the presentation possibilities of a transboundary information system.

Figure 7 Cross-border groundwater information

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State of the art on monitoring and assessment of groundwaters


EFFECT OF THE RIVER NILE FLOW REGIME ON THE DRAINAGE WATER QUALITY OF THE NILE DELTA

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The general trend in water quality management is to gather and use information on water quality variables for purposes of planning and operation of water resources systems to meet the target functions of the system and different stakeholders. However, the growing concern for environmental quality has given rise to a new trend with respect to the impact of water quality variables on human and environmental health and life conditions. Thus, there is a need for better understanding of water quality processes both in time and space under natural and man induced conditions.

In 1998, Egypt faced a relatively high flood condition that led to the release of extra water from High Aswan Dam. The purpose of the release was to maintain the permissible water level in Lake Nasser, one of the world’s largest artificial lakes. The aim of this paper is to assess the effect of the River Nile flow regime as a natural condition on the drainage water quality of Nile Delta. Around 140 locations were monitored on a monthly basis through the water quality monitoring program developed and operated by the Drainage Research Institute of National Water Research Center. The assessment includes TDS (Total Dissolved Solid), TSS (Total Suspended Solid), BOD (Biochemical Oxygen Demand), and heavy metals.

The Eastern Nile Delta was selected to carry out this research as a demonstration region for the Nile Delta. The percentages of changes in the selected water quality variables for all monitored locations are calculated. Correlation analysis is performed to investigate if there is a significant correlation between the variation of the water quality variables and the drainage water discharge. As a general remark, the water quality variables improved for almost all of the monitored locations due to the effect of the extra floodwater. Number of locations shows a correlation between the water quality changes and discharges. A water quality index was calculated for the most strategic locations in the Nile Delta region to investigate if there is a difference before and after the high flood regime.

In summary, the results show that the drainage water quality in the Nile Delta region has significantly improved.

INTRODUCTION

The irrigated agriculture of Egypt is sustained through provision of adequate land drainage systems. They were developed over the past four decades to mitigate the hazards of water logging and salinity resulting from perennial irrigation of one of the most intensive irrigated agriculture of the world. The drainage water in Upper Egypt is pumped or flows by gravity back to the Nile. In the Delta, the network of open drains discharges their flows mainly into the Northern Lakes or the sea as shown in Figure 1.

The total length of these drains in the Nile Delta is over 16,000 km long. The drainage system in the Nile Delta serves 22 drain catchments with area served 4.7 million acres of the total 7.4 million acres of agriculture land in Egypt.

High Aswan Dam was built as a flood control dam on river Nile to store storm runoff and reduce flooding downstream. In 1998, Egypt received relatively high floodwater that led to extra water being released from High Aswan Dam in order to maintain the permissible water level in Lake Nasser. This situation of high flows from the high Aswan Dam results in extra water for the irrigation and drainage systems. Moreover, it is expected that changes in the drainage flow could impact on the drainage water quality in terms of refreshing the system storage and the water stored in coastal Lakes.
The overall objective of this paper is to assess the effect of the River Nile flow regime as a natural condition on the drainage water quantity and quality of Nile Delta. The analysis includes the water quality in terms of concentration and load.

DRAINAGE WATER QUALITY MONITORING

One of MWRI major goals is to develop an effective long-term drainage water quality monitoring program which would provide better knowledge of Egypt’s drainage water. The monitoring program would provide reliable information to decision and policy makers about the drainage water reuse potential.

A monitoring network of 90 measuring sites on the main drains in the Nile Delta and Fayoum was established in the early 1980’s providing flow, salinity and macro-ions data. The current monitoring network in the Nile Delta and Fayoum consists of 140 sites (Figure 2). The Drainage Research Institute (DRI) of the National Water Research Center (NWRC) monitors the delta and Fayoum regions on a monthly basis. Under the program samples are analysed for physical, biological, microbiological, macro and micro ions.

HYDROLOGICAL REGIME BEFORE AND AFTER HIGH ASWAN DAM

The natural hydrological regime of the Nile has two main features; a low flow period from December to July and a flood flow period, which begins in early August and lasts until mid-November. In order to introduce perennial irrigation for crop production, the High Aswan Dam (HAD) was constructed in 1964/1968 on the River Nile. The high Aswan Dam created a huge artificial lake (reservoir) with a total length of 500 km and total capacity of 162 BCM. The live storage capacity is 90.7 billion m³ (operational and management zone) with an operation range of 147m to 175m. The Nile’s average annual discharge amounts to 84 BCM (Abdel-Karim 1992). Since the completion of the High Aswan Dam, the flow in the River Nile is regulated by storage in Nasser Lake. The Ministry of Water Resources and Irrigation in Egypt does not allow the lake level to exceed 175 m on August 1st of any year (Shahin, 1985). This corresponds to Lake Storage of 121 BCM. The maximum design water level for the dam is 182 m, which corresponds to 162 BCM. The Toshka spillway can release water to the Toshka depression in the western desert of Egypt when the reservoir level exceeds 178m. The maximum storage capacity of the Toshka depression is evaluated as 120 BCM (Khadr, 1992). The flow released from HAD is entirely pre-scheduled to meet downstream water needs which include irrigation, municipalities, industries and adequate navigation depths. The average releases at the HAD ranged from 235 million m³/day in mid-July to 75 million m³/day in mid-January (the minimum for navigation level).
During the high flood period between August and November (maximum peak of water level) the agricultural water requirements are minimum. This is the transient period between summer and winter crops and is the period where land preparation for winter crops is taking place. During the last high flood in 1998, the Ministry of Water Resources and Irrigation released 305 million m³/day from the High Aswan reservoir.

ANALYSIS APPROACH

Study Area

The Eastern Nile Delta was selected to carry out this research as a demonstration region for the Nile Delta. The Eastern Delta, except for few catchments, drains into Lake Manzala, which in turn discharges into the Mediterranean Sea as shown in Figure 2. A considerable area is drained into Lake Manzala by gravity for two main drain systems, Bahr El Baqar and Bahr Hadus Drain system.

![Figure 2: Water quality monitoring locations in Eastern Delta](image)

Scope of Analysis

The water quality data from the monitoring program over two periods, August to December 1997 and August to December 1998, were used for the comparison analysis. The first period represents the normal flood while the second represents the high flood. The data analyses are carried out for selected sensitive and indicator variables. The selected variables are Biochemical Oxygen Demand (BOD), Total Suspended Solids (TSS), Copper (Cu), Zinc (Zn), Lead (Pb) and Total Dissolved Salts (TDS) as well as the drain discharges (DRI, 2000).
The percentage of change in concentrations of these variables for all monitored sites in the Eastern Nile Delta was calculated as follows: Percent of change = \((C_{98} - C_{97})/C_{97} \times 100\); where \(C_{98}\) and \(C_{97}\) are the average concentrations for any variables over the flood periods of 98 and 97 respectively.

Additional analysis were carried out for the outfalls of the drainage systems in the Eastern Nile Delta (EB10, EB09, EF01, EM01 and ES01) as follows:
- The percentage of change in pollutant concentrations (mg/l) and loads (t/day);
- Statistical F-Test (Two Sample) for Variances was performed to examine if there is a statistical difference between the two compared flood periods;
- Correlation analysis between the discharge and the water quality variables;
- Water quality index for both flood periods.

RESULTS AND DISCUSSIONS

Many factors influence the drainage water quality in the Nile Delta including, but not limited to:
- Development of new reclamation projects;
- Operation status of the new and existing wastewater treatment plants;
- Pollution loads from the industrial sector;
- Flood regime for the August to December period;

However, the main factor having the greatest effect on the drainage system is the flood regime for the period August to December period; the other factors mentioned above have more or less the same level of effect for the entire year and therefore can be removed from the analysis.

Changes in Water Quality

The percentage of water quality concentration change in the Eastern Nile Delta is presented in Table 1 and Figures 3 to 5. The negative values indicate decrease in the concentration levels and vice versa for the positive values. It is apparent that concentrations for most of the water quality variables at all sites were remarkably decreased. This is an indication of water quality improvement. The TSS is the most improved variables followed by metals. BOD shows the least improvement.

Table 1: Summary of change in concentrations in the Eastern Nile Delta (%)

<table>
<thead>
<tr>
<th></th>
<th>BOD</th>
<th>TSS</th>
<th>Cu</th>
<th>Zn</th>
<th>Pb</th>
<th>TDS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Minimum</td>
<td>-37.3</td>
<td>-87.9</td>
<td>-77.8</td>
<td>-64.3</td>
<td>-54.2</td>
<td>-46.3</td>
</tr>
<tr>
<td>Maximum</td>
<td>56.0</td>
<td>44.4</td>
<td>-22.0</td>
<td>-15.0</td>
<td>31.0</td>
<td>16.3</td>
</tr>
<tr>
<td>Average</td>
<td>-2.2</td>
<td>-52.1</td>
<td>-43.1</td>
<td>-36.1</td>
<td>-16.0</td>
<td>-15.1</td>
</tr>
</tbody>
</table>

Figure 3: The percentage of change in concentrations in Bahr El Baqar drain during two flood seasons (1997 and 1998)
The percentage of change in loading in the Eastern Nile Delta is presented in Table 2 and Figures 6 and 7. It is apparent that most of the water quality variables concentrations for all sites were remarkably decreased while the loads were increased. As presented in Table 2, the loads for most of the variables increased by an average of 30%. The increase in loads is mainly due to the increase in drainage water flow in addition to the minor leaching and washing processes that took place through the drainage system.

Table 2: Summary of change in loads in the Eastern Nile Delta during two flood seasons (1997 and 1998)

<table>
<thead>
<tr>
<th></th>
<th>BOD</th>
<th>TSS</th>
<th>Cu</th>
<th>Zn</th>
<th>Pb</th>
<th>TDS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Load (flood 97) ton/day</td>
<td>240</td>
<td>756</td>
<td>0.10</td>
<td>0.11</td>
<td>0.03</td>
<td>5786</td>
</tr>
<tr>
<td>Load (flood 98) ton/day</td>
<td>320</td>
<td>963</td>
<td>0.13</td>
<td>0.14</td>
<td>0.05</td>
<td>9665</td>
</tr>
<tr>
<td>Minimum %</td>
<td>14</td>
<td>-20</td>
<td>-28</td>
<td>-19</td>
<td>-8</td>
<td>51</td>
</tr>
<tr>
<td>Maximum %</td>
<td>220</td>
<td>149</td>
<td>96</td>
<td>206</td>
<td>153</td>
<td>135</td>
</tr>
<tr>
<td>Average %</td>
<td>35</td>
<td>27</td>
<td>28</td>
<td>34</td>
<td>48</td>
<td>57</td>
</tr>
</tbody>
</table>
The F-test (two sample variance) was carried out (confidence level * = 90%) to examine the reliability of the changes in water quality for concentrations and loads at the outfall sites. The results show that the change in concentrations of TSS, Cu and Zn, are significant. Similarly, the change in loads of BOD, TSS, Zn, Pb and TDS were also significant. A representative sample of the F-test results is presented in Table 3.

Table 3: F-Test (Two-Sample) for Variances of loads at outfall sites
(Confidence level $\alpha = 90\%$)

<table>
<thead>
<tr>
<th></th>
<th>BOD - 97</th>
<th>BOD - 98</th>
<th>TSS - 97</th>
<th>TSS - 98</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean</td>
<td>47.96958</td>
<td>103.8513</td>
<td>151.2659</td>
<td>192.6755</td>
</tr>
<tr>
<td>Variance</td>
<td>629.6022</td>
<td>4802.982</td>
<td>1324.772</td>
<td>29237.28</td>
</tr>
<tr>
<td>Observations</td>
<td>5</td>
<td>5</td>
<td>5</td>
<td>5</td>
</tr>
<tr>
<td>df</td>
<td>4</td>
<td>4</td>
<td>4</td>
<td>4</td>
</tr>
<tr>
<td>F</td>
<td>0.131086</td>
<td></td>
<td>0.045311</td>
<td></td>
</tr>
<tr>
<td>P(F&lt;=f) one-tail</td>
<td>0.037181</td>
<td></td>
<td>0.005474</td>
<td></td>
</tr>
<tr>
<td>F Critical one-tail</td>
<td>0.156538</td>
<td></td>
<td>0.156538</td>
<td></td>
</tr>
<tr>
<td>There is difference</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
**Correlation between Flow and Quality Variables**

Correlation analysis was carried out between the water quality variables covering both flood periods. The calculated correlation coefficients were low indicating a weak correlation between the quality variables. However, the correlation between the flow and the water quality variables showed some correlation for specific variables. A summary of some correlation coefficient for outfall sites is given in Table 4. More data are required to carry out the correlation analysis between the different variables.

**Table 4: The correlation coefficients between the quality variables and the flow**

<table>
<thead>
<tr>
<th>Variable</th>
<th>EF01</th>
<th>EM01</th>
<th>ES021</th>
<th>EH17</th>
<th>EB11</th>
</tr>
</thead>
<tbody>
<tr>
<td>BOD</td>
<td>0.45</td>
<td>0.50</td>
<td>0.60</td>
<td>0.19</td>
<td>-0.07</td>
</tr>
<tr>
<td>TSS</td>
<td>-0.08</td>
<td>-0.69</td>
<td>-0.47</td>
<td>0.05</td>
<td>-0.05</td>
</tr>
<tr>
<td>Cu</td>
<td>-0.08</td>
<td>-0.55</td>
<td>-0.60</td>
<td>0.42</td>
<td>-0.37</td>
</tr>
<tr>
<td>Zn</td>
<td>0.29</td>
<td>-0.12</td>
<td>0.05</td>
<td>-0.15</td>
<td>-0.20</td>
</tr>
<tr>
<td>Pb</td>
<td>0.25</td>
<td>0.19</td>
<td>-0.12</td>
<td>-0.27</td>
<td>0.15</td>
</tr>
<tr>
<td>TDS</td>
<td>-0.79</td>
<td>-0.64</td>
<td>-0.71</td>
<td>0.40</td>
<td>0.08</td>
</tr>
</tbody>
</table>

**Water Quality Index**

Due to the large number of water quality variables to be monitored, a presentation of summary information related to these variables is difficult. One approach, which can be used, is an empirical water quality index (WQI), which combines the output data from several variables into one common numeric indicator of water quality. This numeric indicator represents an attempt to consider the relative water quality improvement due to high flood regime. One of the additional elements related to the index is the assignment of relative importance weights to the nine selected variables, which is presented in Table 5 (Ott, 1987).

**Table 5: The selected variables and relative weights for WQI (Ott, 1987)**

<table>
<thead>
<tr>
<th>Variable</th>
<th>DO</th>
<th>FC</th>
<th>pH</th>
<th>BOD</th>
<th>NO3</th>
<th>P</th>
<th>Temp</th>
<th>TSS</th>
<th>TDS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Weight</td>
<td>0.17</td>
<td>0.15</td>
<td>0.12</td>
<td>0.10</td>
<td>0.10</td>
<td>0.10</td>
<td>0.10</td>
<td>0.08</td>
<td>0.08</td>
</tr>
</tbody>
</table>

Two formulations have been considered for the WQI, one represents an arithmetic weighted approach (WQIa) and the other a geometric weighted approach (WQIg). The arithmetic method is calculated by the following formulations:

\[
WQI_a = \frac{\sum_{i=1}^{n} I_i W_i}{n}
\]

where: \( I_i \) = the functional rating relationship of the variable;
\( W_i \) = the final weight of the variable.
The value of WQI is used to classify the water quality according to the numerical range as presented in Table 6.

Table 6: Stream classification based on WQI (Ott, 1987)

<table>
<thead>
<tr>
<th>Numerical Range</th>
<th>Classification</th>
</tr>
</thead>
<tbody>
<tr>
<td>0 – 25</td>
<td>very bad</td>
</tr>
<tr>
<td>26 – 50</td>
<td>bad</td>
</tr>
<tr>
<td>51 – 70</td>
<td>medium</td>
</tr>
<tr>
<td>71 – 90</td>
<td>good</td>
</tr>
<tr>
<td>91 – 100</td>
<td>excellent</td>
</tr>
</tbody>
</table>

This procedure was followed to classify the drainage water quality for both flood periods as presented in Table 7.

Table 7: WQI for the outfall sites in the Eastern Nile Delta

<table>
<thead>
<tr>
<th>Discharge</th>
<th>EF01</th>
<th>EM01</th>
<th>ES021</th>
<th>EH17</th>
<th>EB11</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flood-97</td>
<td>29.5</td>
<td>18.4</td>
<td>22.9</td>
<td>27.9</td>
<td>10.1</td>
</tr>
<tr>
<td>Flood-98</td>
<td>33.5</td>
<td>22.7</td>
<td>28.6</td>
<td>31.4</td>
<td>11.2</td>
</tr>
</tbody>
</table>

The WQI calculations show that the drainage water quality can be classified as very bad and bad for all outfall sites. However, minor improvements are noticed for the high flood of year 1998.

CONCLUSIONS AND RECOMMENDATIONS

The major conclusions from this study can be summarised as follows:

- The concentrations for all sites decreased remarkably indicating water quality improvement. The TSS was the most improved variable followed by metals variable while BOD was the least improved one.
- The concentrations of TSS, Cu and Zn, and the loads of BOD, TSS, Zn, Pb and TDS change significantly with the flood regime.
- It is apparent that most of the water quality variable concentrations for outfall sites were remarkably decreased while the loads were increased by an average of 30%. The increase in loads is mainly due to the increase in drainage water flow in addition to the leaching and washing processes that took place through the drainage system.
- The results of the F-test (two sample variance) show that the change in concentrations of TSS, Cu and Zn, are significant. Similarly, the change in loads of BOD, TSS, Zn, Pb and TDS were also significant.
- The correlation between the quality variables is very weak while the correlation between the flow and the water quality variables showed that there was a fair degree of correlation for some of variables for limited number of sites.
- The WQI calculations show that the drainage water quality can be classified as very bad or bad for all outfall sites. However, minor improvement is noticed for high flood in 1998.

It is recommended that further analysis be conducted for the coming flood period. Also, further study is recommended to assess the effect of the increased loads during the flood period on Lake Manzala quality.

REFERENCES


RESULTS FROM THE FIRST THREE YEARS OF THE REVISED NASQAN PROGRAM: Lessons for Monitoring Fluxes on a National Scale

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In 1996 the U.S. Geological Survey (USGS) converted its National Stream Quality Accounting Network Program (NASQAN) into a national flux-based water quality monitoring program in the Mississippi, Columbia, Colorado, and Rio Grande Basins. Suspended sediment flux estimates for temporal spans of a year generally are accurate to within ±15% or better. Discharge and suspended sediment varied to a much greater extent than trace element concentrations. Most trace element levels are not elevated except Zn and Hg in the Ohio River as well as several others in the upper Columbia River which is impacted by mining waste. The majority (>70%) of the Cu, Zn, Cr, Ni, Ba, P, As, Fe, Mn, and Al are transported in association with suspended sediment; Sr transport seems dominated by the dissolved phase, whereas the transport of Li and TOC seem to be divided equally between both phases.

INTRODUCTION

The United States Geological Survey’s (USGS) National Stream Quality Accounting Network Program (NASQAN) began in 1973 with 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 water quality assessments. The program operated more-or-less continuously under these guidelines through 1995 [Ficke and Hawkinson, 1975; Office of Water Quality (OWQ), 1996]. In 1994, in response to substantially diminishing resources, changes in data requirements, and to better integrate the program with other USGS ambient water-quality monitoring programs, NASQAN was redesigned (OWQ, 1996). Initially, the new program, which began in late 1995 (the 1996 water year), had two objectives: 1) to characterize large U.S. river basins by measuring the flux of selected constituents at critical nodes in various systems; and 2) to determine the fluxes of a variety of chemical constituents to the coastal zone. The selected basins were the Mississippi, Columbia, Colorado, and Rio Grande. The determination of coastal zone fluxes was dropped after the first year because of insufficient spatial and temporal coverage due to funding limitations. Based on historical data, the revised NASQAN network should account for >80% of the annual movement of water within the conterminous U.S. Further, the basins represent four of the seven largest sources of suspended sediment in the conterminous U.S., and should account for the discharge of more than 260 Mt to the coastal zone (Meade and Parker, 1985).

DESIGN CONSIDERATIONS/CAVEATS

Each of the four NASQAN basin sampling designs were established with a view to understanding and evaluating the movement of water and a variety of water-quality-associated parameters. In general, and particularly in large rivers where point sources (e.g., an industrial outfall) usually have only relatively local impacts, water and dissolved constituents tend to behave conservatively; therefore, large river segments tend to be fairly compositionally homogeneous. Thus, contributions from sizable portions of a basin, and/or potential sources or sinks for a variety of chemical parameters, can be estimated by adding/subtracting the contributions/losses of large river segments. Further, hydrophilic substances tend to move through large basins along with the water itself. Hence, ongoing sampling of the same segment of water (i.e., Lagrangian sampling) as it moves through the system will permit the delineation of concentration changes and potentially, the physical/chemical processes acting on that substance to engender those compositional changes. Finally, the travel time for the impacted segment can be readily determined/predicted if the discharge or water velocity is known. Contrariwise, suspended sediment does not behave conservatively; it moves in and out of suspension, and there are constant exchanges between the water column, the river bed, and the river banks. Thus, the particles making up a ‘packet’ of sediment, and their associated chemical constituents, continuously change as material moves in (deposition) and out (resuspension) of ‘storage’ while the packet is transported downstream. As such, the packet rarely retains its
original particle composition, even over relatively short distances, and travel times are difficult to predict. Thus, suspended sediment and sediment-associated chemical constituents display much more marked spatial and temporal variability than dissolved constituents (e.g., Horowitz, 1995). Hence, continued sampling of the same packet of water as it moves downstream will not readily permit the delineation of ongoing particle-specific concentration changes, nor potential physical/chemical processes acting on that sediment-associated substance. As a result of this non-conservative behavior, contributions from sizable portions of a basin, and/or potential sources or sinks for a variety of sediment-associated chemical parameters, cannot be reasonably estimated by adding/subtracting the contributions/losses of large river segments.

As a result of the behavioral differences between water and suspended sediment, a sampling program designed to monitor dissolved fluxes may not be applicable to monitoring sediment and sediment-associated chemical fluxes. This can create network design problems, particularly when funding is limited. As an example, examine the sampling scheme for the middle/lower Mississippi River [from Thebes to St. Francisville, including the location of the Old River Control Structure, as well as the location of the Melville sampling site on the Atchafalaya River (Figure 1)]. Note that there are no main-stem sampling sites below the confluence of the Ohio and the Mississippi Rivers before St. Francisville, a distance of over 1100 km. Between the Ohio and Mississippi confluence and St. Francisville, the Arkansas River joins the Mississippi River, and about 25% of the Mississippi River is diverted into the Atchafalaya River at the Old River Control Structure (Figure 1). Also, the Atchafalaya sampling site (Melville) is downstream from its confluence with the Red River. As such, the effects of suspended sediment and the sediment-associated chemical contributions from the Ohio and the Arkansas Rivers on the mainstem Mississippi River cannot be readily delineated. Further, due to the location of the Melville sampling site, and the lack of a site at or near the Control Structure, there is no way to separate out the suspended sediment and suspended sediment-associated trace element contributions to the Atchafalaya between the Mississippi and Red Rivers. Water chemists resolve this by assuming that dissolved constituents behave conservatively; thus, the dissolved concentrations determined at St. Francisville are assumed to be equivalent to those in water diverted by the Control Structure. However, sediment chemists cannot make this assumption. In fact, substantial sediment resuspension has been observed between the Control Structure and the St. Francisville sampling site (R.H. Meade, USGS, oral comm., 1999). Thus, a flux monitoring program may require two separate sampling schemes, one for water and one for suspended sediment.

ESTIMATING SUSPENDED SEDIMENT FLUXES

Although each NASQAN site was instrumented to determine mean daily discharge, actual water and suspended sediment samples were collected no more than 12 to 15 times per year (OWQ, 1996). Sampling schedules were established to try to cover >80% of the typical range of annual flows; however, sampling tended to be biased in favor of non-base-flow periods. As a result of the limited physical sampling program, and the perception that at least a mean daily suspended
sediment concentration would be required to produce precise and accurate flux calculations (Walling and Webb, 1981; 1988; de Vries and Klavers, 1994; Horowitz, 1995; Roberts, 1997; Phillips et al., 1999), each site either had to be instrumented with some type of automated sampling/measuring device (Horowitz, 1995), or a site-specific discharge-based regression equation had to be developed for predicting suspended sedi-ment concentration (de Vries and Klavers, 1994; Phillips et al., 1999). No NASQAN sites were in-strumented with either automatic sampling equipment nor measuring devices; hence, the only means of estimating mean-daily suspended sediment concentrations was by developing a site-specific regression equation/model relating suspended sediment concentration to discharge.

Table 1. Summary of the results and associated errors for the suspended sediment predictive

<table>
<thead>
<tr>
<th>Basin</th>
<th>Site</th>
<th>Flux (Mt)</th>
<th>Pred. Flux (Mt)</th>
<th>Avg. Flux (Mt)</th>
<th>Method¹</th>
<th>n²</th>
<th>D³ (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mississippi</td>
<td>Ohio at Greenup</td>
<td>5.9</td>
<td>5.7</td>
<td>3.0</td>
<td>RM w/c</td>
<td>109</td>
<td>-3</td>
</tr>
<tr>
<td></td>
<td>Ohio at Cannellon</td>
<td>12.1</td>
<td>13.3</td>
<td>6.6</td>
<td>RM w/c</td>
<td>116</td>
<td>+10</td>
</tr>
<tr>
<td></td>
<td>Wabash at New Harmony</td>
<td>1.7</td>
<td>1.7</td>
<td>1.4</td>
<td>RM w/c</td>
<td>100</td>
<td>&lt;1</td>
</tr>
<tr>
<td></td>
<td>Tennessee at Paducah</td>
<td>0.4</td>
<td>0.4</td>
<td>0.4</td>
<td>RM w/c</td>
<td>134</td>
<td>-2</td>
</tr>
<tr>
<td></td>
<td>Ohio near Grand Chain</td>
<td>25.0</td>
<td>23.7</td>
<td>17.1</td>
<td>RM</td>
<td>258</td>
<td>-5</td>
</tr>
<tr>
<td></td>
<td>Mississippi at Clinton</td>
<td>2.0</td>
<td>2.0</td>
<td>1.6</td>
<td>RM w/c</td>
<td>163</td>
<td>&lt;1</td>
</tr>
<tr>
<td></td>
<td>Missouri River near Cubertson</td>
<td>1.8</td>
<td>2.0</td>
<td>2.0</td>
<td>RM w/c</td>
<td>127</td>
<td>+11</td>
</tr>
<tr>
<td></td>
<td>Yellowstone near Sydney</td>
<td>16.3</td>
<td>12.6</td>
<td>8.6</td>
<td>RM</td>
<td>235</td>
<td>-23</td>
</tr>
<tr>
<td></td>
<td>Missouri at Omaha</td>
<td>13.3</td>
<td>12.3</td>
<td>A</td>
<td>A</td>
<td>184</td>
<td>-8</td>
</tr>
<tr>
<td></td>
<td>Missouri at Hermann</td>
<td>61.2</td>
<td>63.5</td>
<td>43.3</td>
<td>RM</td>
<td>253</td>
<td>+3</td>
</tr>
<tr>
<td></td>
<td>Mississippi at Thebes</td>
<td>347</td>
<td>344</td>
<td>270</td>
<td>RM</td>
<td>1,418</td>
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</tr>
<tr>
<td></td>
<td>Mississippi at St. Francisville</td>
<td>70.9</td>
<td>71.0</td>
<td>66.4</td>
<td>RM</td>
<td>210</td>
<td>&lt;1</td>
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<tr>
<td></td>
<td>Atchafalaya at Melville</td>
<td>26.7</td>
<td>27.0</td>
<td>23.3</td>
<td>RM</td>
<td>152</td>
<td>+1</td>
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Columbia

<table>
<thead>
<tr>
<th>Basin</th>
<th>Site</th>
<th>Flux (Mt)</th>
<th>Pred. Flux (Mt)</th>
<th>Avg. Flux (Mt)</th>
<th>Method¹</th>
<th>n²</th>
<th>D³ (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Columbia</td>
<td>Columbia at Northport</td>
<td>0.3</td>
<td>0.2</td>
<td>A</td>
<td>A</td>
<td>162</td>
<td>-12</td>
</tr>
<tr>
<td>Columbia</td>
<td>Columbia at Vernita</td>
<td>0.2</td>
<td>0.2</td>
<td>A</td>
<td>A</td>
<td>116</td>
<td>-6</td>
</tr>
<tr>
<td></td>
<td>Snake at Burbank</td>
<td>0.7</td>
<td>0.6</td>
<td>0.4</td>
<td>RM</td>
<td>157</td>
<td>-13</td>
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<tr>
<td></td>
<td>Columbia at Warrendale</td>
<td>1.4</td>
<td>1.4</td>
<td>1.2</td>
<td>RM w/c</td>
<td>156</td>
<td>&lt;1</td>
</tr>
<tr>
<td></td>
<td>Willamette at Portland</td>
<td>0.8</td>
<td>0.8</td>
<td>0.4</td>
<td>RM w/c</td>
<td>205</td>
<td>-2</td>
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<tr>
<td></td>
<td>Columbia near Beaver Army Termin.</td>
<td>2.7</td>
<td>2.7</td>
<td>1.8</td>
<td>RM</td>
<td>68</td>
<td>&lt;1</td>
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Colorado

<table>
<thead>
<tr>
<th>Basin</th>
<th>Site</th>
<th>Flux (Mt)</th>
<th>Pred. Flux (Mt)</th>
<th>Avg. Flux (Mt)</th>
<th>Method¹</th>
<th>n²</th>
<th>D³ (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Colorado</td>
<td>Colorado near Cisco</td>
<td>5.5</td>
<td>4.4</td>
<td>3.6</td>
<td>RM</td>
<td>213</td>
<td>-20</td>
</tr>
<tr>
<td></td>
<td>Green River at Green River</td>
<td>6.4</td>
<td>4.9</td>
<td>4.6</td>
<td>RM</td>
<td>225</td>
<td>-23</td>
</tr>
<tr>
<td></td>
<td>San Juan near Bluff</td>
<td>6.8</td>
<td>5.8</td>
<td>A</td>
<td>207</td>
<td>-13</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Colorado above Diamond Creek</td>
<td>0.5</td>
<td>0.4</td>
<td>A</td>
<td>16</td>
<td>-9</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Colorado at NIB</td>
<td>1.0</td>
<td>1.2</td>
<td>0.5</td>
<td>RM w/c</td>
<td>122</td>
<td>+20</td>
</tr>
</tbody>
</table>

Rio Grande

<table>
<thead>
<tr>
<th>Basin</th>
<th>Site</th>
<th>Flux (Mt)</th>
<th>Pred. Flux (Mt)</th>
<th>Avg. Flux (Mt)</th>
<th>Method¹</th>
<th>n²</th>
<th>D³ (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rio Grande</td>
<td>Rio Grande at El Paso</td>
<td>0.2</td>
<td>0.2</td>
<td>0.1</td>
<td>RM w/c</td>
<td>135</td>
<td>-11</td>
</tr>
<tr>
<td></td>
<td>Rio Grande at Foster Ranch</td>
<td>1.2</td>
<td>1.0</td>
<td>0.3</td>
<td>RM</td>
<td>151</td>
<td>-20</td>
</tr>
<tr>
<td></td>
<td>Rio Grande at Laredo</td>
<td>0.3</td>
<td>0.3</td>
<td>0.1</td>
<td>RM</td>
<td>95</td>
<td>+3</td>
</tr>
<tr>
<td></td>
<td>Arroyo Colorado at Harlingen</td>
<td>0.02</td>
<td>0.01</td>
<td>A</td>
<td>83</td>
<td>-7</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Rio Grande at Brownsville</td>
<td>0.2</td>
<td>0.2</td>
<td>0.1</td>
<td>RM w/c</td>
<td>136</td>
<td>-12</td>
</tr>
</tbody>
</table>

Method¹: RM = regression model, RM w/c = regression model with 'smearing' correction, A = average values
n² = number of samples in the calibration set and used to estimate the error.
D³ = percent deviation from the calibration set used to develop the site-specific models.

Although there are various approaches (extrapolation or interpolation plus various correction factors) for estimating suspended sediment concentration from discharge data, the most common are based on extrapolation from a log-log regression model (a rating or transport curve) relating the two (e.g., Walling and Webb, 1981; de Vries and Klavers, 1994; Phillips et al., 1999). The efficacy of this procedure depends on the number of paired data points available to develop the rating curve, and how well they represent the range of discharges and suspended sediment concentrations at a site (Walling and Webb, 1981; Roberts, 1997). With the exception of the Mississippi River at Thebes, which is a daily sediment station, the number of available data points from the 3-year NASQAN flux program is not large, and ranges from a low of 10 to a maximum of 43. Fortunately, all but one of the sites in the revised program also were former NASQAN sites, hence, a substantial amount of historic data is available.

The procedure for developing estimates of a single daily suspended sediment concentration was the same for all 30 NASQAN sites. All current and available historic suspended sediment and
discharge data for each site were combined into a single calibration set, and log transformed. A series of regression equations (e.g., linear, polynomial) were then calculated; the one with the highest $R^2$ was selected, and a residual analysis performed. If the pattern for the residual analysis was random, then predicted daily suspended sediment concentrations were generated. In turn, these were used to calculate instantaneous daily fluxes for each set of data points; then the individual, as well as the sum of all the predicted and actual fluxes were compared for accuracy and bias. As can occur with these types of calculations, the conversion from log space to arithmetic space may produce a negative bias; when compensation was necessary to reduce/eliminate the bias, it was done using a 'smearing correction' (Duan, 1983). When no significant regression (based on the value of $R^2$) could be calculated for a site-specific calibration set, mean suspended sediment concentration was used to estimate the instantaneous daily fluxes (Table 1). The percent deviations ($\%$ D's) represent the difference between the actual as opposed to the calculated (based on the model-derived daily suspended sediment concentrations) sum-med instantaneous daily fluxes for the individual calibration sets for each site (Table 1). Hence, the $\%$ D's provide an overall indication of error, whereas the sign provides an indication of bias. Errors using this approach usually are less than $\pm 10\%$ for the 3-year reporting period; however, these errors increase substantially for temporal spans shorter than 1-year.

As a daily sediment station, the Mississippi River at Thebes provides an ideal case for examining both the nature of the errors associated with the use of the rating-curve method of estimating suspended sediment concentrations (fluxes), as well as the relative errors associated with different levels of temporal resolution. Examination of the actual daily versus the calculated daily fluxes for the site indicates that the rating curve method tends to underpredict the highs and overpredict the lows; thus, the method fails to encompass the complete range of variance in the daily fluxes (suspended sediment concentrations) at the site. As such, the range of errors associated with shorter timeframes are likely to be sub-stan-tially larger than those associated with longer timeframes (Table 2). Hence, for the Mississippi River at Thebes, where 3 years of daily suspended sediment concentration and discharge data were used to develop the predictive model, the $\%$ D’s range from -74 to +146 for daily values, -48 to +68% for monthly values, -27 to +24% for quarterly values, -14 to +12% for annual values, and <2% for three or more years (Table 2). The predicted annual suspended sediment flux for the 1998 water year (about 94 Mt) was generated without any 1998 suspended sediment data in the calibration set and had a -6% D (Horowitz, et al., 2000, in press).

Table 2. The range of errors (%) for different levels of temporal resolution for estimated suspended sediment fluxes for the Mississippi River at Thebes, Illinois.

<table>
<thead>
<tr>
<th>Temporal Resolution</th>
<th>Minimum</th>
<th>Maximum</th>
<th>Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>Daily</td>
<td>-74</td>
<td>146</td>
<td>30</td>
</tr>
<tr>
<td>Monthly</td>
<td>-48</td>
<td>68</td>
<td>18</td>
</tr>
<tr>
<td>Quarterly</td>
<td>-27</td>
<td>24</td>
<td>12</td>
</tr>
<tr>
<td>Yearly</td>
<td>-14</td>
<td>12</td>
<td>8</td>
</tr>
</tbody>
</table>

The patterns of prediction for rating curve-derived single daily suspended sediment concentrations (fluxes) displayed by the model for the Mississippi River at Thebes are mirrored by all the other models for the various NASQAN sampling sites. That is, all the models tend to underpredict the high and overpredict the low single daily suspended sediment concentrations at their respective sites. Also, note that the $\%$ D’s for sand-dominated rivers (e.g., Yellowstone River at Sydney, Colorado River near Cisco, Green River at Green River) tend to be substantially worse than those where the suspended sediment is dominated by silt- and clay-sized particles (Table 1). Thus, it would seem that the rating-curve method for predicting suspended sediment concentration is less effective for large sand-dominated rivers. However, because all the models for the other revised NASQAN program sites are based on substantially smaller and less complete calibration sets, the $\%$ D’s associated with the estimations of total fluxes for the 3-year period tend to be larger than those cited for the Thebes site (Table 1). Further, the errors associated with temporal periods shorter than 3 years will be greater than those determined for the Thebes site. Obviously, based on the foregoing, it is possible to produce ‘reasonable’ estimates of annual suspended sediment fluxes using the rating curve method, without actually obtaining daily suspended sediment concentrations. However, it should be noted that the
relative success of this approach, especially considering the short time span/limited number of samples occurred because there was a substantial amount of historic data available to enhance the calibration sets. Finally, end-users must determine the maximum acceptable error limits for different levels of temporal resolution. In the case of the revised NASQAN program, an acceptable limit seems to be quarterly flux estimates; more accurate, shorter-term estimates would require a substantial increase in sampling frequency or, in the absence of actual measurements, a different approach for estimating mean daily sediment concentrations.

SEDIMENT-ASSOCIATED TRACE ELEMENT CONCENTRATIONS/FLUXES

The traditional, as well as the regulatory method of determining suspended sediment-associated trace element concentrations entails the collection and analysis of filtered (dissolved) and unfiltered (whole water) sample pairs, with subsequent subtraction of the former from the latter [Office of Water Data Coordination (OWDC), 1978; 1982]. This approach is problematic because: (1) it does not provide total concentrations (with 'total' being defined as >95% of the constituent concentration present); (2) the small sample masses typically collected can be affected by inhomogeneities; and (3) the small sample masses, combined with the dilution effects of the associated water in whole-water samples, can lead to analytical detection limit problems (e.g., Horowitz, 1995). Based on the foregoing, the revised NASQAN program adopted a different approach. Field crews collected large-volume (10 to 100 l) depth- and width-integrated isokinetic whole-water samples. The intent was to collect sufficient volumes of whole water such that aggregate suspended sediment masses would be between 1.00 and 1.25 g. All the large-volume whole-water samples were shipped to a central location for subsequent de-watering by flow-through centrifugation and total trace element analysis.

All attempts at developing regression models for predicting suspended sediment-associated trace element concentrations using either discharge or suspended sediment concentration as the independent variable proved inadequate (based on $R^2$). However, site-specific intra- and interannual suspended sediment-associated trace element variations were markedly less (usually no more than a factor of two) than those for either discharge or suspended sediment concentrations in all the NASQAN basins (Table 3). In fact, except where concentrations approach the reporting limit, the differences between site-specific interannual trace element means/medians typically do not exceed the errors (generally ±10%) associated with the analytical methods used to determine them (Table 3). With the failure of the regression model approach, this relative lack of variability provided one of the few means of calculating annual sediment-associated trace element fluxes, through the use of derived mean/median concentrations. Errors associated with the use of this approach ranged from <1% to as much as 75% and were determined using the same method employed with the suspended sediment fluxes, by comparing the differences between the actual as opposed to the calculated (based on the selected mean/median concentrations) summed instantaneous daily fluxes for actual samples. NASQAN suspended sediment-associated P and organic carbon were determined by both the traditional/regulatory paired whole-water/filtered-water approach and directly (dewatered suspended sediment). The latter method consistently generated significantly higher (by factors ranging from 1.5- to 10-fold) concentrations/fluxes. The differences seem to result from a combination of sampling and analytical factors, with the latter likely to be more significant. This could be important for such issues as nutrient transport, eutrophication, algal blooms, coastal productivity, and the determination of global carbon and phosphorus cycles.

DISSOLVED TRACE ELEMENT CONCENTRATIONS AND FLUXES

Dissolved (filtered water-associated) trace element concentrations were determined as part of the original NASQAN program. Unfortunately, substantial portions of these data appear to have been affected by contamination introduced during collection, processing, and chemical analysis (Shiller and Boyle, 1987; Windom, et al., 1991; Taylor and Shiller, 1995). As a result, NASQAN ceased collecting this type of data in 1991. On the other hand, although the majority of most trace elements are trans-ported in association with suspended sediment, the dissolved fraction can make a substantial contribution (e.g., Horowitz, 1995). To prevent a recurrence of the problems encountered during the original NASQAN program, the revised program employed a newly designed clean protocol, as well as exten-sive QA/QC procedures, to ensure the integrity of the dissolved trace element data (Horowitz, et al., 1994). In fact, the revised NASQAN program may well be the first large-scale (national) program employing such a procedure.
Filtered water-associated (dissolved) trace element concentrations were markedly lower than those determined during the original NASQAN program. This seems more the result of the use of the clean protocol rather than to improvements in water quality, and produced a marked number of censored values. Censored values are an acceptable result in a standard regulatory/traditional monitoring program because concentration, per se, is a major issue of concern. Unfortunately, this is not the case for a flux-based program. As a result of the censored data, fluxes for filtered water-associated (dissolved) Ag, Pb, Cd, Cr, Co, V, Be, As, Sb, Hg, and Ti, as well as the total (filtered water plus suspended sediment-associated) fluxes for these constituents, could not be estimated. Hence, the tradeoff for obtaining uncontaminated/accurate dissolved trace element data is either an inability to calculate fluxes at all, or the need to

Table 3. Annual minimum, maximum, mean, and median values for discharge, suspended sediment, and sediment-associated chemical concentrations for selected NASQAN
employ much more sensitive (expensive) analytical procedures to permit flux esti-mations. Where sufficient uncensored dissolved concentration data were available, it would appear that the majority (\(\geq 70\%\)) of the Cu, Zn, Cr, Ni, Ba, P, As, Fe, Mn, and Al are transported in association with suspended sediment; Sr transport seems dominated by the dissolved phase, whereas the transport of Li and TOC seem to be divided equally between both phases. As such, a flux-based monitoring program must incorporate the determination of both dissolved as well as suspended sediment-associated parameters.

**CONCLUSIONS**

1) As a result of the behavioral differences between water and suspended sediment, a flux monitoring program may require two separate sampling schemes, one for water and one for suspended sediment.

2) Estimates of site-specific suspended sediment fluxes are determined by summing daily instant-aneous fluxes that were calculated based on log-log regression (rating curve) predictions of concen-tration developed by using a combination of historic and current discharge and suspended sediment concentration data; the errors associated with this approach are much less than \(\pm 10\%\) for the 3-year reporting period; however, for shorter periods of temporal resolution, the errors can increase sub-stantially.

3) Site-specific intra- and interannual variations in suspended sediment associated-trace element concentrations tended to be much smaller (usually less than a factor of two) than those for discharge or suspended sediment concentrations (typically more than an order of magnitude); as there were no strong interrelations between discharge or suspended sediment concentrations, and suspended sedi-ment-associated trace element chemistry, the use of derived mean/median concentrations provided one of the only means of calculating annual sediment-associated trace element fluxes.

4) The concentrations, and hence the annual fluxes, for suspended sediment-associated P and organic carbon, determined from the direct analyses of dewatered suspended sediment samples were markedly higher (fluxes were larger by factors ranging from 1.5- to 10-fold) than those determined using the more traditional/regulatory paired whole-water/filtered-water approach; this could be import-ant for such issues as eutrophication, algal blooms, coastal productivity, and models of global carbon and phosphorus cycles.

5) Filtered water-associated (dissolved) trace element concentrations were markedly lower than those determined during the original NASQAN program, as a result of the use of clean/ultraclean sam-pling, processing, and analytical protocols; as such, the fluxes for filtered water-associated (dissolved) Ag, Pb, Co, V, Be, Sb, and Se, as well as the total (filtered water plus suspended sediment-associated) fluxes for these constituents could not be estimated.

6) Based on the results from the first three years of the revised NASQAN program, the majority (\(\geq 70\%\)) of the Cu, Zn, Cr, Ni, Ba, P, As, Fe, Mn, and Al are transported in association with suspended sediment; Sr transport seems dominated by the dissolved phase, whereas the transport of Li and TOC seem to be divided equally between both phases.

**REFERENCES**


WATER QUALITY TRENDS OF LARGE FINNISH LAKES DURING 1970-1999

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Finnish Environment Institute, P.O.Box 140, FIN-00251 Helsinki, Finland

There are altogether 188 000 lakes in Finland. However, only 46 of them are larger than 100 km². These large lakes cover approximately 43 % of the total surface area and 68 % of the total volume of Finnish lakes. The objective of this study was to evaluate possible water quality trends in these lakes. The monitoring results of the large lakes during 1970 - 1999 revealed both positive and negative water quality changes. However, in most cases the water quality variables did not show any significant trend. Phosphorus concentrations showed a slight decreasing trend in lakes loaded by the pulp and paper industry or major municipalities in the past. The impact of intensive agriculture could be detected through slight elevated phosphorus levels in some lakes. Nitrogen did not show as clearly an increasing or decreasing trend.

INTRODUCTION

From the total estimated number of 8.3 million lakes on the Earth (Pourriot & Meybeck 1995), the proportion of 188 000 Finnish lakes seems to be quite small. However, only in few other European countries lakes are as numerous as in Finland. Lakes cover almost 10 % of the total area of Finland. Most of the Finnish lakes are small; approximately 99 % of them are smaller than 1 km². The number of so-called "large lakes" with a surface area of over 100 km² is 46. This is almost as much as the total number of large lakes in the other EEA (European Environment Agency) area countries altogether. The large lakes cover about 43 % of the total surface area (32 600 km²) and 68 % of the total volume (235 km³) of Finnish lakes (Raatikainen & Kuusisto 1990, Kuusisto 1992). The size of the biggest Finnish lake, L. Saimaa (1 147 km²), is far from the largest natural lakes in Europe, L. Ladoga (17 670 km²) and L. Onega (9 670 km²) in Russia and L. Vänern (5 670 km²) in Sweden. L. Saimaa is sometimes defined as one large labyrinthine lake system consisting of several lake basins having almost the same water levels and being connected with narrow sounds. Defined as such, the greater L. Saimaa system has an area of 4 380 km² and amazing number of 13 710 islands. In this study, the sub-basins are, however, handled as individual lakes.

The current Finnish surface water monitoring system is based on the guidelines prepared by the EEA to create a European monitoring and observation network for inland waters, EUROWATERNET. The Finnish EUROWATERNET, designed by the Finnish Environment Institute, consists of 264 lake sampling sites and 194 river sampling sites. The sites were selected among the existing national, regional and statutory monitoring network sites. The network covers all the large lakes in Finland. The Finnish Water Authorities spend about 1 million euros a year for maintaining the national EUROWATERNET.

In addition to the Finnish EUROWATERNET a great number of other regional and statutory monitoring programmes are carried out by the Regional Environment Centres, major cities, and industrial plants. The obligation to statutory monitoring (i.e. local pollution control) is based on the Finnish Water Act. Individual statutory monitoring programmes cover ca. 1 700 polluters and consist of about 4 500 sampling sites around the polluted parts of Finnish watercourses. In addition to these ongoing monitoring programmes, the Finnish Water Authorities made two extensive lake surveys covering almost 1000 randomly selected lakes in the 1980s and 1990s. All the 46 large lakes were included in the latter survey. The current Finnish surface water monitoring system will be reviewed and revised to meet the requirements set by the EU Water Framework Directive within the next few years.

This paper deals with the monitoring results of the 46 large Finnish lakes during 1970 - 1999. The aim of the study is to evaluate and present the overall water quality of these lakes and to detect possible long-term trends of ecologically important variables.
MATERIAL AND METHODS

Description of the study sites

Large lakes are mainly situated in the Lake District of central and eastern Finland (Figure 1). Three of the large lakes are over 1000 km²: L. Saimaa (1 147 km²), L. Inari (1 102 km²) and L. Päijänne (1 054 km²). Over a half of the large lakes are smaller than 200 km². Most of the large lakes are natural of their origin. However, two of them are artificially made for hydropower purposes: the Lokka reservoir (317 km² at the mean water level) and the Porttipahta reservoir (125 km²). The sizes of these northern Finland reservoirs may vary enormously, from 216 to 417 km² in the Lokka reservoir and from 34 to 214 km² in the Porttipahta reservoir. These reservoirs, built in the turn of the 1960s and 1970s, have undergone a drastic development from poor quality reservoirs, widely known for their methyl mercury problems, to more or less natural humic lakes (Lepistö & Pietiläinen 1996). The other 44 lakes were formed or reshaped by the last glaciation period about 10 000 thousand years ago. Most of the study lakes are originally oligotrophic or oligo-dystrophic. The present state of the lakes, however, is not quite pristine any longer due to more or less significant human impact.

Sampling and chemical analyses

Water samples were collected from the surface layer (1 m or 0 - 2 m composite samples) of a representative area (often the deepest point) of each lake most but not every August during the 30 year period 1970 - 1999. The depth and timing of sampling were aimed at obtaining a reliable and comparable picture of the chemical status of the trophic zone of each lake during the period when most excessive algal blooms and other nuisance eutrophication phenomena are likely to occur. Water quality data from the deepest water column of some lakes were also included in the study in order to reveal possible oxygen deficiency problems and potential release of algal growth enhancing internal phosphorus loading. Two sub-basins of L. Saimaa and L. Näsijärvi were also studied in order to demonstrate the positive effect of water protection measures taken into use in these areas, which had been heavily polluted by point source loaders in the past.

The variables included in the study were: alkalinity, pH, conductivity, suspended solids, chemical oxygen demand, total nitrogen, nitrate nitrogen, ammonium nitrogen, total phosphorus, phosphate phospho-rus, chlorophyll a, colour, and oxygen. All analytical work was carried out according to the standard methods used by the Finnish Water Authorities.

We present some basic statistical indicators (mean, median, percentiles) and use the Kendall Tau b test to detect possible long term changes in water quality. The water quality of large lakes is compared, if available, to that of the whole Finnish lake population. The estimate for the whole lake population is based on the statistical sampling of 873 lakes carried out in a joint Nordic Lake Survey in 1995 (Henriksen et al. 1997, Mannio et al. 2000).
RESULTS AND DISCUSSION

Alkalinity and pH

Finnish lakes are relatively sensitive to acidification and pH changes due to the base-poor geology and subsequent low acid buffering capacity of sub-surface soil in the country. This is also reflected in low alkalinity values and low buffering capacity of Finnish surface waters, i.e. our lakes are generally soft. The median alkalinity values for the large lakes (0.14 mmol L\(^{-1}\)) and for the whole Finnish lake population (0.12 mmol L\(^{-1}\)) were relatively low (Figure 2). However, acid neutralising capacity is even lower in many areas of Norway and Sweden (Henriksen et al. 1997). A slight but statistically significant increasing trend of alkalinity was observed in 34 large lakes during 1970 - 1999 (Table 1). This is probably due to a clear reduction in acid sulphate deposition since the 1970s in Europe (Skejelvåle et al. 2000). Alkalinity decreased in one study lake and remained stable in 11 lakes. pH values increased in every third lake. Decreasing pH trends were not observed at all. As much as 3000 Finnish lakes are estimated to still get a sulphur load exceeding the critical values (Henriksen et al. 1997). The large lakes are, however, not as sensitive to rapid alkalinity and pH changes as small lakes.
Conductivity

Human impact can easily be seen on the elevated conductivity levels in the study lakes. The median value of 4.6 mS m⁻¹ is clearly higher than that of an average Finnish lake (Figure 2). The median value for the large lakes is, however, far from those observed widely e.g. in many central and southern European lakes (Skejelvåle et al. 2000). Conductivity showed an increasing long-term trend in 14 cases, a decreasing trend in 6 cases, and no trend at all in 26 cases (Table 1).

Suspended solids

The median suspended solids concentration of the large lakes (1.1 mg L⁻¹) indicates that large Finnish lakes are typically clear water lakes. However, the impact of clayish soil and intensive agriculture in the drainage basin can be observed in several lakes. A good example of this is L. Onkivesi in central Finland with its median suspended solids value of 6.4 mg L⁻¹ (Figure 2). The visibility depth of the study lakes varied a lot, from about half a metre in L. Onkivesi to nearly 10 metres in L. Puruvesi, a pristine lake in eastern Finland. The great variability in Secchi depth can be largely explained by the great variability in the suspended sediment and the humus content of the water.

Chemical oxygen demand

Chemical oxygen demand (CODₘₜₙ) is a widely used estimator of the amount of organic substances in the water. In Finland a major proportion CODₘₜₙ in fresh waters is due to humic substances. In the study lakes the median CODₘₜₙ value (7.7 mg L⁻¹) was quite high. Decreasing CODₘₜₙ trends were observed in 9 study lakes while increasing trends were not detected at all (Table 1). The high number of decreasing trends and the total lack of increasing CODMn trends clearly show that water quality has improved in respect to organic matter loading. The decrease of organic matter loading is mainly due to the major improvements in industrial and municipal waste water treatment systems but possibly also due to the decrease of organic load originating from peat lands which were drained for wood production in the 1960s and the 1970s.

Total nitrogen

Nitrogen and phosphorus are the two main nutrients causing eutrophication of surface waters. In Finland the role of nitrogen seems not to be as pronounced as that of phosphorus in controlling eutrophication. The classifications of the trophic status of surface waters are usually based on the amount of phosphorus and/or chlorophyll a in the water. The median total nitrogen concentration in the study lakes (400 µg L⁻¹) was approximately the same as the estimate for the whole Finnish lake population (Figure 2). These values are far from the elevated nitrogen levels (up to several thousands) observed widely in central and southern Europe (EEA 1998). Total nitrogen correlated quite strongly with total phosphorus (r² = 0.712**) suggesting that both the nutrients originate from the more or less same sources. Nitrogen concentrations were highest in southern Finland and in the Lokka reservoir in the north.

Total phosphorus

The median total phosphorus concentration was slightly lower in the studied large lakes (9.8 µg L⁻¹) than in Finnish lakes on average (13 µg L⁻¹) (Figure 2). This is partly due to longer retention time in large lakes and subsequent efficient sedimentation of small particles rich in phosphorus. The median total phosphorus value for the study lakes is clearly lower than the values of 50 - 500 µg L⁻¹ often observed in various central and southern European lakes (EEA 1998). In large European lakes phosphorus concentrations seldom exceed 125 µg L⁻¹ (Kristensen & Hansen 1994). These numbers indicate the good or even excellent status of Finnish large lakes in the European scale. According to the boundary values proposed by OECD (1982) half of the large Finnish lakes fall into a group of oligotrophic water bodies. Based on total phosphorus concentrations, only two of the 46 study lakes were regarded as eutrophic and none as hypertrophic. In addition to the quite strong correlation with nitrogen, total phosphorus correlated even more strongly with chlorophyll a (r² = 0.889**) and suspended solids (r² = 0.935**). The strong correlation between total phosphorus and chlorophyll a strongly suggest that phosphorus is the major algal growth-regulating factor in large Finnish lakes.
Figure 2. Cumulative distribution of the median values of some water quality variables describing the basic characteristics of 46 largest Finnish lakes. Estimates of the median values for the whole Finnish lake population (size > 4 hectares) are also given when available (Mannio et al. 2000).
Table 1. Statistically significant trends (Kendall Tau test, p < 0.01) of some water quality variables in the large lakes during the late productive season (August) in 1970-1999.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Decreasing trend</th>
<th>No trend</th>
<th>Increasing trend</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alkalinity</td>
<td>1</td>
<td>11</td>
<td>34</td>
</tr>
<tr>
<td>pH</td>
<td>0</td>
<td>31</td>
<td>15</td>
</tr>
<tr>
<td>Conductivity</td>
<td>6</td>
<td>26</td>
<td>14</td>
</tr>
<tr>
<td>Chemical oxygen demand</td>
<td>9</td>
<td>37</td>
<td>0</td>
</tr>
<tr>
<td>Total nitrogen</td>
<td>5</td>
<td>41</td>
<td>0</td>
</tr>
<tr>
<td>Total phosphorus</td>
<td>8</td>
<td>34</td>
<td>4</td>
</tr>
<tr>
<td>Chlorophyll a</td>
<td>3</td>
<td>42</td>
<td>1</td>
</tr>
<tr>
<td>Colour</td>
<td>8</td>
<td>36</td>
<td>2</td>
</tr>
</tbody>
</table>

Figure 3. Examples of phosphorus trends observed in the large lakes in 1970 -1999.
Figure 4. Examples of major water quality improvements in the hypolimnion of two sub-basins of L. Saimaa and L. Näsijärvi. The impact of extensively reduced point source loading can be clearly seen on increased oxygen saturation values and decreased phosphorus concentrations in both lakes.

In the case of phosphorus, both clearly increasing and decreasing long-term trends were observed (Figure 3). Phosphorus concentration increased in 8 lakes and decreased in 4 lakes. In 34 lakes no statistically significant trend was observed at all. However, in the most heavily polluted parts of some study lakes a recovery from poor water quality basins to more or less excellent surface waters was observed. Good examples of the impact of improved water protection measures and renewed sewage water purification systems are two restricted areas of L. Saimaa and L. Näsijärvi (Figure 4). These sub-basins were highly polluted by pulp and paper industry and municipalities in the 1970s. Since the first half of the 1980s a more or less drastic recovery was observed in both lakes. The recovery can be seen in the increased oxygen contents and decreased phosphorus concentrations in the hypolimnion of both lakes. Nowadays the oxygen situation has improved and the concentration of phosphorus has decreased to mesotrophy indicating levels.

**Chlorophyll a**

Chlorophyll \(a\) is a widely used estimator of the trophic state of surface waters. The large Finnish lakes have relatively low chlorophyll \(a\) concentrations compared to those observed in central Europe. According to the limit values presented by OECD (1982) the median value of 4.2 µg L\(^{-1}\) for large lakes suggest them being mesotrophic on average. Based on median chlorophyll \(a\) concentrations, ten of the study lakes are regarded as eutrophic and one, L. Onkivesi, hypertrophic. This means that total phosphorus and chlorophyll \(a\) values give quite a different picture of the trophic state of the study lakes. One possible explanation for this disparity is the role of humic substances and dark water in elevating chlorophyll \(a\) content abnormally high compared to clear water lakes.

**Colour**

Finland is characterised by forests and peatlands capable of producing large amounts of organic matter in the runoff water and, thus, causing the high colour values of Finnish inland waters. The median colour value of large lakes (38 mg L\(^{-1}\) Pt) is not as high as the estimate of the whole
Finnish lake population (100 mg L\(^{-1}\) Pt) (Kortelainen 1993). The colour values of the 46 large lakes varied from 5 up to 120. The variation is mainly due to the differences in the bedrock and soil structure and land use in the drainage basin.

**Algal growth limiting nutrient**

It is widely known that phosphorus and nitrogen are the two main nutrients capable of enhancing and regulating primary production in fresh waters. Phosphorus is considered as the major nutrient limiting algal growth in Finnish lakes (Pietiläinen 1997). The role of nitrogen seems to be more important in rivers and, especially, in the Baltic Sea (Kivi et al. 1993, Pietiläinen 1997). Nitrogen may also increase primary production in some lakes, especially in more eutrofied lakes. In most pristine lakes, which are typically small, phosphorus and nitrogen concentrations are often so low that both of the nutrients can potentially limit algal growth.

The limiting role of nutrients can be estimated either by measuring the concentrations and ratios of nitrogen and phosphorus in lake water or by using algal bioassays made in the laboratory or in the lake itself. In this study the limiting roles of nitrogen and phosphorus were estimated indirectly by measuring their concentrations and ratios in each study lake. We only used biologically available inorganic forms of nitrogen (nitrate, ammonium) and phosphorus (phosphate) when assessing the limiting nutrient. Only the samples taken during August were included in order to cover the most favourable season for blue-green algal blooms.

Phosphorus revealed to be the main limiting nutrient in the study lakes: 8 of the lakes were considered as being limited by nitrogen, 10 simultaneously by both nitrogen and phosphorus, and 28 by phosphorus alone. This conclusion is based on the fixed boundary values given by Forsberg & Ryding (1980). Forsberg and Ryding presented that the dissolved inorganic nitrogen:phosphorus ratio of 5 or less refers to nitrogen limitation, and the ratio of 12 or greater to phosphorus limitation. If the ratio is between 5 - 12 both nutrients can be potential algal growth limiting factors. However, if both nitrogen and phosphorus are found in excessive quantities they do not limit algal growth at all. In such case some other variables, such as light and temperature, become the factors regulating algal growth.

Phosphorus limitation was often very strong in the study lakes, i.e. bioavailable phosphorus (PO\(_4\)-P) concentrations were extremely low in most lakes while bioavailable nitrogen (NH\(_4\)-N + NO\(_2\)-N) concentrations were one or even two orders of magnitude higher at the same time (Figure 5). As rough boundary values for lakes, practical experience suggests that concentrations of biologically available phosphorus of less than 5 µg L\(^{-1}\) indicate potential phosphorus limitation, while concentrations of biologically available nitrogen less than 20 µg L\(^{-1}\) suggest nitrogen limitation (Ryding & Rast 1989). If the concentrations of both nutrients are higher than these values neither nitrogen nor phosphorus is the primary algal growth limiting factor. The median phosphate phosphorus concentration was 5 µg L\(^{-1}\) or lower in 44 lakes and the median nitrogen concentration 20 µg L\(^{-1}\) or higher in 28 lakes. These values also suggest the major role of phosphorus in controlling primary production of the study lakes. Only the most eutrofied (= phosphorus rich) study lakes, L. Onkivesi and L. Vesijärvi, were supposed to be strongly limited by nitrogen from time to time. These lakes are famous for their impressive blue-green algal blooms.
Figure 5. Average dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) concentrations and ratios in the surface waters of the large lakes in August 1970 - 1999.

CONCLUSION

The results show that the state of the 46 biggest Finnish lakes is in most cases good or excellent in respect to acidification and eutrophication. The results also indicate that the water and air pollution protection measures used during the last few decades have improved water quality in many lakes. The recovery is best seen in acidification indicators; alkalinity and pH values are now higher than in the 1970s and 1980s in almost all the lakes. The impact of improved sewage water purification can be observed in many lakes in the form of decreased phosphorus concentrations. However, phosphorus concentrations have slightly increased in lakes polluted by agriculture and other non-point source loading.

REFERENCES


THE GERMAN GROUNDWATER MONITORING NETWORK

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During the last few years several groundwater monitoring networks were established in Germany. These networks are the general groundwater monitoring network, the nitrate groundwater monitoring network and the pesticides groundwater monitoring network. The different investigative concepts and selected results of the networks are presented.

INTRODUCTION

In Germany the federal states are responsible for the monitoring and protection of groundwater. Individual monitoring programmes have been in place in the various federal states for many years. In the past, data compilations for the whole of Germany have been drawn up only sporadically. During the last few years there has been a steady growth in the number of reports demanded by the European Commission and the European Environment Agency. It was becoming more and more difficult for the federal states and the Joint Water Commission of the Federal States, the (LAWA) to fulfill all existing reporting obligations in time, because it is very time-consuming process to collect the relevant data. Therefore the LAWA decided to implement a federal groundwater monitoring network in close co-operation with the Federal Environmental Agency.

The German Groundwater Monitoring Network consists of three sub-monitoring-network-systems, the general groundwater monitoring network, the nitrate monitoring network, and the pesticides monitoring network. The federal states are responsible for the selection of suitable sampling sites, sampling analysis, and data transmission. The task of the Federal Environmental Agency is to collect and verify the data and report it to the European Environment Agency, the European Commission and other relevant institutions.

GENERAL GROUNDWATER MONITORING NETWORK

The investigative concept for the General Groundwater Monitoring Network is based on the idea of providing a representative survey of groundwater quality. Of major interest are the impacts of diffuse (non-point source) anthropogenic inputs of contaminants, for example nitrates, pesticides, acidifying components and other pollutants, on groundwater quality. Sampling sites were selected from existing monitoring networks of the federal states. They reflect the known distribution of contaminated and uncontaminated groundwater bodies within each state. Sampling sites representing contaminated groundwater bodies are located in regions in which groundwater contamination is more frequent.

<table>
<thead>
<tr>
<th>Federal state</th>
<th>Sampling sites</th>
<th>Area in km²</th>
<th>Federal state</th>
<th>Sampling sites</th>
<th>Area in km²</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bremen</td>
<td>2</td>
<td>400</td>
<td>Saxony-Anhalt</td>
<td>51</td>
<td>20400</td>
</tr>
<tr>
<td>Hamburg</td>
<td>5</td>
<td>800</td>
<td>Hesse</td>
<td>49</td>
<td>21100</td>
</tr>
<tr>
<td>Berlin</td>
<td>5</td>
<td>900</td>
<td>Mecklenburg-Western Pomerania</td>
<td>38</td>
<td>23800</td>
</tr>
<tr>
<td>Saarland</td>
<td>6</td>
<td>2600</td>
<td>Brandenburg</td>
<td>60</td>
<td>29100</td>
</tr>
<tr>
<td>Schleswig-Holstein</td>
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<td>15700</td>
<td>North Rhine-Westphalia</td>
<td>77</td>
<td>34100</td>
</tr>
<tr>
<td>Thuringia</td>
<td>30</td>
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<td>Baden-Württemberg</td>
<td>79</td>
<td>35800</td>
</tr>
<tr>
<td>Saxony</td>
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<td>18300</td>
<td>Lower Saxony</td>
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<td>47400</td>
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<tr>
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<td>50</td>
<td>19800</td>
<td>Bavaria</td>
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<td>70600</td>
</tr>
<tr>
<td>total:</td>
<td></td>
<td></td>
<td></td>
<td>791</td>
<td>357100</td>
</tr>
</tbody>
</table>

Table 1: Number of sampling sites in each of the federal states
The federal network consists of about 800 sampling sites distributed more or less equally over the 16 federal states (Figure 1). The number of sampling sites in each state depends on its size (Table 1).

![Map of General and Nitrate Groundwater Monitoring Network sampling sites](image)

**Figure 1: Map of General and Nitrate Groundwater Monitoring Network sampling sites**

At present there is an average of one sampling site per 450 square kilometres. City states such as Berlin, Hamburg and Bremen are overrepresented in the number of sampling sites in relation to their area.

This apparent over weighting was found necessary in order to better describe groundwater quality in these states and the variations therein. The Federal States, in close co-operation with the Federal Environmental Agency, established the information necessary to characterise the sampling site and its catchment area. Important informational parameters are location and type of sampling site, petrographic composition of the aquifer and predominant land use in the catchment area.

In addition to temperature, conductivity and acidity, analytical parameters for characterising groundwater quality include all major anions and cations, selected heavy metals, metalloids, some organic components and selected pesticides (Table 2).
Table 2: Informational and analytical parameters to characterise the General Groundwater Monitoring Network sites

<table>
<thead>
<tr>
<th>Sampling site:</th>
<th>Important parameters analysed:</th>
</tr>
</thead>
<tbody>
<tr>
<td>Code number and location of sampling site</td>
<td>Groundwater specific data</td>
</tr>
<tr>
<td>Altitude of site and filter position</td>
<td>Temperature; pH; electrical conductivity.</td>
</tr>
<tr>
<td>Type of sampling site (well, spring etc.)</td>
<td>O₂; NH₄⁺; NO₂⁻; NO₃⁻; o-PO₄³⁻; Cl⁻; SO₄²⁻; B⁻;</td>
</tr>
<tr>
<td>Land use</td>
<td>DOC K; Na⁺; Ca²⁺; Mg²⁺</td>
</tr>
<tr>
<td>Hydrogeology (stratigraphy, petrographie)</td>
<td>Heavy metals/metals/metalloids Al; As; Pb; Cd; Cr; Fe; Cu; Mn; Ni; Zn</td>
</tr>
<tr>
<td>Type of aquifer (porous rock, fissured rock, karst)</td>
<td>Aliphatic halogenated hydrocarbons</td>
</tr>
<tr>
<td>River basin</td>
<td>Pesticides</td>
</tr>
</tbody>
</table>

In general sampling at the sites should be conducted at least twice a year. The data are reported to the Federal Environmental Agency once a year using a specific data transfer format. The federal states have also agreed to deliver data from earlier years for these 800 sampling sites, starting with 1990. In June 2000 a database was installed at the Federal Environmental Agency to store, verify and summarise the information delivered. The data are verified prior to inclusion in the database by checking for compliance with certain formal criteria: data is verified by use of the ion balance for each sample and by comparing the reported sample concentrations with known concentration ranges of similar types of groundwater and with samples from the sites where they were measured.

The General Groundwater monitoring network is intended to give a representative picture of groundwater quality in Germany. This is being verified inter alia by comparing the distribution of the nitrate concentrations determined from this network with the findings of the LAWA (1995) nitrate report, based on the analysis of data from several thousand sampling sites (Figure 2). With the exception of the nitrate concentration classes of less than 1 mg/l and 50 to 90 mg/l the comparison showed very good agreement, however because up to now nitrate concentrations for the general Groundwater monitoring network in 1995 are available for only 468 sampling sites, this question can not yet be answered conclusively.

At least for the nitrate parameter it seems that the groundwater network presents a realistic and representative picture of the distribution of nitrate in German groundwater. Work to verify representativity will be continued to include other substances such as pesticides and acidity.

Figure 2: Frequency distribution of nitrate in German groundwater according to the LAWA-Report (1995) compared with the results of measurements in 1995 at 468 sites of the Federal General Groundwater Monitoring Network
NITRATE GROUNDWATER MONITORING NETWORK

The Nitrate Directive (COUNCIL DIRECTIVE 1991) requires regular reporting about the preventive nitrate reduction measures taken by the member countries. It provides that, at the end of each four-year programme (1995-99, 2000-2003), and for each water monitoring report of measures associated with this programme, a report describing the situation and its development be submitted to the Commission. Germany has refrained from designating vulnerable zones since the measures to limit nitrate contamination are applied throughout its territory.

To depict the existing nitrate contamination of German groundwater resources and to evaluate the effectiveness of the measures enforced, a special Nitrate Groundwater Monitoring Network was established. It is focused on regions with significant groundwater contamination by nitrates. The network consists of 181 monitoring sites predominantly situated in the upper groundwater layer. Sampling sites selected had to be significantly influenced by nitrate from agricultural sources (Figure 1).

This means that unlike the General Groundwater Monitoring Network, the Nitrate Groundwater Monitoring Network is not representative of the nitrate status of German groundwater. Compared to the "representative" nitrate distribution represented in Figure 2, a significant shift is to be seen for the nitrate distribution of the nitrate network (Figure 3).

The reporting guidelines for the Nitrates Directive mandates that trends in nitrate contamination must be evaluated. The general trend is derived from the change in concentrations between the first and the second sampling of the monitoring network, carried out in 94/95 and 98/99 respectively. An upward trend was found for about 37 % of the measuring sites whereas 57 % showed a downward trend. On the whole, a decrease in nitrate pollution can be observed in affected areas.

PESTICIDES GROUNDWATER MONITORING NETWORK

A very different component of the German monitoring network is the pesticide residues groundwater monitoring programme. Data collection here is not based on a fixed set of sampling sites but rather summarises all available information about pesticide detects in groundwater. The aim of the network is to collect data on pesticide detects (analytical results) for use in authorisation of the individual pesticides.

As part of the pesticide authorisation procedure, the behaviour of pesticides in soil and on their way into the groundwater is modelled to predict pesticide behaviour but there was concern that there could be a discrepancy between the real behaviour of pesticides and the results obtained from the model. The monitoring programme was started in 1989, when the limit value for pesticides in drinking water was established. The first step towards identifying problematic pesticides was an agreement with the federal states for them to deliver all data on pesticides in water to the Federal Environmental Agency.
In 1998 data reporting was revised. Data will now be reported separately for different types of water (that is groundwater, surface water, drinking water and bank filtered water) and related to the number of measuring sites analysed. Detects have to be reported separately for each pesticide and year, according to 4 concentration classes (n.d. = not detected; detected at a concentration below 0.1 µg/l; > 0.1 - 1.0 µg/l; and > 1.0 µg/l). In addition the federal states agreed to report their data to the Federal Environmental Agency regularly once a year. Since 1990 a list of the pesticides most frequently detected has been published by the Federal Environmental Agency and has been made available for use in pesticides authorisation. Many frequently detected pesticides, are now no longer authorised or the authorisation includes use restrictions.

Of the pesticides most frequently detected between 1990 and 1995 there are almost no significant differences except for the occurrence of 2,6-dichlorobenzamide for which testing has only been carried out since 1996. Monitoring data proved to be useful in the re-authorisation of mobile pesticides that have been in use for several years. Frequent detects indicate that there may still be unknown pathways of pesticide movement to groundwater that have to be examined. For two pesticides (Isoproturon and Terbutylazin) post-authorisation monitoring was imposed combined with a shortened authorisation period of just 4 years. Within this time the applicant has to investigate the cause of pesticide occurrence in groundwater. Based on the results of this investigation a decision will be made as to whether regular authorisation is possible or whether a limited authorisation is required.

The Pesticide Groundwater Monitoring Approach provides no information about the areal extent of groundwater contamination by pesticides. A special "groundwater report" of the
Federal States, entitled "Grundwasserbericht – Pflanzenschutzmittel", published at the end of 1997 (LAWA 1997), describes the results obtained at 12,886 sampling sites throughout Germany. The report indicates that 9.7 % of the sampling sites a pesticide concentration of 0.1 µg/l was exceeded. At 18.6 % of the sites pesticides were detected at concentration levels below 0.1 µg/l and at 71.7 % no pesticides were detected (Figure 4).

![Figure 4: Pesticide detects in shallow groundwater (LAWA 1997)](image)

The trend in pesticide contamination of groundwater in Germany within the last five to ten years is yet to be fully evaluated. It is expected that the measures to identify and eliminate frequently found pesticides as well as restrictive authorisation have caused groundwater contamination to decrease. Hopefully success will become visible over the next years and be apparent at the 800 sampling sites of the General German Groundwater Monitoring Network.

CONCLUSION

The Water Framework Directive will bring further reporting requirements in its wake. Most of these requirements can be fulfilled by the existing network. Further sampling sites may have to be added to the network however, the current parameters measured should adequately describe the status of groundwaters as required by the Water Framework Directive.

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NEW APPROACHES TO WATER MONITORING AND ASSESSMENT IN BOSNIA & HERZEGOVINA

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Unexpected circumstances in Bosnia and Herzegovina (B&H) during the last decade of the past century designated prospective feasibility in modelling a new up-to-date monitoring system.

Traditional politics in water management should be transformed into "Integrated Water Management", already promoted in almost all European countries. The new system for enhanced monitoring and assessment activities is proposed within the framework of a just ended project, "Water Sector Institutional Strengthening".

The objective of this paper is to report on ongoing activities in establishment of new model for management of "River Basin Type", setting forth new approaches on future monitoring and assessment of water resources in B&H, giving special attention to the transboundary water pollution control. Since most of the water resources in B&H are of transboundary character (rivers Sava and Drina in Danube river basin and rivers Cetina and Neretva in Adriatic Sea Watershed), the neighbouring countries, Republic of Croatia and Federal Republic of Yugoslavia, are to be included in monitoring and assessment programs. The joint monitoring program has already started at Cetina river basin, which seems to be of interest for both countries at the moment. Considering current circumstances, it is presumed that the mentioned activities should enable a prospective co-operation at international level that was interrupted years ago.

INTRODUCTION

Background

Bosnia and Herzegovina (B&H) is a very young state, once one of the states which comprised the former Yugoslavia. Dayton Peace Accord in 1995 determined the end of the war and provided a Constitution of State of B&H. According to the Constitution, the state is composed of two entities: the Federation of B&H (FB&H) and the Republic of Srpska (RS).

The FB&H is decentralised entity and consists of ten Cantons where each one of these is governmental entity, with a high degree of discretion in establishing fundamental functions and carrying them out. Regarding the environment, all of the Cantonal Constitutions also state that both the Federation and the Cantons are responsible for environmental policy, in accordance with the Federal Constitution. The Cantons have the possibility to transfer their responsibility regarding the environment to the municipalities and/or the Federation. On the Federal level, the relevant Ministries are Ministry of Forestry, Agriculture and Water Management and Ministry for Physical Planning and Environment.

The RS is a centralised entity divided in to 7 regions. Local administration exists only at the municipal level. The Republic is responsible for ensuring environmental protection while municipalities, in accordance with law, take care of meeting specific needs of citizens regarding environmental protection. On the Republic level, the relevant Ministries are Ministry for Forestry, Agriculture and Water Management and Ministry for Urbanism, Housing-Communal Work, Civil Engineering and Ecology.

Unfortunately, the co-operation between the above-mentioned entity ministries either does not exists or is negligible. Laws and by-laws for both entities are created independently and do not comply with each other, resulting in non-harmonised activities of these bodies. Additionally, all existing legislation is insufficient and is not yet in compliance with the EU principles and standards.

However, all recent activities carried out in the water sector are directed toward establishment of integral water management. As the cooperation and co-ordination between responsible
bodies is very weak, the priority is to establish stronger bounds between the entities. The first step is already made and two Steering Committees, both for environment and water, are established. They act as the inter-entity bodies responsible for co-ordination of work related to the environmental and water issues between the two entities.

**Current Activities**

Water management issues are treated under the Water Law, different for both entities. In accordance with these laws, two major watersheds are recognized at the territory of B&H:

- **Sava River watershed**, as a part of Danube River Basin, consisting of rivers Una, Vrbas, Bosna, Drina and Sava (surface area of 38,719 km²), and
- **Adriatic Sea watershed**, consisting of rivers Neretva, Trebisnjica and Cetina, and nearby coastal catchment area (surface of 12,410 km²).

To overcome the above-mentioned problems, European Union initiated the "Program for Environment in B&H". In the scope of this program, the EU supported and financed the re-organization and re-establishment of the water sector as a whole throughout the Project "Water Sector Institutional Strengthening". The project involved five essential aspects: (i) Institutional Aspect; (ii) Legal Aspect; (iii) Financing and Cost Recovery Aspect; (iv) Water Quality Aspect; and (v) Human Resources Aspect.

The Project has proposed for two relevant ministries from both entities to join in one Ministry per entity called Ministry of Water and Environment, with the same responsibilities and rights in both entities. The Ministry should treat all the problems related to water and environment, as it has become a common practice in almost all west European countries. This would open the way for easier communication between Ministries regarding the environmental issues and it would promote further cooperation regarding decision-making policies.

The background of the entire project of "Water Sector Institutional Strengthening" is the promotion of new water resource management system, based on the principle of "Integrated Water Management" for each river basin. The River Basin Bodies (RBBs) composed of experts from both entities are going to be established for eight (8) identified river basins in B&H. The RBBs are going to be governed from one centre, thus minimising the conflict of interests even at the lowest possible level. Geographically, three (3) out of eight (8) river basins belong to the Adriatic Sea Watershed while five (5) of them belong to the Sava River Watershed.

A new modified methodology related to research, monitoring, sampling, investigation and data processing and storage shall be in accordance with the European practice, legislative, norms and standards, respecting sustainable use and exploitation of natural water resources.

In 1998, "Life Third Countries Program" financed the project "Institutional Strengthening of Mediterranean Action Plan (MAP) Office for B&H". Within the framework of this project, MAP Office staff has drawn up the "National Action Plan (NAP) for Mediterranean Region of B&H for Pollution Reduction and Integral Management Application". The objective of NAP was to provide the guidelines for achievement of sustainable development in the Mediterranean region of B&H, defining policy, strategy and actions for prevention, control and reduction of environmental degradation and integral coastal and river basin management.

**MONITORING SYSTEM**

The new river basin management model foresees complete modification of old water monitoring and control systems. Since most of the rivers have inter-entity and transboundary character, the most important question in the new model shall be how to coordinate and harmonize all activities in order to generate real and satisfactory, as well as feasible water quality control system.

**Situation before the war**

In the pre-war period, the Republic’s Hydro-Meteorological Institute performed monitoring of surface waters, assessment of biological indicators and reported obtained data on a regular basis, while the groundwater monitoring was performed occasionally. All the data was stored in
the form of Yearbooks as a hard copy, and was not available to the public. The monitoring process involved determination of standard biological and physical-chemical parameters, three to four times per year at 58 monitoring stations (related to water quality and quantity data in parallel).

Table 1 gives the list of key parameters for each river basin determined before the war, as well as the pollution loads from point and diffused sources. The heaviest pollution load is concentrated in Sava River watershed (up to 88 %). The most polluted river in Sava River watershed was river Bosna, with over 50 % of the population and about 30 % industry (mining, heavy industry, and chemical industry) being located in this catchment area. The Adriatic Sea watershed contributed with 12 % of the pollution load. Neretva river basin, including some parts of Trebisnjica river, is the most densely populated area, while industrial production mainly consists of the metal finishing industry.

### Table 1. Pollution Sources in B&H (by River Basin) [1]

<table>
<thead>
<tr>
<th>Basin</th>
<th>Municipal Waste Waters</th>
<th>Industrial Waste Waters</th>
<th>Diffused Pollution</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Q (m³/s)</td>
<td>SS (kg/d)</td>
<td>COD (kg/d)</td>
</tr>
<tr>
<td>Una</td>
<td>0.558</td>
<td>13,418</td>
<td>27,153</td>
</tr>
<tr>
<td>Vrbas</td>
<td>0.801</td>
<td>19,104</td>
<td>38,271</td>
</tr>
<tr>
<td>Bosna</td>
<td>2.951</td>
<td>70,414</td>
<td>144,769</td>
</tr>
<tr>
<td>Drina</td>
<td>0.325</td>
<td>7,871</td>
<td>15,800</td>
</tr>
<tr>
<td>Sava</td>
<td>0.591</td>
<td>13,664</td>
<td>27,800</td>
</tr>
<tr>
<td>Cetina</td>
<td>0.068</td>
<td>1.694</td>
<td>3.363</td>
</tr>
<tr>
<td>Trebi, njica</td>
<td>0.604</td>
<td>14,389</td>
<td>29,009</td>
</tr>
<tr>
<td>Neretva</td>
<td>0.604</td>
<td>14,389</td>
<td>29,009</td>
</tr>
<tr>
<td>Total</td>
<td>5.90</td>
<td>140,574</td>
<td>286,165</td>
</tr>
</tbody>
</table>

* PE = population equivalent
** Settlements over 5000 inhabitants

### Situation after the war

After severe destruction during the war, system of monitoring and control in the post war period has not yet been fully re-established. Comparing to the pre-war situation, only few stations for monitoring of water quantity and quality are in function. Several new hydrological stations for water quantity monitoring have been installed. Figure 1 compares the number of installed monitoring stations in pre- and post-war period.

At present, 49 hydrological stations are in function at the whole territory of B&H and water quality is being monitored at only 23 profiles (Table 2). It should be noted that due to the war, most of the industries have been destroyed, damaged or working with half of their pre-war capacity. Although the quality of water streams improved, the problem B&H is facing now is how to preserve that quality.

### Table 2. Number of Stations/Profiles in B&H [2], [3]

<table>
<thead>
<tr>
<th>Area</th>
<th>Number of stations / profiles</th>
<th>Water quantity</th>
<th>Water quality</th>
</tr>
</thead>
<tbody>
<tr>
<td>F B&amp;H</td>
<td>40*</td>
<td>5**</td>
<td></td>
</tr>
<tr>
<td>RS</td>
<td>9</td>
<td>18</td>
<td></td>
</tr>
<tr>
<td>Total in B&amp;H</td>
<td>49</td>
<td>23</td>
<td></td>
</tr>
</tbody>
</table>

* including 5 short term operational stations
** including automatic tele-transmission data station that monitor 5 parameters of water quality

Federal Water Law does not clearly define the responsibilities for water quality monitoring. Since 1998, systematic monitoring (12 samplings annually) has been performed only at 4 profiles of Neretva river. Analyses do not include hydro-biological analyses, thus the general
assessment of examined stream category can not be identified. At the end of 1999, water authorities of RS re-established the measurements on a series of measurement profiles (3 samplings annually).

Generally, monitoring of B&H rivers' water quantity and quality is not satisfactory. Activities are not harmonized between F B&H and RS and data obtained is not transparent.

Figure 1. Monitoring stations before (1991) and after the war (1999) [4]
PROPOSALS FOR MONITORING NETWORK DESIGN

Taking as an example the systems used in almost all developed countries, project "Water Sector Institutional Strengthening - Water Quality Aspect" gave some general proposals on the future monitoring and assessment system, as well as the necessary guidelines on how to act and establish new monitoring network in order to meet the inter-entity and transboundary requirements.

Strong financial support from the international community will possibly enable establishment of a completely new and up-to-date monitoring network that shall be in accordance with the EC requirements. The EURO WATER NET SYSTEM program foresees setting up of about fifty monitoring stations for water quantity and quality control at the whole territory of B&H. Since most of the rivers flow through both entities the number of foreseen stations is probably underestimated. The monitoring stations should be located on both sides of entity line which would result in increased number of monitoring stations.

Considering the position of the country and present complicated on-field situation, the new network shall consist of:

- **Stations at the state border**
  - all the sites where the strong impacts are expected (practically upstream and downstream of all mouth of rivers flowing into border of Sava river),
  - the sites along the border of Drina river (downstream of particular hot spots),
  - border station at Neretva River (flowing to the Adriatic Sea).

- **Stations within the state**
  - all sites upstream and downstream of the entity line where significant sources of pollution are expected in order to escape the possible conflicts,
  - sites downstream of present hot spots to provide permanent control of possible impacts,
  - sites that were previously installed, and other specific sites where necessary.

Considering the state of current network, some rearrangement should be done in order to fulfil all recommendations, constraints and necessities.

MONITORING IN CETINA RIVER BASIN - CASE STUDY

Cetina catchment area is divided between Croatia and Bosnia and Herzegovina as indirect (2,440 km² in B&H) and direct river basin (1,200 km² in Croatia). Cetina river is one of the most important water streams in the middle, karst, and coastal area of Croatia. Its importance lies in the water richness and its natural beauty. It is used for purposes of water supply, agriculture, irrigation, fishery, industry, hydropower production, and recreation. The water is also the main ecological factor influencing the existence of number of valuable ecosystems and landscapes. Due to the karstic geological characteristics, groundwater in the catchment area is highly vulnerable to the pollution. Recently ending war in both countries caused migrations in the area and shift in the population pattern. It is estimated that, in 1999, total of 157,000 people were living in the catchment area, which is 17% less than in 1981 and 18% less than in 1991.

Intensive use of natural environment as a consequence of rapid urbanization and industrialization, lead toward a need for integrated coastal and river basin management. Considering that the Cetina river basin belongs partly to Croatia and partly to Bosnia and Herzegovina, only joint coordinated action would give real and applicable results.

Both Croatia and Bosnia and Herzegovina expressed their interest and intention to start the joint project related to protection and management of transboundary waters in the Cetina catchment area and to implement UN/ECE Guidelines on monitoring and assessment of transboundary ground waters. The joint monitoring program has been already prepared and its implementation is under way. The main objectives of the programs are:

(i) implementation of the action plan to resolve the transboundary concerns;
(ii) improvement of the monitoring program according to the UN/ECE Guidelines on
monitoring and assessment of groundwater in order to enable the acquisition of the needed information for transboundary groundwater management and the establishment of a water quality data base. The application of the UN/ECE Guidelines should guarantee harmonization of the monitoring and assessment activities;

(iii) establishment of a joint body for the protection and management of the transboundary catchment area of the Cetina river and its groundwater;

(iv) identification of gaps and incompleteness of the UN/ECE Guidelines and proposal of improvements, in particularly related to the karstic areas;

(v) identification of pollution hot spots in the river basin.

It is expected that this Program will:

(i) prevent the pollution of transboundary waters, and in particular the karstic groundwater body;

(ii) where applicable, improve the water quality;

(iii) provide more effective and balanced use of available surface and groundwater resources that will lead to the sustainable development of the region;

(iv) generate knowledge about the flow regime, monitoring, protection measures and assessment of transboundary karstic groundwater which can be used to improve and broaden the guidelines on monitoring and assessment of groundwater in karstic regions.

For this purpose, relevant bodies from both countries prepared a joint Action Plan that gives the list of priority actions necessary to implement this program successfully. According to this plan, the following should be done:

(i) Based on the existing data:

- identification of pollution sources in the immediate and nearby zone of Cetina River basin;
- identification of interaction between surface and ground water flow regime in the immediate and nearby catchment area;
- quantification of point and non-point pollution sources of transboundary character;
- balancing the water resources flow;
- quality assessment of surface and ground waters.

(ii) Based on the new field surveys

- keeping the public informed, stressing the importance of positive acting related to water and environment protection issues;
- identification of needs in immediate and nearby zone of catchment area, including identification of existing illegal and uncontrolled solid waste landfills, small uncontrolled sewerage systems, stockyards, agricultural lands, small manufacturers and other possible activities that may influence water quality in the catchment area;
- elaboration and harmonization of protocols for Standard Operational Procedures (SOPs);
- elaboration and harmonization of the Program for reporting the monitoring data;
- positioning of key monitoring stations for monitoring of transboundary pollution;
- preparation of the strategy for particular regions implementing the program according to the list of priorities in the framework of institutional set up;
- assessment of possible impacts on state of transboundary water quality based on the identified pollution sources;
- proposals for solving the conflict of interests (at local, sub-regional, national and international level);
- analysis of resources vulnerability based on the probability of occurrence or increased risk upon assessment of expected consequences.

Although the Program is already prepared, its implementation is waiting for financial resources. Since this Project is of vital importance for both countries, it is expected that the funds shall be provided very soon. Considering the past and current circumstances, it is presumed that the mentioned activities shall enable prospective co-operation at international level that was interrupted years ago, not just for the Cetina river basin, but for the Neretva and Una river basins as well.
CONCLUSIONS

Concerning all the facts, conditions and particular circumstances the following conclusions can be drawn:

- Monitoring and assessment of surface and ground water in B&H is very unsatisfactory and should be fully established and improved in the nearest future;
- The activities related to the monitoring and assessment should be intensified and harmonized with the current European practice;
- Harmonization of activities should be achieved through ongoing EU project "Preparation of Environmental Legislation";
- Harmonization and coordination of activities between entities should be designated and agreed at the state level;
- More attention should be paid to cooperation with neighboring countries. The pilot project for Cetina River should be taken as a conceptual example for how the harmonization could be enhanced and proficiency achieved.

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POLLUTION IN KARST AQUIFERS

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Hydrological conditions in karstic springs are very convenient for the drinking water supply. However they are highly sensitive to the pollution within their karstic catchment area. The estimation of the catchment area is very demanding and expensive task. In the frame of described project we have tried to find out which time period and frequency of sampling for physical and chemical analyses would meet the satisfactory statistical confidence level. The results had shown that for the reliable estimation of water quality three-year observation period with monthly sampling is advisable for complex karstic systems. For the hydrogeological parameters we would recommend one year weekly samples to get data reliable enough for planning the tracer tests.

The final scope of the project was the verifying and supplementing of previous estimated vulnerability and resumed appointing of the protected areas in the investigated karst areas.

INTRODUCTION

According to the geological investigations about 46 % of Slovenia are Karst areas. Regarding a very convenient hydrological conditions many karstic springs are used for the drinking water supply. On the other hand the karstic springs are highly exposed to the pollution within their Karst catchment areas. The largest problem of protecting the springs against pollution from its catchment area is its significant changes under different hydrological conditions. Therefore the research of the catchment areas of this springs and pollution transport through the underground water connections is of the highest priority in the National Environmental Protection Program. This was the reason why the international project Pollution transport in Karst regions - Tracer Experiments and Models in Various Aquifers, Investigation in Slovenia, was carried out. The region of Trnovo Plateau was chosen as a model. At the foot of the plateau four karstic springs are used for drinking water supply for more than 1 million people. As the springs are the only abundant sources of drinking water in this part of Slovenia, their karst background must be protected against pollution. Due to hydrographical complexity within this karst aquifer, it has been not possible earlier to define the extent, size and capacity as well as treat to each karst spring separately. For the same reason, the protection measures were not introduced separately but for the area as a whole. Previous hydrological and geological investigations indicated the main drainage directions of karst water flow, but a series of unsolved hydrological questions remained. Solving the open questions would enable better exploitation and more reasonable protection of water resources in the area. Geological, hydrogeological, speleological, hydrological, isotopic, physical, chemical and biological properties of the springs have been studied during years 1993-1996 (Kranjc, 1997).

INVESTIGATION PROGRAM AND RESULTS

The research high karst area Trnovsko Banjški Plateau (TBP) is located in the western part of Slovenia. It is a relatively well-confined mountainous area bounded by lower non karstic areas from almost all the parts. Karst springs on the border of high karst are captured for water supply. Central karst plateau reach altitudes from 600-1500 m a.s.l., while the springs from 50-300 m a.s.l. TBP is about 50 km long, 10-50 km wide and extend over 700 km². The rainfall from this entire area sinks into deep karst aquifer feeding abundant karst springs located along its foot. Within the high karst area the underground watershed between Adriatic and Black Sea was determined (Habiè, 1989). From a climatic point of view the high karst of TBP is a typical transitional area between the Mediterranean climatic influences of Adriatic Sea and alpine climatic region. The central part of TBP already has the real alpine climate with abundant snow during rather cold winters. The precipitation is abundant all year around and the central part of Trnovski gozd receives more than 3000 mm yearly. The total discharge of the karst springs in dry periods reaches more than 2 m³/s (7% of Slovenian karst underground water). During high water the main karst springs (Mrzlek, Lijak, Hubelj, Vipava, Podroteja, Divje jezero, Kajža and Hoteški) drain about 280 m³/s of water (Janež, 1997).
The investigation program contained the following topics:
- Natural background
- Water balance
- Physical, chemical and biological investigation
- Isotopic investigation
- Four tracing experiments at different locations and different hydrological conditions
- Verifying and supplementing of previous estimated vulnerability and resumed appointing of
  the protected areas in the investigated karst areas.

The investigation program was very extensive and expensive. The data collected by different
European institutes were presented and interpreted detailed in the common report (Kranjc,
1997).

After finishing the project we continued the data processing to look for additional information
from hydrogeological data that were collected during the project. The data were collected in
three different levels:
- Regular long-term state water quality monitoring program: samples were taken at five main
  springs, four times per year and the extensive chemical, bacteriological and biological
  analyses were performed.
- Three years monthly samples were taken in 12 springs and hydrogeological parameters
  were analysed (macro cations and anions).
- Weekly samples were taken in the Hubelj and the Vipava.
- Six months the hydrogeological parameters were analysed in daily samples of the Hubelj,
  the Vipava and Mrzlek.

The sampling, physical and chemical analyses were performed in conformance with the
methodology recommended by international standards. The results of the long-term national
water quality monitoring program have been a warning that there is a need to take measures to
protect the water quality in springs (Kranjc, 1997). The spring water quality was classified as
relatively good. The chemical parameters in most of the water samples did not exceed the
standards for drinking water. On the other hand, toxic substances, heavy metals, and organic
compounds were present in some of the analysed water samples and in most of the sediment
samples. Relatively high concentrations of mercury, cadmium (Figure 1), lead and copper were
measured in some of sediment samples, which means that the pollution was nevertheless
present in the investigated springs. Likewise the phenolic compounds and polycyclic aromatic
hydrocarbons (PAH) were very often present in water samples of some springs (Figure 2). The
number of present PAHs determined in GC/MS screening was high in water as in sediment
sample extracts. In all investigated springs numerous compounds originating from different
human activities were determined in GS/MS screening of water and sediment extracts as well.

For the establishment of statistical processing and seasonal changes assessment we used
AARDWARK (WRc 1995) seasonal model. We compared the results obtained for springs Hubelj
and Vipava, which were included to all sampling programs during the project. In the Hubelj
spring the parameters characterizing the geological origin showed seasonal changes. The
conclusion could be that the hydrological conditions in the hinterland of the Hubelj are not
varying much. In the Vipava spring only conductivity, the sum Ca+Mg and bicarbonate
disclosed seasonal changes, while the concentrations of Ca and Mg are changing very
unregularly. We would presume the catchment area of the Vipava is changing at different

![Figure 1: Heavy metals in sediment of karstic springs](image)
hydrological conditions. Very similar behaviour was established by the comparison of the ratio Ca/Mg with the flow. In the Hubelj the ratio is very constant while in the Vipava much more variable (Figure 3). However we observed the diminution of the Mg concentration in the Hubelj up to the discharge about 5 m3/s while in the Vipava this interdependence did not occur (Figure 4).

Finally we tried to find out which time period and frequency of sampling would meet the satisfactory statistical confidence level. The calculation of statistical characteristics showed that the one-year weekly measurement gave us almost the same result as three years long weekly sampling. Similar conclusions came from the comparison of the year 1994 which was according to hydrological characteristics a dry year and the year 1995 which was according to hydrological characteristics an average year (Figure 5).

The data collected during half a year from daily samples gave quite similar statistical values as the data series of one and three year weekly samples. Slightly less reliable are the data collected in monthly intervals. We assume that there is no good comparability of the hydrological data in time. However, the monthly data are still applicable as the basis for tracing experiment planning.

Data of polluters and chemicals used in the catchment areas have been collected as well. Tracing experiments under different hydrological conditions to estimate the underground water connections between catchment areas and springs have been done as well. The extensive research in the mentioned karstic springs has shown a reliable link-up between the irregular waste disposals and untreated waste water discharges on the plateau and the pollution in the springs. On the basis of the research results the attainable rehabilitation programs and methods will be defined.
CONCLUSIONS

The hydrochemical analyses of weekly samples in the period of three years and the analyses of daily samples taken during six-month period showed very similar characteristics. One year weekly sampling is informative enough to foresee the similarities in spring water chemical behaviour as well. The results of one-year monthly samples are not informative enough, while the three year sampling would give us approximate information, which still enabled us to plan the tracing experiment nearly reliable. On the basis of the performed investigation one year weekly sampling was determined as optimal solution, which is informative enough at the still reasonable cost.

The extensive research in the mentioned karstic springs has shown very reliable link-up between the irregular waste disposals and untreated waste water discharges on the plateau and the pollution in the springs. On the basis of the research results the attainable rehabilitation programs and methods will be defined.

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GROUNDWATER QUALITY MONITORING TAILORED FOR POLLUTION RISK ASSESSMENT IN BARBADOS

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The small but densely populated island of Barbados is almost entirely dependent on groundwater for its public water supplies. Groundwater is drawn from a karstic, coralline limestone aquifer, which is highly vulnerable to pollution. Pollution risk assessments of two catchments which provide 87% of the total public supply identified agricultural and industrial activities and rapid urbanisation as potential sources of pollution. To support these assessments, groundwater quality monitoring was established, to see whether the identified risks were actually causing deterioration in groundwater quality. A targeted set of indicator parameters was developed to allow the possible impact of agriculture, industry and urbanisation to be evaluated, but taking account of analytical capabilities and costs. The network of sampling points and frequency of sampling also needed to balance technical requirements and available resources. Regular monitoring, established at the beginning of the study in 1987 has, as a result of this tailor-made approach, been sustained ever since. The results demonstrate that groundwater abstraction has been managed to avoid saline intrusion. There are indications of modest impacts of agriculture and urbanisation on groundwater quality from elevated nitrate concentrations, but little evidence that quality is deteriorating overall.

INTRODUCTION

Establishment of groundwater quality monitoring programmes often involves the manager or designer in difficult choices about where, when and how to sample, and what to monitor for. The design process inevitably involves compromises between what is technically desirable to meet the overall objectives of monitoring, and what is feasible in terms of human and financial resources. Given that financial constraints almost invariably mean less effort than really necessary can be devoted to monitoring, it is essential that no steps in the monitoring cycle (UN ECE, 1996) are left out. While the steps may each require widely differing effort, none can be completely ignored or the whole process fails.

The present paper describes the development of a monitoring programme to support the assessment of groundwater pollution risk in Barbados. The objectives of the study were to carry out detailed surveys of all activities which could present a pollution threat to groundwater and to establish a groundwater quality monitoring and assessment programme. Two catchments in the south of the island, Belle and Hampton, were chosen for the study. Together, they provide about 87% of the island's public water supply, including most of that for the Bridgetown commercial areas and the hotel and tourism development on the south coast. Protection of groundwater quality in these catchments is thus vital to the economy of the island.

HYDROGEOLOGICAL SETTING

Barbados is a small island of about 430 km² largely composed of coral limestone. This covers 86% of the land area and provides a highly permeable and productive aquifer up to 100 m thick (Figure 1), underlain by mudstones, shales and clays which make up the remaining 14% of the island's surface. The permeable and karstic nature of the limestone is indicated by the almost total absence of surface runoff and by the spectacular tourist attraction of the caverns at Harrison's Cave. As is typical for such coral limestone islands in the Caribbean (Foster & Chilton, 1993), groundwater occurs as a freshwater lens in the south and west of the island; the "sheetwater" zone. There is also a thin zone of "streamwater" at the junction between the permeable coral limestone and the underlying impermeable marls in the interior (Figure 1). The freshwater lens is 20 m thick at Belle pumping station but thins to 3 m near the south and west coasts. It is, therefore, as in many similar situations in the region, a fragile resource, vulnerable to pollution from human activities at the land surface and susceptible to saline intrusion if groundwater abstraction is not undertaken with care.
LAND USE PATTERNS AND DEVELOPMENT CONTROLS

Barbados has a long history of sugarcane cultivation on large plantations which, apart from some diversification to horticulture during 1939-45, continued into the 1970s. Since then there has been a move away from sugarcane through a policy of agricultural diversification, encouraged in the last few years by low sugar prices and high wage costs. The replacement horticultural crops have a wide variety of pests, and have required the use of an increasing quantity and rang of pesticides. The population of the island is about 250,000, giving an overall average density of about 580/km². The growth rate is very low (0.2% per year) but housing, industrial and commercial development have been rapid. Part of central Bridgetown has sewered sanitation, and a recently-completed scheme will soon serve part of the densely populated south coast area. Most of the interior of the island, however, depends on unsewered sanitation, with discharge of both domestic and industrial wastewater directly into the limestone aquifer.

The possible impact of such disposal on groundwater quality has been of concern for many years. To protect the bacteriological and chemical quality of groundwater used for public supply, the Government established a policy of Development Control Zones around existing and proposed public supply sources in 1963. Five Zones were established, based on a simplified concept of pollutant travel time through the aquifer. Thus, a travel time of 300 days for Zone 1 was selected to be significantly greater than the subsurface survival time of enteric bacteria, and a 600 day travel time was selected for Zone 2. Further details of the characteristics and distribution of the zoning and the controls imposed within them are given in Chilton et al (1990). This was an important and far-sighted piece of legislation, and represents one of the earliest examples of a groundwater protection policy. There had, however, been little or no groundwater quality monitoring since its introduction from which the effectiveness of the zoning could be assessed. The initiation in 1987 of the risk assessment and the associated monitoring described in this paper presented the first opportunity for that to be done.

POLLUTION RISK ASSESSMENT

Groundwater pollution risk can be considered as the interaction between the natural vulnerability of the aquifer and the pollutant loading which is imposed as a result of human
activity (Foster, 1987). The pollution loading can be controlled or modified but the vulnerability of an aquifer depends on its intrinsic properties. The use or value of the groundwater then determines whether any pollution risks identified will result in serious threat to groundwater supplies.

An assessment of aquifer vulnerability requires knowledge of those characteristics which determine the likelihood of being adversely affected by an imposed pollutant load. These characteristics are:

- The thickness and nature of the strata making up the unsaturated zone
- The attenuation capacity of the soil and unsaturated zone, as a result of physico-chemical retention, chemical reactions and biochemical degradation (Foster, 1998).

Methods have been developed (Foster, 1998) for quantifying or indexing and mapping aquifer vulnerability, based on combinations of the characteristics outlined above. Coral limestone aquifers are highly permeable, often containing fissures, and usually have a cover of thin, permeable soils with relatively low clay content and little organic matter. These two factors combine to produce very high vulnerability, which is only partly offset in Barbados by the relatively deep (30 to 70 m) water tables. The whole aquifer was, therefore, considered to be vulnerable.

The second element of pollution risk assessment, the applied loading has two main characteristics:

- the manner of pollutant disposition, especially in relation to whether it by-passes the soil and the unsaturated zone, and the magnitude of the associated hydraulic loading
- the physico-chemical mobility and persistence of the pollutants involved (Foster, 1987).

In the context of the risk assessment for Barbados, efforts to quantify the pollutant loading required knowledge of the types of activities which could cause groundwater pollution, and determination of their magnitude and distribution. Thus, three categories of such activities were identified from the outset: 1) intensification of agriculture, 2) effluent disposal from unsewered sanitation and 3) uncontrolled discharge of industrial effluents. For domestic sanitation, adequate information on population distribution and densities was already available, but detailed surveys were required of agricultural and industrial activities. The approach developed and the results of the risk assessment for the Belle catchment are summarised in Chilton et al (1990).

ESTABLISHMENT OF MONITORING

Primary considerations in the design of the monitoring programme were that it should be:

- adequate to meet the objectives of supporting the pollution risk assessment and providing data from which the effectiveness of the development control zoning could be evaluated,
- within the capability of the Environmental Engineering Division (EED) and therefore likely to be sustained.

With a small staff complement, many existing routine environmental health tasks and the pollution risk surveys to be undertaken, only a limited commitment could be made to regular groundwater quality monitoring. While the project partners could, and did, provide support during the lifetime of the study, it was agreed at the outset that the monitoring programme should be kept to a scale that was likely to be sustained after the study. The number of sampling points, frequency of sampling and analytical parameters were determined on this basis, as described below.

Selection of Sampling Sites

In most groundwater monitoring programmes, overall cost considerations dictate that sampling must be undertaken from existing wells or boreholes, as construction of new, purpose-built sampling points is very expensive. The present study was no exception, and the groundwater quality component commenced with inventories of wells in the Belle and Hampton catchments
using a prepared survey form (Chilton, 1991). In total, 32 wells were surveyed in the Belle catchment and 36 in Hampton. Wells are generally 2-3 m in diameter and 40-80 m deep, unlined, and equipped with electric submersible pumps for public supply use or irrigation. Ten to 12 wells were chosen in each catchment for monthly sampling from October 1987, selection being based on:

- good geographical coverage of the two catchments
- distribution representative of the different control zones
- adequately covered and protected from direct rainfall and storm runoff
- in regular use with a pump of reasonable discharge
- access (tap) for sampling from the rising main close to the well head
- possibility of telephoning in advance to ensure the well was operating prior to sampling

In mid 1988, following review of the early monitoring results, the number of wells was reduced and the frequency increased to once every two weeks. The purpose of this was to improve the scope for assessing any seasonal variations in groundwater quality caused by the seasonality of rainfall and perhaps in agricultural activities, but without increasing the overall workload in either field sample collection or laboratory analysis. This was considered more important at the time than maintaining the higher density of sampling points. The sampling sites are shown in Figure 2.

Selection of Parameters

In all groundwater quality monitoring programmes, the choice of parameters is governed by the objectives of the monitoring, the use or uses to which the water is put, the anticipated water quality issues arising from either natural hydrochemical variations or human impacts, local groundwater quality standards and, not least, by cost considerations (Chilton and Foster, 1997). At the outset of the risk assessment, intensification of agriculture, discharge of industrial effluents and unsewered sanitation were identified as possible sources of pollution, and indicator parameters were required for these, as shown in Table 1. While some, particularly chloride and nitrate, appear more than once in the table and are individually non-specific, it is often associations or combinations of (high or low) parameter values which can provide indications of the source of pollution.

A further consideration was that the selected parameters should as far as possible be readily and economically analysed within Barbados. In order that the monitoring programme could be established quickly and easily, HACH™ field kits were provided from external project funding for nitrate, chloride, sulphate, ammonia and phosphate, and a Del Agua field kit for faecal coliforms, pH and EC (electrical conductivity). Monitoring was initiated using these kits, but the analytical work was soon taken over by the Barbados Government Laboratory. Early in the programme, comparisons were made between lab and field and between the Barbados and

![Figure 2 Location of Monitoring Wells in the Belle and Hampton Catchments](image)

Figure 2 Location of Monitoring Wells in the Belle and Hampton Catchments
British Geological Survey (BGS) laboratories. Samples for TOC (total organic carbon) and solvents were taken to the BGS and commercial laboratories in the UK respectively, as they could not be analysed on the island.

Selection from among the wide range of pesticides in use on the island was made on the basis of the agricultural survey results. For the most heavily used compounds, preliminary estimates were made of the likelihood of leaching into the subsurface and the risk this might present of groundwater pollution, from consideration of their physical and chemical properties – solubility, mobility and persistence – and their toxicity (Chilton et al, 1990). Thus, atrazine, ametryne, dimethoate and diazinon were selected for monitoring, based on their susceptibility to leaching, known occurrence in groundwater elsewhere in the world, overall usage and ease of analysis within Barbados.

<table>
<thead>
<tr>
<th>Natural Water Quality Variations and Saline Intrusion</th>
<th>Potential Pollution Impacts</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Agricultural Practices</strong></td>
<td><strong>Urbanisation</strong></td>
</tr>
<tr>
<td>EC</td>
<td>Nitrate</td>
</tr>
<tr>
<td>Chloride</td>
<td>Phosphate</td>
</tr>
<tr>
<td>pH</td>
<td>Sulphate</td>
</tr>
<tr>
<td>Chloride</td>
<td></td>
</tr>
<tr>
<td>Potassium</td>
<td>Faecal Coliforms</td>
</tr>
</tbody>
</table>

Table 1. Parameters selected for groundwater quality monitoring in Barbados

RESULTS OF MONITORING

The analytical results are compiled into suitable spreadsheets. Guidance on water quality assessments (Chapman, 1996) and monitoring (UN ECE, 1996) emphasises the need to convert data into information that can be used for water management, closing the last link of the monitoring cycle (UN ECE, 1996). For its present purpose and length limitation, the paper illustrates how this can be done, rather than providing detailed interpretation of the groundwater quality. A longer version of the paper to do this will be prepared for journal publication.

Natural Quality and Saline Intrusion

The duplicate samples brought to the UK for full chemical analysis at the beginning of the study provide an indication of baseline hydrochemical conditions in the aquifer. These are typical for a coral limestone, with pH slightly above neutral (7.2-7.6) and calcium (60-90 mg/l) and bicarbonate (140-250 mg/l) being the dominant dissolved constituents. Natural groundwater quality is good, with low overall mineralisation (EC 375-750 mS/cm), which gradually increases from the centre of the island towards the coast, but is suitable for domestic, agricultural and industrial uses.

A primary consideration in the development of groundwater in Barbados has been to avoid deterioration of chemical quality due to saline intrusion. A small amount of sampling of public supply wells was undertaken as part of a major water resources study in 1977. This indicated consistently low EC and chloride concentrations, with little variation during the 8-month sampling period, and the picture at that time was one of generally stable groundwater quality. The much more comprehensive monitoring initiated in the present study confirms this. There is a general increase in chloride concentration from 30-50 mg/l in the streamwater zone to 60-120 mg/l in the sheetwater zone (Chilton et al, 1990). Concentrations have remained remarkably constant, indicating that overall groundwater quality has not deteriorated due to saline intrusion. Figure 3 shows an example of chloride concentrations from the start of monitoring in 1987 to the end of 1999 for the Hampton supply well, together with mean monthly rainfall for the Hampton catchment. While there is some evidence that chloride concentrations may become slightly elevated during dry periods or years (eg 1995) and conversely decrease in response to heavy rainfall and recharge, the converse is apparently seen
in 1997-1998. This may be a response to changes in abstraction regime at the well, pointing to the need for additional hydrogeological data (Chapman, 1996) to assist in the interpretation. However, the overall picture, confirmed by the results from both catchments, remains one of generally stable conditions with respect to salinity.

Agricultural Impact

Nitrate and pesticides provide the best indications of the potential impact on groundwater quality of agricultural activities. For most wells, current groundwater nitrate concentrations are between 5 and 10 mg NO₃-N/l, compared to the WHO Drinking Water Guideline value of 11.3 mg NO₃-N/l, which is equivalent to 50 mg NO₃/l. There are no consistently low nitrate concentrations, of say 1mg NO₃-N/l, which would suggest pristine conditions, and the impact of human activities is felt throughout both catchments. In the dominantly agricultural areas of the upper part of the Belle catchment and the Hampton catchment, concentrations are consistent with nitrate leaching losses of 30-50 kg N/ha from the 130 N kg/ha applied annually to sugarcane (Chilton et al, 1990).

The long monitoring record which is now available assists in the consideration of both seasonal variations and longer term trends. From the time series data, there is evidence from some wells that nitrate concentrations are higher in the dry season and lower in the rainy season, perhaps in response to dilution in the larger volumes of infiltration, although perusal of all of the time series indicates this is by no means universal. As an example, Figure 3 shows nitrate concentrations and monthly rainfall for well 18 at National Hatcheries in the Hampton catchment (Figure 2) showing lower nitrate in the 1990 and 1995 rainy seasons, but apparently the opposite in 1988 and 1994, and no response at all in 1992. While the thick unsaturated zone referred to above might imply long infiltration times, in practice the highly fractured karstic limestone responds rapidly to the influence of rainfall. This well is one of the few which suggests an increase in nitrate over the period of record, but most wells show no overall trend of rising nitrate concentrations with time.

Plotting annual average concentrations provides a useful means of summarising large volumes of data and comparing trends at a number of wells. Figure 4 presents nitrate data for the Belle catchment in this way, confirming the generally stable groundwater quality situation, with little evidence that the situation is becoming worse with regard to nitrate leaching from agriculture. Indeed, there are suggestions from some of the wells in the agricultural part of the catchment (13 and 14, Figure 2) of a slight overall decline. Data from the Hampton catchment (not shown) presented in a similar way support this general picture of stable quality, but also confirm the slight suggestion from Figure 3 that concentrations are increasing at well 18, National Hatcheries.

Urban Impact

Nitrate in groundwater may have more than one origin, and indeed is often considered to be a useful indicator of the impact of urbanisation (Table 1), either in the form of leaking sewers or infiltration from unsewered sanitation. Thus, although almost all domestic premises use pit latrines or soakaway pits in both the Belle and Hampton catchments, the population density in the southern urbanised part of the former is some 30 persons/ha, compared to 4 persons/ha in the rural areas. Given an estimated per capita wastewater generation of about 250 l/person/d and an assumed annual per capita load of 5 kg from nitrogen excretion, nitrate concentrations in groundwater recharge in the more densely-populated urban areas could reach 25-30 mg NO₃-N/l (Chilton et al, 1990). This assumes that all of the nitrogen deposited in this way is oxidised and reaches the water table. The equivalent maximum estimate for the rural areas, again assuming all of the nitrogen were mobilised, would be some 5 mg NO₃-N/l. The results of the water quality monitoring appear to support this. Moderate nitrate concentrations in rural areas (wells 13, 26 and 14, Figure 4) reflect inputs from agriculture and low-density rural housing, while higher concentrations (well 1) reaching the WHO guideline value are mostly restricted to the more densely populated, southern, urban part of the Belle catchment. Indeed well 27 (Figure 2), has an average nitrate concentration of 24.7 mg NO₃-N/l in 1988 (Chilton et al, 1990) and, as a consequence, was taken out of use and monitoring had to be discontinued. Well 20 in Figure 4 has the lowest nitrate concentration, but is also the only well in either catchment with significant ammonia concentrations. This may be an indication of less oxidising (ie more reducing) conditions, associated with effluent disposal from the nearby industrial area (Figure 2), particularly the large numbers of livestock in a nearby dairy complex.
Figure 3 Examples of Chloride and Nitrate Time Series, Hampton Catchment
The results of bacteriological analyses for selected monitoring wells are summarised in Table 2. All of the public supply sources (Figure 2) are equipped with in-well chlorination and it is not, therefore, possible to collect groundwater samples at these wells before chlorination. While clearly desirable from a public health point of view, it does preclude easy monitoring of the untreated water, which would be the most effective way of seeing whether the development control zones are preventing or reducing bacteriological pollution. Most of the irrigation wells in Table 2 have regular positives FC detections and high counts, confirming the highly vulnerable nature of the karstic limestone aquifer and the need for effective protection of the public supplies.

Table 2 Summary of Bacteriological Results

<table>
<thead>
<tr>
<th>Location</th>
<th>Sampling Period</th>
<th>Number of Samples</th>
<th>Positives</th>
<th>Faecal Coliform Counts per 100 ml</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>No</td>
<td>%</td>
</tr>
<tr>
<td>Belle</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Govt House (1)</td>
<td>1988-94</td>
<td>79</td>
<td>51</td>
<td>65</td>
</tr>
<tr>
<td>Pine Central (20)</td>
<td>1987-99</td>
<td>90</td>
<td>70</td>
<td>78</td>
</tr>
<tr>
<td>Mount Wilton (28)</td>
<td>1987-91</td>
<td>44</td>
<td>25</td>
<td>57</td>
</tr>
<tr>
<td>Hampton</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Brighton (33)</td>
<td>1990-99</td>
<td>104</td>
<td>46</td>
<td>44</td>
</tr>
<tr>
<td>Edgecumbe (5)</td>
<td>1987-99</td>
<td>147</td>
<td>75</td>
<td>51</td>
</tr>
<tr>
<td>Kendall (15)</td>
<td>1987-99</td>
<td>134</td>
<td>69</td>
<td>51</td>
</tr>
<tr>
<td>Corbins (8)</td>
<td>1987-99</td>
<td>108</td>
<td>66</td>
<td>61</td>
</tr>
<tr>
<td>Nat Hatcheries (18)</td>
<td>1987-99</td>
<td>135</td>
<td>98</td>
<td>72</td>
</tr>
<tr>
<td>Packers (21)</td>
<td>1987-99</td>
<td>136</td>
<td>66</td>
<td>49</td>
</tr>
</tbody>
</table>

\(\text{tntc} = \text{too numerous to count}\)

Industrial Impacts

Barbados is not an industrial community, and while the survey revealed a large number of very small industrial and commercial premises, very few were engaged in activities which would produce significant volumes of polluting effluents. The small number of analyses of dissolved organic carbon early in the study were generally relatively low (0-2 mg/l), and isolated higher concentrations which could not be replicated in repeat and duplicate samples were eventually
ascribed to sample contamination (Chilton, 1991). Analyses of a small number of industrial organic compounds were all below detection limits.

CONCLUSIONS

A groundwater quality monitoring programme was established in Barbados in 1987 to assist in evaluating the effectiveness of the development control zone policy and to see whether risks identified in the pollution assessment were actually causing deterioration in groundwater quality. A limited and targeted, in fact tailor-made, selection of indicator parameters was used to allow the impacts of urbanisation, agriculture and industry to be assessed. The network of sampling points in the two catchments and the frequency of sampling were also tailored to the available staff resources for sampling and analysis. That the programme design was indeed sustainable and within the capabilities of EED is demonstrated by its continuation after the completion of the study to the present time, and a valuable body of data has been accumulated.

The results show that the natural quality of groundwater in the Coral Rock aquifer is extremely good and suitable for all domestic, agricultural and industrial uses. Groundwater abstraction from the restricted and fragile resource of the freshwater lens has been managed effectively to minimise the impact of saline intrusion. There are clear indications of the ubiquitous but modest impact of agricultural activities, as groundwater nitrate concentrations in agricultural areas are in the range 5-8 mg NO₃-N/l. The albeit limited and inconsistent evidence of seasonal responses of chloride and nitrate to rainfall variations are considered to have justified the increase in sampling frequency to twice per month. There are slight decreasing or increasing trends at individual wells, but little evidence of an overall increase in nitrate concentrations. Somewhat higher nitrate concentrations, which reach or just exceed the WHO Guideline value of 11.3 mg NO₃-N/l, reflect the additional impact of unsewered sanitation in the more densely populated urban areas. These are often, but not exclusively, associated with high levels of bacteriological contamination. The monitoring results indicate that the long-established system of development control zones has served Barbados well in protecting its valuable groundwater resources.

ACKNOWLEDGEMENTS

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REFERENCES


ROBUST STANDARD ERROR ESTIMATORS FOR TREND DETECTION

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In 1990 the North Sea riparian countries and the countries along the river Rhine agreed to reduce the input of loads to the North Sea for some 30 to 40 pollutants. Data about concentrations of substances are gathered for water management and policy evaluation of the North Sea. A protocol, called Trend-y-tector, for the detection of trends in yearly loads has been developed and offers a structured methodology for trend analysis. The significance of the trend is determined by the ratio of the size of the trend and its standard error. This latter is here based on the residuals of the trendline. In general the residual distribution has thicker tails and outlying values compared to the normal distribution. In this paper several estimators for the standard deviation are investigated and compared in order to improve the performance of the tests on trend detection. The estimator $S_{MAD}$ is found to be most suited for estimating the standard error.

INTRODUCTION

The OSPAR, the Oslo and Paris Convention for the Protection of the Marine Environment of the North East Atlantic, is one of the organisations that define policy measures. One of the objectives of OSPAR is to reduce input of contaminants and reach concentrations close to zero in the sea. From the beginning of the 1980s concentrations of pollutants in organisms and later in sediments were being monitored in a co-operative way. This means that monitoring methods, quality assurance and assessment criteria are developed on a common basis. One of the assessment tools is the assessment of trends. Monitoring and assessment aim on producing data to be used in Quality Status Reports of the marine environment. In the course of time, many time series of contaminants have been collected which are available for trend analysis.

In general, the main purpose of pollutant monitoring is to evaluate policy measures. When concentrations of natural or man-made substances are detected in too high concentrations in the environment, policy measures to reduce concentrations will need to be taken. During the policy life cycle, different kind of monitoring may be needed to produce the necessary information. In the first stage, it is describing the state of the system, followed by the compliance of concentrations with assessment criteria. Once, measures are taken, the temporal trend monitoring starts with the objective to detect trends in concentrations. In a final stage when concentrations are satisfactorily low, a kind of extensive compliance monitoring can be started, to assure policy and management that concentrations do not rise above assessment criteria again.

The methods chosen within the OSPAR context so far, are the subject for this article. First, the starting points and the framework of the trend assessment will be given, then one of the problems within this is described with the approach to tackle the problem, and finally the results of a research study are presented.

FRAMEWORK TREND-Y-TECTOR

The starting point for the methodology was a workshop containing members of the OSPAR working group responsible for collecting information on the loads of contaminants being discharged into the seas plus members of the ICES working group on statistical aspects of marine monitoring (ICES, 1997). Here, a trend was defined as the tendency of the concentration of a substance to go down or up over a period of years. In general, one is interested in trend over about ten years. From this point on there were some points of discussion, amongst others the choice between yearly and monthly values, the need for adjusting data for natural variability (e.g. flow in the case of riverine input) and whether a strict protocol can be satisfying for different situations.
It was defined that the statistical method used to assess trends should be (OSPAR, 2000):

- robust, i.e., to be both routinely applicable to many data sets, and to be as insensitive as possible to adverse numerical features such as extreme data values, partial bulking of samples, and less-than values;
- intuitive, i.e., the results of the analysis should be understandable without a detailed understanding of statistical theory;
- revealing, i.e., to provide easy access to several layers of information about the major features of the data, such as evidence of simple trends, extreme values, missing values, etc.

Following these general objectives, and based on lengthy experience with assessments of trends four separate but complementary components for a trend assessment were identified:

1) graphical presentation of the time series with, for example, a summary line to indicate the general trend and tolerance lines to reveal potential data anomalies (e.g. outliers and other suspected data);
2) a formal test of trend, with trend defined in an appropriate way for the context of the assessment, and possibly with a power curve which reflects the detectability of the given trend;
3) a measure of the tendency to increase or decrease;
4) a comparison of the current level against some reference level or a level in a previous year.

Some of components 2), 3), and 4), but especially 3) and 4), might then be combined in a statistical meta-analysis to provide summaries across regions, or across contaminants, etc. This framework implies the choice for yearly instead of monthly values, because marine monitoring often has a yearly sampling frequency or because input data are collected on a yearly basis.

Further, every time series has, from a statistical point of view, its own characteristics: the sampling frequency can differ from stratified on one time in a year to regularly every week in the year; the frequency can change throughout the period of interest; and the method of analysis can change which results in more or less censored values or even in a systematic shift in concentration level. The statistical handling of these time series is much more complicated than the straightforward analysis of independent assumed yearly data. For the purpose of OSPAR assessments, such in depth analyses are not appropriate, although they should be stimulated in other scientific forums. The yearly indices (e.g. median, annual input) are based on underlying raw data, and it is assumed that correct processing has taken place, for example, that the indices are adjusted yearly loads, or the logarithms of median concentrations measured in fish.

In an evolving way of feedback between the users of the methods within OSPAR, the statistical experts within ICES and the lead country for developing the method (The Netherlands) the described framework for trend assessment has become available as an Internet application which is called Trend-y-tector. The word has to be pronounced more or less as ‘trend detector’ and the ‘y’ stands for year pointing out the application is designed for assessment of yearly values. Further details can be found in (OSPAR, 2000) and on the Internet-site of the Trend-y-tector www.waterland.net/rikz/osparwg.

PROBLEM DESCRIPTION

The core of trend analysis consists of two different aspects, being trend detection and trend estimation, in both the estimation of the standard error is essential. The significance of the trend is determined by the ratio of the size of the trend and the standard error. This latter is here based on the residuals of the trendline. In general is assumed that the residuals are identically, independent observations following a Normal distribution, but this assumption often does not hold.

For water quality data, the residual distribution in general has thicker tails and more outlying values compared to the Normal distribution. Therefore this estimator will result in a bias with regard to the real standard error, and thus influencing the type I and II errors on the test on trend detection.

The idea behind this study is that a better, in statistical terms unbiased and effective, estimation of the standard error would lead to smaller detectable trends. This would make the test as described later more powerful.

The trend test is based upon a smoother (LOESS) which is used to capture the general trend.
Since there is an interest in revealing outliers, the trend line and the prediction limits should be derived from a robust smoother and a robust estimator of the standard deviation. For the smoother, a trend test is developed which has high power to detect monotonic and non-monotonic trends under Normality (Fryer and Nicholson, 1999). The disadvantage is high susceptibility to extreme values at the start and the end of the times series. The test statistic of the slope of the smoother against the residuals after smoothing is defined as:

\[ t = \frac{[z(n) - z(l)]}{s \sqrt{\left( (SS')_{11} + (SS')_{nn} \right)^{1/2}}} \]

- \( z(i) \) the \( i \)th value of the smoother
- \( S \) the smoother matrix as defined in Fryer and Nicholson (1999), and \( S' \) is its transpose.
- \( (SS')_{ij} \) the elements of the diagonal of the product matrix of \( S \) and \( S' \). These elements are constants. \((SS')_{11} = (SS')_{nn} = 0.5154\). Thus \([ (SS')_{11} + (SS')_{nn} ]^{1/2} = 1.0153\)
- \( s \) is the standard deviation of the residuals after smoothing.
- \( n \) is the number of yearly indices

In the current protocol this statistic is compared with the quantiles of the t-distribution with degrees of freedom equal to \( n - \text{tr}(2S-SS') \), with tr stands for trace of the matrix.

The output from this test will usually consist of the probability that the yearly averaged concentration could have arisen by chance when there is no trend. If this is less than some prespecified value (e.g. 5% or 10%), the result is considered to be significant, i.e., the null hypothesis of no trend is rejected.

A part of the study for alternative methods for estimating the standard deviation is a literature study for estimators and a statistical definition of the problem. The study contains a theoretical approach to describe the behaviour of the estimators and a simulation study in which the estimators are compared using distributions that often occur in water quality data.

**ESTIMATORS**

Five alternative estimators for the standard deviation are considered:

1. The traditional sample standard deviation \( S \);
2. The estimator for the standard deviation based on L-moments (Hosking, 1990);
3. The estimator for the standard deviation based on the Median Absolute Deviation (MAD) (Rousseeuw and Croux, 1993);
4. The estimator Sn (Rousseeuw and Croux, 1993);
5. The estimator for the standard deviation based on Moving Ranges (Banens et al., 1994).

These estimators are defined as function of the residuals \( e_i \), for estimating the standard deviation over a period of \( T \) years:

1. **Sample standard deviation:**

\[ S = \sqrt{\frac{1}{T-1} \sum_{i=1}^{T} (e_i - \text{mean}(e_i))^2} \]

2. **L-moment standard deviation:**

\[ S_L = \sqrt{\pi \cdot L_2} \quad \text{with} \]

\[ L_2 = \frac{1}{2} \left( \frac{T}{2} \right)^{-1} \sum_{i>j} (e_i - e_j) \]

or in the notation of Vogel and Fennessey (1993)

\[ L_2 = 2 \cdot b_1 - b_0 \quad \text{with} \]

\[ b_0 = \frac{1}{T} \sum_{i=1}^{T} e_i \quad \text{and} \quad b_1 = \frac{T}{T(T-1)} \sum_{i=2}^{T} \frac{i-1}{T-1} \cdot e_i \]
3. Standard deviation based on MAD:

$$ S_{\text{MAD}} = 1.4826 \times \text{med} \{ |e_i - \text{med } e| \} $$

4. Standard deviation $S_n$:

$$ S_n = 1.1926 \times \text{lomed}\{ |e_i - \text{himed} | \} $$

with $\text{lomed}$ the order statistic with rank $[(T = 1) / 2]$ and $\text{himed}$ the order statistic with rank $[T / 2] + 1$. Thus for $T=2r$, $\text{lomed}$ is the order statistic with rank $r$, and $\text{himed}$ the order statistic with rank $r+1$. While for $T=2r+1$, $\text{lomed}$ and $\text{himed}$ are both the order statistic with rank $r+1$. In sequel this will be denoted with $S_{\text{MMD}}$.

5. Standard deviation based on Moving Ranges :

$$ S_{\text{MR}} = \frac{\sqrt{\pi}}{2} \cdot \frac{1}{T - 1} \cdot \sum_{i=1}^{T-1} |\text{MR}_i| $$

with

$$ \text{MR}_i = |e_{i+1} - e_i| $$

With $e_i$ ($i = 1, 2, \ldots, T$) the residual values after fitting the smoother for each year $i$ and $x_{(i)}$ the $i$th order statistic, $x_{(1)} \leq x_{(2)} \leq \ldots \leq x_{(n)}$. The constants in the formulas for methods 2 to 5 are chosen such that they result in unbiased estimators for the standard deviation in case of Normally distributed residuals.

**APPRAOCH**

In total 5 estimators are being compared based on the definition of the test on trend detection and general theory of robustness. In general, years are represented by one value, and main interest will be on the number of observations between 7 and 30. In this section the estimators are introduced together with the comparison criteria and the design of the simulation study.

On the basis of real data of concentrations and inputs of substances in Dutch fresh and marine waters, a number of distributions have been fitted to be able to randomly generate the residuals needed for the simulations. This dataset consisted of 102 observations over several locations and the corresponding smoothed values, from which the residuals could be estimated.

**CRITERIA FOR COMPARISON**

In order to decide upon the most appropriate method for the estimation of the standard error of the residuals a criterion needs to be defined. Two approaches can be distinguished:

1. Comparison of the estimators on the basis of the properties of the estimators, like bias and variance of the estimator. The robustness of the estimator can be studied by investigating the influence of outlying values.

2. Comparison of the estimators with respect to the power of the trend-test. The performance of a test can be presented in 2 numbers: the size of the type I error and the size of the type II error. The type I error size denotes the probability of rejecting the null hypothesis, when in fact the null hypothesis (= no trend) is true. This type I error size is the significance level $\alpha$. In general this number is set at 5% or 10%. The size of the type II error denotes the probability that the null hypothesis is accepted, when it is false.

**SIMULATION STUDY**

The comparison criteria include amongst others the bias and variance of the estimators and most important the performance of the test. In order to investigate this a simulation study was performed. Simulations for periods of $T = 7, 10, 15, 20, 25$ and $30$ years and for different values of the trends (10%, 20%, ..., 70%) are executed. Each method of estimating the standard deviation is applied in all combinations of periods and trends. Simulations with trend = 0% are performed to find the critical values for a one-sided test corresponding with the relevant significance level $x = 10\%$. 

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In the application field under study the statistical distribution of residual errors on yearly measured values differs strongly from the normal distribution and regularly outliers occur. A mixture distribution was found to describe the dataset best. Two different distributions have been simulated, (i) a mixture of a normal, a lognormal and an extreme value distribution, and (ii) the foregoing mixture distribution (90%) plus 10% outliers at two outlying values, see figure 1.

![Figure 1 Simulated mixture distribution with added outliers of the residuals](image)

**Figure 1 Simulated mixture distribution with added outliers of the residuals**

The random variation in the observations will cause variation in:

a. the estimated trends, which form the numerator of the test statistic (1), and
b. the estimated standard deviation, which is found in the denominator of the test statistic (1).

In studying the impact of the deviations both influences need to be taken into account. In the simulation the following steps are repeatedly taken:

1. Generate residual values from the chosen distributions.
2. Define a linear trend, e.g. 10%, 20%, etc. and calculate the corresponding year values for this trend by adding the residual values from step 1.
3. Perform the LOESS method on the year values to estimate the first and last smoother values \( z(T) \) and \( z(1) \).
4. Generate new residual values from the chosen distributions.
5. Calculate the standard deviation of the residuals with each of the five alternative methods.
6. Calculate the value of the test statistic and the outcome of the test.

Steps 1 to 3 result in variation in the estimates for the trend, and thus in deviations between estimated trends and the real trend (as noted under a.). Steps 4 and 5 give representative outcomes for the estimated standard deviation for all methods (and deals with the variation meant under b.). From these results the bias and variance of the estimators can be derived. Step 6 finally combines these two results in one test outcome for every run.

**RESULTS**

The results of the theoretical study and the simulation study are described in this section. For each trend/period combination 20000 simulations are performed.

**Properties of estimators**

The first criterion upon which the estimators are being compared is on the basis of properties of the estimators. In this section the bias and variance of the estimators together with the sensitivity to outlying values are derived.
Breakdown point

On the basis of theory and simulations, the influence of outlying observations on the estimator can be derived. Staudte and Sheather [1990] and Hettmansperger [1984] describe the breakdown point as a measure of robustness of an estimator. In estimating the standard deviation we wish to avoid an estimator that can be unduly influenced by a small fraction of data. One measure to express this property is the breakdown point. The breakdown point is the fraction of nonsense points that can make the estimator valueless. The larger the breakdown point, the less sensitive the estimator is for outlying values.

The breakdown point of the five estimators is presented in table 1.

<table>
<thead>
<tr>
<th>Estimator</th>
<th>Breakdown point</th>
</tr>
</thead>
<tbody>
<tr>
<td>S</td>
<td>1/n</td>
</tr>
<tr>
<td>S_L</td>
<td>1/n</td>
</tr>
<tr>
<td>S_MAD</td>
<td>[n/2]/n</td>
</tr>
<tr>
<td>S_MMD</td>
<td>[n/2]/n</td>
</tr>
<tr>
<td>S_MR</td>
<td>1/n</td>
</tr>
</tbody>
</table>

Table 1 Breakdown point of the 5 estimators

Table 1 shows that for the sample standard deviation only one outlying observation, will cause the sample standard deviation increase largely. The sample standard deviation is very sensitive to outlying values. The L-moment standard deviation and the moving range method are equally sensitive to outliers as the sample standard deviation, with respect to the breakdown point.

The breakdown point for the MAD and MMD estimators is asymptotically _. This value indicates that only in the case of at least half f the observations are outlying values, the estimator shows a nonsense value. These estimators are not likely influenced by the occurrence of outlying values.

Bias and variance

From the simulation study the bias and the variance of the estimators were derived. Table 2 displays the average value of each estimator divided by the real value of the standard deviation (obtained from simulations) for each distribution. For an unbiased estimator, this number should be close to 1.

<table>
<thead>
<tr>
<th>Distribution without outliers</th>
<th>Distribution with outliers</th>
</tr>
</thead>
<tbody>
<tr>
<td>T = 7</td>
<td>T = 7</td>
</tr>
<tr>
<td>0.942</td>
<td>1.86</td>
</tr>
<tr>
<td>0.976</td>
<td>1.78</td>
</tr>
<tr>
<td>0.840</td>
<td>1.04</td>
</tr>
<tr>
<td>0.796</td>
<td>1.04</td>
</tr>
<tr>
<td>0.977</td>
<td>1.78</td>
</tr>
<tr>
<td>T = 10</td>
<td>T = 10</td>
</tr>
<tr>
<td>0.958</td>
<td>1.95</td>
</tr>
<tr>
<td>0.976</td>
<td>1.79</td>
</tr>
<tr>
<td>0.871</td>
<td>1.04</td>
</tr>
<tr>
<td>0.947</td>
<td>1.19</td>
</tr>
<tr>
<td>0.976</td>
<td>1.79</td>
</tr>
<tr>
<td>T = 15</td>
<td>T = 15</td>
</tr>
<tr>
<td>0.971</td>
<td>2.01</td>
</tr>
<tr>
<td>0.977</td>
<td>1.78</td>
</tr>
<tr>
<td>0.902</td>
<td>1.05</td>
</tr>
<tr>
<td>0.899</td>
<td>1.11</td>
</tr>
<tr>
<td>0.975</td>
<td>1.78</td>
</tr>
<tr>
<td>T = 20</td>
<td>T = 20</td>
</tr>
<tr>
<td>0.977</td>
<td>2.05</td>
</tr>
<tr>
<td>0.978</td>
<td>1.78</td>
</tr>
<tr>
<td>0.916</td>
<td>1.06</td>
</tr>
<tr>
<td>0.952</td>
<td>1.16</td>
</tr>
<tr>
<td>0.979</td>
<td>1.78</td>
</tr>
<tr>
<td>T = 25</td>
<td>T = 25</td>
</tr>
<tr>
<td>0.980</td>
<td>2.07</td>
</tr>
<tr>
<td>0.977</td>
<td>1.78</td>
</tr>
<tr>
<td>0.923</td>
<td>1.06</td>
</tr>
<tr>
<td>0.923</td>
<td>1.13</td>
</tr>
<tr>
<td>0.978</td>
<td>1.78</td>
</tr>
<tr>
<td>T = 30</td>
<td>T = 30</td>
</tr>
<tr>
<td>0.982</td>
<td>2.09</td>
</tr>
<tr>
<td>0.976</td>
<td>1.78</td>
</tr>
<tr>
<td>0.926</td>
<td>1.06</td>
</tr>
<tr>
<td>0.950</td>
<td>1.15</td>
</tr>
<tr>
<td>0.975</td>
<td>1.78</td>
</tr>
</tbody>
</table>

Table 2 Mean value of estimators divided by real value

For the distribution without outliers the estimators underestimate the value of the standard deviation. This is due to the form of the distribution with relatively heavy tails. The robust estimators S_MAD and S_MMD have larger bias than the other estimators. The S_L and S_MR estimator have a constant relative bias, not depending on the number of observations.

For the distribution with outliers, the robust estimators S_MAD and S_MMD are definitely performing better than the other three estimators.
Table 3 shows the efficiency of the estimators, which is the standard deviation from the simulations divided by the real value of the standard deviation for the mixture distribution, with and without outliers. For this table the smaller the value, the higher the estimator efficiency.

<table>
<thead>
<tr>
<th>Distribution without outliers</th>
<th>$T = 7$</th>
<th>$S_L$</th>
<th>$S_{MAD}$</th>
<th>$S_{MMAD}$</th>
<th>$S_{MR}$</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>$T = 10$</td>
<td>0.320</td>
<td>0.322</td>
<td>0.408</td>
<td>0.351</td>
</tr>
<tr>
<td></td>
<td>$T = 15$</td>
<td>0.268</td>
<td>0.261</td>
<td>0.284</td>
<td>0.237</td>
</tr>
<tr>
<td></td>
<td>$T = 20$</td>
<td>0.196</td>
<td>0.181</td>
<td>0.242</td>
<td>0.218</td>
</tr>
<tr>
<td></td>
<td>$T = 25$</td>
<td>0.174</td>
<td>0.160</td>
<td>0.222</td>
<td>0.181</td>
</tr>
<tr>
<td></td>
<td>$T = 30$</td>
<td>0.159</td>
<td>0.144</td>
<td>0.199</td>
<td>0.172</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Distribution with outliers</th>
<th>$T = 7$</th>
<th>1.11</th>
<th>1.00</th>
<th>0.633</th>
<th>0.621</th>
<th>1.08</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>$T = 10$</td>
<td>0.960</td>
<td>0.823</td>
<td>0.460</td>
<td>0.534</td>
<td>0.890</td>
</tr>
<tr>
<td></td>
<td>$T = 15$</td>
<td>0.802</td>
<td>0.668</td>
<td>0.360</td>
<td>0.371</td>
<td>0.719</td>
</tr>
<tr>
<td></td>
<td>$T = 20$</td>
<td>0.700</td>
<td>0.581</td>
<td>0.302</td>
<td>0.331</td>
<td>0.624</td>
</tr>
<tr>
<td></td>
<td>$T = 25$</td>
<td>0.627</td>
<td>0.519</td>
<td>0.273</td>
<td>0.282</td>
<td>0.557</td>
</tr>
<tr>
<td></td>
<td>$T = 30$</td>
<td>0.575</td>
<td>0.476</td>
<td>0.243</td>
<td>0.262</td>
<td>0.509</td>
</tr>
</tbody>
</table>

Table 3 Standard deviation of estimators divided by the real value of the standard deviation

It is clear that in case of the mixture distribution $S_{MAD}$, $S_{MMAD}$ and $S_{MR}$ have larger mean squared error relative to the real value. When outliers are present $S_{MAD}$ and $S_{MR}$ are more efficient.

Summarising the above, we can say that when no outliers are to be expected, $S$ and $S_L$ are the best estimators for the standard deviation. When outliers are likely to occur, $S_{MAD}$ is to be preferred.

Performance of the test

The second criterion upon which the estimators were compared was the performance of the test. The main goal of this study is to improve the test on trend detection. In this section the influence of the estimators on the test results are shown.

A test can be characterised by two numbers, the significance level $\alpha$ and the power of the test $\beta$. The significance level is set at 10%, thereby fixing the critical value of the test.

The power curve (out of the simulation) gives the probability that a trend is detected, depending on the real size of the trend. For 10 years of observation, the power curves are shown. The lower the probability of not detecting the trend, when a trend exists, the higher the power of the test.

When no outliers are present, there are only slight differences between the five alternative test statistics. For a trend range of 30% - 60%, the test statistics based on the standard deviation estimated by $S_{MAD}$ and $S_{MMAD}$ perform little worse than the test statistics based on the other estimators. However, the maximum deviation between the power curves is always less than 6%. For the residuals distribution including 10% outliers, the power curves show large differences. In this situation the test statistics based on the robust estimators $S_{MAD}$ and $S_{MMAD}$ outperform the other test statistics based on $S$, $S_L$ and $S_{MR}$ for an existing trend. However in case no trend exists, the power of the test based upon $S_{MAD}$ and $S_{MMAD}$ is somewhat smaller than the test based on $S$, $S_L$ and $S_{MR}$. For the specific application of this test, this was not seen as a problem. The goal is to detect a trend if one exists, with reasonable probability.

Based upon these results, the performance of $S_{MAD}$ was studied in more detail. The constant was chosen such that the estimator was unbiased for a Normal distribution. However the distribution of the residuals is not well described by a Normal distribution. For different substances residuals were calculated and again distributions were fitted. Based upon this study it was concluded that the suitable constant for $S_{MAD}$ is 1.75. The estimator of the standard deviation to be used is therefore $S_{MAD} = 1.75 \times \text{median} |e_i - \text{median} e_i|$.
CONCLUSIONS

Data about concentrations of substances are gathered for water management and policy evaluation of the North Sea. A protocol, called Trendy-tector, for the detection of trends in yearly loads has been developed and offers a structured methodology for trend detection. In this application field the statistical distribution of residual errors on yearly measured values is mostly different from the Normal distribution and regularly outliers occur. Knowing this, one can imagine that the sample standard deviation is not the most appropriate way of estimating the residual standard error. The performance of the statistical tests used for trend detection is largely influenced by the method of estimating the standard deviation. In this paper the impact of the different estimators on the performance of the tests has been investigated.

The robustness of the estimators is evaluated. The estimators $S_{MAD}$ and $S_{MMD}$ have the disadvantage of being not very efficient for distributions close to Normal, but are very robust to outlying observations. The other three estimators in this study ($S$, $S_{L}$ and $S_{MR}$) are all sensitive to outlying values.

Figure 2 Power curves for the test on trend detection as function of the various estimators
The simulations showed that the robust estimators perform well for a distribution with outliers. The power of the tests, with the standard deviation estimated by $S_{MAD}$ and $S_{MMD}$, is a bit lower for the mixture of distributions chosen on the basis of real life data. However, when outliers are present, these estimators perform significantly better than the non-robust estimators. We see throughout the whole simulation study that for the chosen distribution without outliers, the performance of the robust estimators is a little lower than the non-robust estimators. If outliers are added to the distributions, the robust estimators perform significantly better than the non-robust estimators do.

$S_{MAD}$ and $S_{MMD}$ are performing similarly with respect to the properties of the estimator and the performance of the test. However $S_{MAD}$ is preferred, because of the computational complexity of $S_{MMD}$ and a slightly better performance (bias and variance).

As a result of this study SMAD is implemented in the calculations of Trend-y-ector, with the constant suitable for the inherent distributions (i.e. 1.75).

REFERENCES

INPUT. 00/09/01-E
ASSESSMENT OF LONG-TERM CHANGES IN THE GROUNDWATER HEAD IN THE NETHERLANDS

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The objective of the present study was to obtain a nation wide overview of the development of the groundwater head during the period 1955 – 2000. Therefore 149 groundwater head series are analysed. The ‘natural’ behaviour of the groundwater head is filtered out using Box Jenkins transfer/noise modelling. The residual series is divided into periods of 5 years. For each period of five years the average value is taken. The differences between the five-year averages give an indication of the long-term change. An important step in the assessment of the long-term change is the presentation. A graphical presentation was developed such that a nation wide overview is obtained. Also the results are aggregated over all series. Aggregated presentations should be used carefully, because information about spatial and temporal variability is neglected. In this paper some aspects concerning the aggregated presentation are discussed.

PREFACE

This paper presents results from the study "Trend Development of Groundwater 2000, Period of analysis 1955 – 2000" (Kremers and Van Geer, 2000).

INTRODUCTION

In many areas in the Netherlands the groundwater head has changed during the last 50 years. In particular, a large scale lowering of the groundwater head occurred in the period 1950 – 1980. This lowering, in turn, causes negative changes in groundwater related eco-systems. An objective of the water management is to increase the groundwater head, in particular in nature areas. To be able to design and evaluate water management measures, real world information about the long-term changes of the groundwater heads is needed. This information can be obtained from observed groundwater head series. The groundwater head is observed twice a month at many locations throughout the country. In 1989 a study was carried out (Rolf, 1989), concerning the period 1950 – 1986. In this study it was concluded that the groundwater head has decreased in the sixties and seventies. In the eighties at most locations a stabilisation occurs.

In the past years measures have been taken in order to stop the lowering and preferable to rise the groundwater head to a desired target level. The Netherlands Institute of Applied Geoscience TNO carried out two studies to the long term behaviour of the groundwater head up to the end of 1999 by order of the Institute for Inland Water Management and Waste Water Treatment (RIZA). To evaluate the behaviour of the groundwater head observed groundwater series from 149 locations are analysed (Kremers and Van Geer, 2000). The objectives of this study over the period 1955 – 2000 are:

- To determine whether the stabilisation during the eighties is continued until 2000.
- To attain a general reference of the groundwater situation for large scale studies, like climatic changes.
- To gather information to support the research necessary for the Netherlands water management.
- To evaluate to which extent the effects of measures to rise the groundwater head are detectable from the observed series.

To meet the above mentioned objectives a methodology was developed. The method should produce results that are statistical sound, discriminative and presented in a user-friendly form. Discriminative implies that the method should present the permanent long-term changes of the groundwater head. The fluctuations caused by the natural behaviour of the groundwater system should be filtered out. Much attention is paid to the presentation of the results, in particular, the
balance between simple aggregated presentation and the detail at the level of individual series. This point is the main topic of the present paper.

An observed groundwater head series reflects the response of the groundwater system to many influences. Part of these influences is considered to be the natural behaviour of the groundwater head. The driving force of the natural behaviour is the fluctuation of the weather. In most cases in The Netherlands the seasonal period of one year is dominant, but also the succession of dry and wet years can be observed. The head fluctuations due to the natural behaviour of the groundwater head ranges from several tens of centimetres in lower areas with a controlled water system, to over 4 metres in the higher parts of The Netherlands. The natural behaviour of the groundwater head consists of variations around a constant mean value, even if this natural behaviour shows a pattern over several years. In addition to the natural behaviour many other causes influence the groundwater behaviour. The most important influences are changes in surface water management, land use (urbanisation) and groundwater abstraction. These artificial influences might result in a permanent change of the mean value.

The method used consist of two steps:
1. For each groundwater head series detect significant changes of the mean value,
2. Aggregate and present the results in a nation wide overview.

TREND ANALYSIS PER LOCATION

In principle, a long-term change of the mean value can be detected by estimating the mean value over two periods and testing whether these estimates are significantly different. The length of the periods is chosen to be five years. The significance with which the mean value can be estimated depends on the length of the period, the number of measurements, the fluctuation around the mean value and the serial correlation. For many groundwater head series the fluctuation around the mean and the serial correlation is dominated by the natural behaviour. In order to estimate the mean value with a smaller uncertainty, the groundwater series is split into two parts. One part accounts for the natural behaviour and the other part is the residual series. Differences in the residual series can be estimated with a smaller uncertainty.

The groundwater head series \( h_t \) is split into two parts with transfer/noise modelling (Box and Jenkins, 1976). With transfer/noise modelling a linear input response function is estimated between the groundwater head series and a series of the precipitation excess. The latter series represents the weather. The second part of the transfer/noise model is the residual series, representing all variation in the groundwater head that can not be explained by the natural behaviour. The general form of the transfer/noise model is given with the following three expressions:

\[
h_{p,t} = \delta_1 h_{p,t-1} + \delta_2 h_{p,t-2} + \ldots + \omega_0 p_t - \omega_1 p_{t-1} - \ldots
\]

\[
n_t = \phi_1 n_{t-1} + \phi_2 n_{t-2} + \ldots + \alpha_t - \phi_1 a_{t-1} - \ldots
\]

\[
h_t = h_{p,t} + n_t
\]

with:
- \( h_{p,t} \) the component of the groundwater head due to the precipitation excess at time \( t \),
- \( p_t \) the precipitation excess at time \( t \),
- \( \delta_i \) and \( \omega_i \) coefficients
- \( n_t \) the residual series
- \( a_t \) the white noise series
- \( \phi_j \) and \( \theta_j \) coefficients

The residual series represents all variation in the groundwater head that can not be explained by the natural behaviour. From the residual series the average value over five years periods is calculated. These block averages are plotted, where the most recent period (1995 – 2000) is taken as reference value. An example of these block averages is given in figure 1.
The objective is to test whether the mean value of the groundwater head in one period differs significantly from the mean value in another period. The block averages are the estimates of the mean of the underlying process (= the expected value). The difference between the expected value and the block average is the estimation error. The order of magnitude of the estimation error can be characterised by its variance, which depends on the variance and the serial correlation of the residual series. The variance of this estimation error for a residual series $h_{res}$ in period $A$ with $n$ observations, measurement interval $\Delta t$, variance $\sigma^2_{x,A}$ en serial correlation function $\rho(i\Delta t,A)$, can be calculated as:

$$\sigma^2_{hres,A} = \frac{\sigma^2_{hres,A}}{n} \left[ 1 + \frac{2}{n} \sum_{i=1}^{n} (n_i - i) \cdot \rho(i\Delta t,A) \right]$$

Assuming both periods independent the estimation error variance of the difference of the two periods ($A$ and $B$) equals the sum of the estimation variances of both periods:

$$\sigma^2_{\text{difference}} = \sigma^2_{x,A} + \sigma^2_{x,B}$$

In case the estimation errors are distributed Gaussian, the 95% confidence interval is determined as:

$$\pm 1.96 \cdot \left[ \sigma^2_{x,A} + \sigma^2_{x,B} \right]$$

The difference between the block averages is significant if it is larger than the confidence interval.

AGGREGATION AND PRESENTATION FORM

The analysis per location provides a clear picture of the structural development of the groundwater head at that location. However, to use the analysis for evaluating and developing water management, a nation wide overview is required. Therefore the results have to be aggregated and presented in such a way that general conclusions can be drawn. To show the spatial distribution the figures for each location (like figure 1) are plotted on a map. A part of this map is given in figure 2.
Figure 2. Part of the map with the results of the analysis per location.

The dark colour indicates nature areas, where lowering of the groundwater head was detected and measures are taken to stabilise or increase the head. These areas are further addressed as "nature areas". To cover the whole of the country, seven maps (size A3) are necessary. This set of maps give detailed information about the development of the groundwater head, but is not suitable to give a clear general overview. Therefore, the average development over all locations is calculated, reducing the average development of the groundwater head to one figure, similar to figure 1.

Another way of aggregating the information is to determine the percentage of significant increase, significant decrease and no significant change. The recent groundwater head in the reference period 1995 – 2000 is compared to two other periods (1955 – 1960 and 1985 – 1990). According to the confidence interval three classes of residual series can be distinguished:

- higher = the block average of the reference period is significantly higher compared to the period considered
- neutral = the block average of the reference period is not significantly different compared to the period considered
- lower = the block average of the reference period is significantly lower compared to the period considered.

Furthermore distinction is made between groundwater head series inside or outside the nature area, resulting in a total of six classes.

RESULTS

The analysis is applied to 159 groundwater head series, evenly spread over The Netherlands. All series are representing the shallow groundwater. Little more than half of these series was measured during the entire period 1955 – 2000. Most other series started in the period 1980 – 1995, while a few series were stopped in the period 1990 – 2000. Only 40 groundwater head series are located inside the nature areas.

The average development over all measurement series is given in figure 3. This figure it shows the average groundwater head drops in the period 1955 – 1965. Thereafter, the average groundwater head shows only a small change. In the period 1995 – 2000 the lowest average groundwater head occurs.

The results of testing significant changes for the periods 1955 – 1960 and 1985 – 1990 with respect to the reference period are summarised in the tables 1 and 2.
As shown in table 1, the majority of the groundwater head series remains at the same level. Outside the nature areas an important part (29%) of the groundwater head series has still decreased during the last 15 years. Inside the nature areas, 18% of the groundwater head series shows an increase. This increase is one of the objectives of the water management.

Comparing the groundwater heads in the period 1995 – 2000 to the period 1955 – 1960 leads to the conclusion that at most of the locations the groundwater head has decreased during the last 45 years.

DISCUSSION

The analysis of the groundwater head series described in the previous sections is straightforward. At the level of individual locations the presentation provides good insight into the groundwater head. However, at the aggregated level much information is lost. Many groundwater managers focus on the aggregated level, figure 3 and the tables 1 and 2. These aggregated results, however, may lead to misinterpretation. The aspects that should be taken into account are discussed below.

- The groundwater head series cover different periods. About 100 series start in the period 1955 – 1960. The rest start in the period 1960 – 1995. Approximately 40 series are shorter than 20 years in length. In the period 1980 – 2000 approximately 20 series are stopped. As a consequence the values given in figure 3 are based on different numbers of series for each five year period. Therefore, differences in values for two periods are the result of two effects: a) a general change in the groundwater head, and b) different sets of groundwater head series.
- For water management, comparison of the tables 1 and 2 is interesting. The number of series in both periods is different. Analogous to the first point the difference between the two tables can be due to either a general change, or a different set of groundwater head series.
- The number of series is rather small. In the nature areas during the period 1955 – 1960 only 15 series are available. The influence of individual series is therefore quite large. For example, comparing the percentage "higher" for the period 1955 – 1960 (20%) and 1985 – 1990 (18%), it should be taken into account that each individual series accounts for 3% to 7%. Therefore only conclusions may be drawn for big differences in percentage.
• The locations used are spread all over The Netherlands. However, this doesn’t mean that the percentages are representative for the whole of the country. Observation wells may be installed to monitor specific effects, positive as well as negative. In statistical terms the set of observations might not be random. This may lead to a bias in the results.

• Figure 3 shows the "average development" of the groundwater head. The aggregated information does not show local developments.

Aggregating the results to a national level it can not be avoided that regional detail is lost. This might lead to misinterpretation if results from, for example figure 3, are used separately. By combining several forms of presentation, for example figure 3 and tables 1 and 2, the risk of misinterpretation can be reduced. In any case the underlying basic results (for example figure 2) should be available. The aggregated presentation is useful for an indication of the general development of the groundwater head. Evaluation of measures has to be done based on the analysis at the level of individual series.

REFERENCES

THE UTILITY OF MULTIVARIATE TECHNIQUES FOR THE ANALYSIS OF FISH COMMUNITY STRUCTURES AND THE DESIGN OF MONITORING PROGRAMMES

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The instalment of informative and cost-effective fish monitoring programmes depends on an effective classification of fish communities and on the identification of key variables, i.e. variables that govern most of the diversity within and between fish communities. Classification of the ‘overall’ structure of fish communities can only be achieved by using the proper multivariate techniques, which also allow for the effective identification of key variables of the fish community. Once these key variables are identified, and explicit questions on possible changes in the fish community are asked, statistical optimisation of the monitoring programme can be achieved. Power analysis is an indispensable tool for this optimisation. In this paper we illustrate the selection of key variables, using data of the fish communities of a number of Dutch lakes. The statistical optimisation by power analysis of one of these key variables, is shown for the development in the bream biomass in Veluwemeer (1971-1987).

INTRODUCTION

The recently installed EU Water Framework Directive (WFD) aims at protecting surface waters and groundwater by integration of water management throughout the member states of the European Union (European Union, 2000). The Directive requires the member states to comply with far-reaching obligations regarding: 1) classification of rivers, lakes, transitional waters, and coastal waters; 2) identification of high, good, moderate, and worse ecological status in these waters; 3) frequency and intensity of monitoring the quality of these water bodies. The Directive aims at reaching at least ‘good’ or ‘potentially good’ quality of all surface waters in the EU by 2016. One of the biological quality elements by which the ecological status of most water bodies (except coastal waters) must be assessed is the fish fauna.

The structure of the fish community is a most integrative indicator of differences in ecological status between water bodies (classification; e.g. Karr, 1981) and of changes in this status within

<table>
<thead>
<tr>
<th>Element</th>
<th>High status</th>
<th>Good status</th>
<th>Moderate status</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fish fauna</td>
<td>Species composition and abundance correspond totally or nearly totally to undisturbed conditions. All the type specific sensitive species are present. The age structures of the fish communities show little sign of anthropogenic disturbance and are not indicative of a failure in the reproduction or development of a particular species.</td>
<td>There are slight changes in species composition and abundance from the type specific communities attributable to anthropogenic impacts on physicochemical or hydromorphological quality elements. The age structures of the fish communities show signs of disturbance attributable to anthropogenic impacts on physicochemical or hydromorphological quality elements, and, in a few instances, are indicative of a failure in the reproduction or development of a particular species to the extent that some age classes may be missing.</td>
<td>The composition and abundance of fish species differ moderately from the type specific communities attributable to anthropogenic impacts on physicochemical or hydromorphological quality elements. The age structure of the fish communities shows major signs of disturbance attributable to anthropogenic impacts on physicochemical or hydromorphological quality elements, to the extent that a moderate proportion of the type specific species are absent or of very low abundance.</td>
</tr>
</tbody>
</table>

Table 1. Normative definitions for high, good and moderate ecological status in lakes, according to the EU Water Framework Directive (the most important aspects of the definitions are printed bold).
water bodies through time (monitoring). Its inclusion in the WFD as an indicator of ecological quality is therefore logical, and its possibilities are promising. However, the fish community structure has yet been poorly identified, which shows from the very general formulation of the so-called 'normative definitions of ecological status classifications' in the Directive. Because of their generality, these definitions of high, good, and moderate status of the fish fauna (and of other quality elements) (example for lakes in Table 1) are not yet prepared for their operationalisation on behalf of future monitoring. Therefore it is necessary 1) to identify the key variables of the fish community structure that are indicative of ecological status of the fish fauna, thereby making the normative definitions operational, and 2) to explicate the incremental changes in these variables that we think are ecologically relevant.

In this study we show that by the use of multivariate analysis of fish communities, complex ecological information can be condensed and presented in relatively simple graphics, which facilitate 1) the classification of lakes, based on their fish community, 2) the following of changes in fish community structure in multivariate space, 3) the identification of the key variables of fish communities that indicate their ecological status, and 4) the optimisation of monitoring programmes producing management-relevant information.

METHODS

The analysis and classification of fish communities were based on the estimated biomass (kg•ha⁻¹) of eight common fish species in 10 Dutch freshwater lakes: smelt (Osmerus eperlanus), roach (Rutilus rutilus), bream (Abramis brama), stickleback (Gasterosteus aculeatus), perch (Perca fluviatilis), pikeperch (Stizostedion lucioperca), ruffe (Gymnocephalus cernuus), and flounder (Platichthys flesus), as distributed over five length categories (0-10, 10-20, 20-30, 30-40, >40 cm). In this way the most important aspects of the normative definitions (printed bold in Table 1) were included in our analysis: presence / absence of species was considered a measure for species composition, biomass as a measure of abundance, and length as a measure of age. Data were kindly provided by the Ministry of Transport, Public Works, and Water Management (RIZA), and the Netherlands Institute for Fisheries Research (RIVO).

Only length categories that actually exist for a particular fish species in Dutch waters were included in the analysis, resulting in 26 species-size categories. These biomass-size data were compiled for the 10 lakes, with a total of 167 lake-year combinations (LYCs). Note that for each of two lakes (Volkerak and Zoommeer), there are two datasets, i.e. one collected by the Netherlands Institute for Fisheries Research (RIVO), and one collected by a consultancy firm (W+B). The former mainly sampled the deeper parts, while the latter sampled littoral areas as well. Since neither the years nor the sampling areas are completely the same, these datasets were kept separate in the analysis. Principal components analysis (PCA) and cluster analysis were performed on all LYCs to assess their differences, to classify them and to identify the species-size categories that govern these differences and similarities (key variables).

Since the biomass-size data were not normally distributed, the following consecutive transformations were made: 1) values were rank-numbered per variable; 2) rank-numbers were normalised using a Van der Waerden transformation (SAS Institute Inc., 1990); 3) Van der Waerden rank numbers were multiplied by the standard deviation in the biomass of the original species-size variables, in order to weight ranks for the overall variability of the original variable.

The transformed variables were used to calculate a variance-covariance and a correlation matrix of all LYCs. These matrices were used as input for the PCA and cluster analysis (unweighted pair-group method, arithmetic average: UPGMA, Rohlf, 1993) respectively. Average biomass-size distributions were calculated for groups of LYCs that were roughly consistent in both the PCA and the cluster analysis. These groups were clustered again. The resulting dendrogram was the basis for a classification of Dutch fish communities.

Besides systematic differences between lakes (classification), temporal changes within lakes were investigated as well. In the PCA biplots the temporal ‘path’ of a fish community was followed through multivariate space, and the major species-size categories that govern these changes were identified. Statistical power analysis of trends was used to investigate whether the monitoring met specific information requirements (Gerrodette, 1987; Peterman, 1990; Sheppard, 1999). The trawl catch surveys of bream in Veluwemeer from 1970-1987 were used for a case study.
RESULTS AND DISCUSSION

Multivariate classification of fish communities
Principal components analysis (PCA) of the biomass-size data of the 167 Dutch lake-year combinations (LYCs) resulted in the biplot shown in Figure 1. The first two principal components explain 72% of the total variance. Each data point represents a LYC. LYCs are more similar if they are closer in multivariate space. The graph shows between-lake variance (e.g. between Tjeukemeer, in the upper right corner, and Markermeer, in the far left of the plot), as well as the (temporal) within-lake variance (the variance between LYCs of one particular lake).

![Figure 1. Biplot of a principal components analysis of 167 Dutch lake-year combinations LYCs.](image)

The five vectors represent the fish community variables that explain most of the variance along the principal component axes. These variables can be considered as key variables that explain most of the variance in LYCs within and between lakes. The position of each LYC in the direction of these vectors indicates the value of the LYC for that particular variable. The LYCs of Tjeukemeer, for instance, are more consistently located in the direction of the vector ‘Bream 20-30’ than any of the LYCs of other lakes, which indicates that the biomass of bream of 20 - 30 cm is very important in the biomass-size distribution of all LYCs of Tjeukemeer. The angles between vectors indicate how strongly the variables are correlated (e.g. ‘Ruffe 0-10’ and ‘Bream 20-30’ are almost perpendicular, indicating a lack of correlation between these variables; ‘Smelt 0-10’ and ‘Bream > 40’ are in opposite directions, indicating a strong negative correlation).
All LYCs were clustered, resulting in a dendrogram indicating the overall resemblance of the biomass-size distributions of the fish communities of each of 167 LYCs (not shown). Based on visual inspection of this dendrogram and the results of the PCA, 20 groups of LYCs were identified that were roughly consistent in both PCA and cluster analysis. The average biomass-size distributions of these twenty LYC-groups were clustered again. The resulting dendrogram (Figure 2) was used as a basis for the classification of the fish communities of Dutch lakes in four classes. This classification is mainly based on the same variables that dominated the PCA, i.e. biomass of bream, smelt and ruffe:

1. High biomass (> 200 kg•ha\(^{-1}\)), dominated by ruffe and/or smelt, with a varying, but usually high biomass of bream (Beulakerwijde 1984-91; Tjeukemeer 1981-93);
2. Intermediate / high biomass (100 - 200 kg•ha\(^{-1}\)), dominated by large bream > 30 cm (Haring-vliet 1984-94; Hollandsch Diep 1978-94; Veluwemeer 1970-87; Wolderwijd 1970-87; Volkerak (W+B) 1994-96);
3. Intermediate biomass (75 - 150 kg•ha\(^{-1}\)), dominated by smelt and ruffe, with usually only a small biomass of large bream (Veluwemeer 1992-96; Wolderwijd 1991-96; IJsselmeer 1970-75 and 1976-94; Markermeer 1970-94);
4. Low / intermediate biomass (< 75 kg•ha\(^{-1}\)), with a relatively diverse fish community, not dominated by smelt, ruffe, or bream (Zoommeer (RIVO) 1994; Volkerak (W+B) 1989; Volkerak (RIVO) 1988-96; Haringvliet 1973-83; Hollandsch Diep 1973-77; Veluwemeer 1966-68).

Figure 2. Cluster analysis of 20 LYC-groups with their average biomass-size distributions, (note that Y-axes differ, the total estimated biomass, in kg•ha\(^{-1}\) is indicated in each graph).
Temporal changes and stability of fish community classes

Temporal changes could be followed in multivariate space, using the same PCA that was used to classify Dutch fish communities. LYCs of a particular lake are closer together in the biplot (Figure 1) if their fish communities did not change greatly during the years for which data were available. Compare, for instance, the relatively compact group of LYCs of Markermeer, with the diffuse groups of LYCs of Veluwemeer and Haringvliet.

If the fish community of a particular lake changed considerably over time, the community may have switched from one class of fish community to another. For example, Haringvliet (Figure 3) changed from a class 4 community in 1973-1983 (intermediate biomass, rather diverse fish community) into a class 2 community (high biomass, dominated by large bream). Haringvliet was an estuary with a brackish to marine environment until 1970, when it was dammed, and turned almost instantly into a freshwater lake. Most of the fish population disappeared, and a new fish community gradually developed. This explains the initially low fish biomass. The period 1973-1983 can therefore be considered as a transitional period for Haringvliet.

The fish community of Veluwemeer has also undergone major changes since its creation in 1956, due to the land reclamation in IJsselmeer (Figure 4). In 1966-1968 the lake water was of good quality, and there was an extensive macrophyte cover. The fish community had low biomass and was diverse (class 4). During the 1970s and most of the 1980s the lake was highly eutrophic, with a high biomass of predominantly large bream (class 2; note the similarity to Haringvliet 1984-1994). After water quality improvements in the late 1980s the fish community changed again (towards class 3): total fish biomass decreased (especially of large bream), and

Figure 3. Temporal changes in the fish community of Haringvliet after its creation in 1970.
the amount of ruffe, roach, and perch increased. Simultaneously, macrophyte cover in Veluwemeer increased greatly.

The stability of the four fish community classes was evaluated (Table 2), in relation to the tendency of lakes to shift towards other classes. Class 1 communities appear to be very stable, while class 4 communities are unstable, and usually shift to class 2 communities. Class 2 and 3 are more or less stable. Only Veluwemeer and Wolderwijd shifted from class 2 to 3 after a strong human-induced improvement in water quality. Shifts from classes 4 to 2 to 3 are illustrated in Figs 3 and 4 for Haringvliet and Veluwemeer respectively.

### Table 2. Transitions between fish community classes as a measure of stability

<table>
<thead>
<tr>
<th>Lake name</th>
<th>Fish community class</th>
<th>4 Low biomass, diverse</th>
<th>2 Intermediate / high biomass, large bream</th>
<th>3 Intermediate biomass, smelt and ruffe</th>
<th>1 High biomass, ruffe / smelt and bream</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tjeukemeer (centuries)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1981-93</td>
</tr>
<tr>
<td>Beulakerwijde (centuries)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1984-91</td>
</tr>
<tr>
<td>IJsselmeer (68 y)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1976-94</td>
</tr>
<tr>
<td>Markermeer (68 y)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1970-94</td>
</tr>
<tr>
<td>Veluwemeer (44 y)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1970-94</td>
</tr>
<tr>
<td>Wolderwijd (32 y)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1978-94</td>
</tr>
<tr>
<td>Haringvliet (30 y)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1991-96</td>
</tr>
<tr>
<td>Hollandsch Diep (30 y)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1994-96</td>
</tr>
<tr>
<td>Volkerak (13 y)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1994 (RIVO)</td>
</tr>
<tr>
<td>Zoommeer (13 y)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1993-96 (W+B)</td>
</tr>
</tbody>
</table>

Figure 4. Temporal changes in the fish community of Veluwemeer after its creation in 1956.
Class 1 fish communities have a higher biomass than the other classes. The reason for this high fish biomass is the inlet of water from IJsselmeer into Tjeukemeer and Beulakerwijde (e.g. in dry springs; Lammens et al., 1985). The water contains high numbers of smelt that survive during the summer months, but that do not establish permanent populations. The biomass in ‘smelt-years’ is therefore a sum of the amount of smelt and of the amount of indigenous fish (mainly bream). The stability of class 1 with its high biomass is therefore (partly) a result of the artificial hydrological situation.

Class 2 fish communities are dominated by large bream with small amounts of planktivorous, small fish (e.g. smelt and young cyprinids). The reason for the absence of smelt is unknown, but could be associated with abiotic factors such as unfavourable substratum for spawning, or hydrological characteristics. The low recruitment of cyprinids could be associated with the virtual absence of macrophytes. The limited stability of the class 2 communities is mainly the result of the longevity and (probably) sufficient food resources for bream. It can be expected however, that pro-longed poor recruitment will eventually change the fish community structure.

Class 3 fish communities are dominated by small fish, mainly by ruffe and smelt. In contrast to class 1, smelt constitutes permanent populations in all class 3 communities. It is not clear why there is such a small amount of bream in these lakes. In some of them it could result from food competition between bream and ruffe (for benthos), or smelt (for zooplankton). However, in Mark-ermeer and IJsselmeer growth of bream is good (Cazemier, 1986), which indicates that bream is not out-competed for food. Other factors, such as limited recruitment, could play a role here.

Class 4 fish communities were all created artificially by the damming of marine estuaries, except Veluwemeer. In the specified periods fish biomass was low, because fish communities had just started to establish again in a water body in which all fish had died off, because of the rapid transition from a marine to a freshwater environment. Class 4 communities are all in a transitional state and are therefore intrinsically unstable.

The relationship between our fish community classification and already existing methods of assessing ecological quality by the use of fish communities is not completely clear yet. Many of these methods are based on the Index of Biotic Integrity (Karr, 1981), which comprises more different aspects than the normative definitions of the WFD and is usually adapted to the specific characteristics of a water body (Belpaire et al., 2000; Simon, 1999)

Statistical optimisation of monitoring programmes

The aim of most fish monitoring programmes is often rather generally stated as "to identify overall temporal changes in the fish community". Monitoring can only be optimised for "overall" changes however, if it is clear what this "overall" status of the fish community is. In case of the Dutch fish communities that were analysed in this study, the most important species-size categories governing differences among LYCs were the biomass of ruffe of 0-10 cm, the biomass of smelt of 0-10 cm, and the biomass of bream larger than 20 cm. Therefore it makes sense to optimise the monitoring programme with these species-size categories as key variables.

After the answer to the question for which species-size categories the monitoring should be optimised, the next question is how the monitoring can be statistically optimised. If we are interested in changes in biomass of a particular species-size category we will have to study the trend of that biomass. This can be done by performing (linear or another type) of regression and testing whether the slope $\beta_1$ is unequal to 0, viz. our null-hypothesis $H_0$: $\beta_1=0$ is tested against our alternative hypothesis $H_a$: $\beta_1 \neq 0$. The probability that we will accept a truly existing trend is called the power of the statistical test. This power depends on: 1) the number of observations ($n$), 2) the variance of the observations ($\sigma^2$), 3) the level of $\alpha$ (significance level, i.e. the maximum allowable probability of rejecting $H_0$, while it is actually true), and 4) the effect size, which is a measure of how different the actual $\beta_1$ is from $H_0$.

Power increases with $n$, $\alpha$, and effect size, and decreases with $\sigma^2$. This means that if we want to design a monitoring programme, four of our five parameters (power, $n$, $\sigma^2$, $\alpha$, effect size) have to be controlled. If, for instance, we want to be able to detect a trend over a five year period with a chance of 80% (i.e. power=0.8), with $\alpha=0.05$, and we can decrease variance by increasing the number of replicates (e.g. trawl hauls per year), we will have to make our $H_a$ explicit in order to decide how many replicates we need. In other words: we will have to
explicate what the effect size is that we want to be able to detect. Hence, an optimisation
criterion could be: the monitoring programme should be able to detect halving or doubling of
the bream biomass, within a five year period, with a power of 80% and a significance level of
5%. The maximally allowed variance, and therefore the effort (number of replicates) follows
from this criterion.

![CpUE of bream in Veluwemeer](image)

**Figure 5.** The development of the catch per unit effort (CpUE, in kg per 10 minute trawl haul)

Since we are dealing with populations of relatively long-lived species, it is expected that serial
correlation will be present in the dataset, i.e. that the population size in a particular year will
depend on the population size in the previous year(s). This effectively decreases the number of
degrees of freedom that are available for the analysis. A test (and possibly a correction) for serial
correlation has to be applied before the power analysis can be correctly performed.

**Monitoring bream biomass**

The catch per unit effort (CpUE in kg per 10 minute trawl haul) of bream (all length classes) of
yearly surveys of Veluwemeer in the period 1971-1987 was used as a historical dataset (Figure
5). A clear trend was absent and the variation in CpUE of individual hauls per year was large
(coefficient of variation of yearly mean values is 0.57). No significant serial correlation was
present in the dataset (which is rather unexpected for a long-lived species such as bream), so
we did not need to apply a correction here.

We assumed that fish biomass changed exponentially. The second assumption we made was
that the relation between CpUE and the coefficient of variation (cv) of CpUE was: \( cv(CpUE) \propto 1/\sqrt{CpUE} \) (Gerrodette 1987). Figure 6 shows the minimum size of the trends that had to exist
in order to be able to detect them with a power of 0.8, and with a power of 0.95 (\( \alpha=0.05, \)
n=17, \( cv=0.57 \)). If power = 0.8, a decrease in bream biomass of at least 11%·year\(^{-1}\), or an
increase of at least 7%·year\(^{-1}\) can be detected. If power = 0.95 the minimal decrease that can
be detected is 18%·year\(^{-1}\), and the minimal detectable increase is 8%·year\(^{-1}\).
For management purposes 17 survey-years, as in the example series, will be too long for the evaluation of the effectiveness of any management measure. A time frame of five years is more realistic. If we return to the optimisation criterion that was formulated before: "the monitoring programme should be able to detect halving or doubling of the bream biomass, within a five year period, with a power of 80% and a significance level of 5%", we can calculate the maximum coefficient of variation that is allowed to still be able to detect these trends. In order to be able to detect halving of bream biomass, the maximum cv is 0.13; doubling can be detected with a maximum cv of 0.18. Other values of maximum allowable values of cv for combinations of power and a are shown in Table 3. Even with power = 0.8, and $\alpha = 0.1$, a maximum cv of only 0.23 is allowed. The values of cv range from 0.15 - 0.85 in the dataset, and only in one year cv < 0.23. The above implies that in order to be able to detect halving or doubling of bream biomass in a five year period of monitoring, the cv of the mean yearly CpUEs has to be drastically reduced. However, a decrease of cv by decreasing sampling variability can only be achieved up to a certain level, since there also are natural fluctuations in bream biomass, independent of sampling techniques. Moreover, decreasing sampling variability might lead to the increase of the serial correlation between years, which is now non-significant, because it is obscured by the large cvs. Since serial correlation decreases power of a survey, decrease of the cv might lead to higher power, but this effect could be (partly) counteracted by a decrease of power, due to the increasing serial correlation.

<table>
<thead>
<tr>
<th>Power</th>
<th>halving $\alpha=0.05$</th>
<th>halving $\alpha=0.1$</th>
<th>doubling $\alpha=0.05$</th>
<th>doubling $\alpha=0.1$</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.8</td>
<td>0.13</td>
<td>0.16</td>
<td>0.18</td>
<td>0.23</td>
</tr>
<tr>
<td>0.9</td>
<td>0.11</td>
<td>0.13</td>
<td>0.16</td>
<td>0.19</td>
</tr>
<tr>
<td>0.95</td>
<td>0.10</td>
<td>0.12</td>
<td>0.14</td>
<td>0.17</td>
</tr>
</tbody>
</table>

Table 3. Maximum coefficients of variation (cvs) of the yearly biomass estimates that are allowed for detecting halving, or doubling of the bream stock in Veluwemeer over a five year period (six surveys), given particular combinations of statistical power and significance levels ($\alpha$).
A decrease in sampling variability can, in some cases, result from an increase in the number of hauls per year. In our data, however, no relationship between the number of hauls and the size of cv was detected, which was probably due to the small range in the number of hauls (in 11 of the 17 years 5 hauls were made). Another option to reduce cv is by stratification of the samples, i.e. the lake is subdivided in homogeneous areas (e.g. shallow parts v. deep parts), and the hauls for different areas are analysed separately. It is expected that more homogenous areas will have lower cv values. Since this study shows that the present variance between trawl hauls is too large to effectively detect the halving or doubling of the bream stock over a five year period, and since no relation between variance and number of hauls could be detected, stratification of trawl hauls appears to be the only way to decrease variance, and so to increase power of the trawl surveys.

CONCLUSIONS

Multivariate analysis of biomass-size distributions of freshwater fish communities provides ways of presenting complex data into relatively simple graphics that can be helpful in tracing the most important species-size categories (key variables) that structure fish communities. The identification of such species-size categories and the assessment of variances in the biomass of these categories are important for the design of informative and cost-effective monitoring programmes, using power analysis for optimisation.

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USE OF ARTIFICIAL NEURAL NETWORKS IN INTEGRATED WATER MANAGEMENT

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An artificial neural network is nowadays recognized as a very promising tool for relating input data to output data. It is said that the possibilities of artificial neural networks are unlimited. Here we focus on the potential role of neural networks in integrated water management. An Artificial Neural Network (ANN) is a mathematical methodology which describes relations between causes (input data) and effects (output data). An ANN can be a powerful modelling tool to relate causes and effects. The applications are widespread and vary from optimization of measuring networks, on-line water management, prediction of drinking water consumption, on-line steering of waste water treatment plants and sewage systems, up to more specific applications as establishing a relationship between the observed erosion of groyne field sediments and the characteristics of passing vessels. Especially where processes are complex neural networks can open new possibilities for understanding and modelling these kind of complex processes. It is therefore recommended to work out various case studies. It is expected that neural networks will become fully accepted mathematical tools in integrated water management within the next few years.

Keywords: Applications; artificial neural networks; case-studies; future; human brain; strength and weakness

INTRODUCTION

The technology of Artificial Neural Networks is recognized as a very promising modelling-tool. In a recent article in H2O (Aafjes et al, 1997) it is shown that using neural networks, drinking water consumption is predicted more accurate compared to conventional techniques. An ANN can also be a powerful tool in other applications. The theoretical possibilities of neural networks are unlimited. Wild stories pass about the applications. A recent article in Intermediair has the announcing title 'Artificial Neural Network goes at the stock floor' (Zuidema, 1997). Predicting stock rates using an ANN should guarantee making a lot of money. In a first impression it is difficult to judge these statements, considering the complexity of the theoretical background. The question rises what the role of artificial neural networks can be in integrated water management. The primary goal of this article is to describe and illustrate the methodology of artificial neural networks. Secondly, the possibilities of artificial neural networks in integrated water management are discussed, based on a strength-weakness analysis. In addition, the use of an ANN is illustrated by two examples.

THEORY

The ANN concept is originally developed to simulate the human brain. The human brain is a close network of connected biological neurons by nerves. An ANN is constructed of artificial neurons, the mathematical elements within the network or the so-called processing elements. The way the neurons are connected determines the shape of the network. The analogy with human brain gives the ANN methodology a challenge. Human beings are able to perform tasks with respect to recognition, combination, and generalisation. These functions are difficult to fit in a computer-algorithm. A computer is able to perform long series of calculations according to explicit algorithms. Herewith, human beings make mistakes. An ANN combines the human association and the analytical power of a computer. Feeding the ANN continuously with new information simulates the association; it is learning from examples and all by itself. Often a neuron has many input paths (figure 1). An ANN consists of many artificial neurons joined together (figure 1). It combines the values of these input paths, usually by summation. A transfer
function then modifies the combined input. This transfer function can be a simple mathematical function like a sigmoid, hyperbolic tangent, sine or linear function. The output of the transfer function is passed directly to the output path of the neuron. Mostly the output path of a neuron is connected to the input paths of other neurons. Connection weights represent the strength of neural connections. Usually neurons are organized in groups called hidden layers. Two layers called buffers are related to the outside world. An input buffer which consists of various inputs where data is presented to the network, and an output buffer which holds response of the network to a given input.

Based on the calculated differences between the derived or measured output of a certain process and the results obtained by ANN, the so-called residuals or errors, parameters or weights are automatically adjusted in order to minimize the final and total error. This is called learning. An ANN is a mathematical technique that searches automatically for the best linear or non-linear relationships between cause (input) and effect (output). Neural computing differs from traditional computing in several important ways. Unlike traditional expert systems where knowledge is made explicit in the form of rules, neural networks generate their own rules by learning from examples shown to them. Learning is achieved through a learning rule which adapts or changes the connection weights of the network in response to the example inputs and (optionally) the desired outputs of these inputs. It is therefore very suitable to simulate relations between causes and effects without taking into account the nature of these processes. There are two main phases in the operation of an ANN, learning and recall (figure 2). Learning is the process of adapting or modifying the connection weights in response to examples. Recall refers to how the ANN performs when new but representative data are presented to the ANN (prediction). Before starting with the learning phase it is very important to verify and screen the data to be sure the examples presented to the ANN are representative examples of the problem to be solved or modelled.
SOFTWARE AND SOCIETIES

Professional software of neural networks is rare. Sometimes, an ANN tool is presented as a module within statistical software packages. On the other hand, more specific and more professional software packages have been developed such as NEURAL-COMPUTING (NeuralWare, Inc., 1996). Societies, such as the society of Artificial Neural Networks in the Netherlands (VANN), follow and discuss the latest developments on neural networks and its applications. Government, semi government, research institutes, universities, and consulting engineers participate in this society.

THE PRODUCT

With help of the software, an ANN, which is in fact a mathematical model, can be developed. As conventional models, an ANN needs to be defined, calibrated and validated. In this respect calibrating is called learning and validating is called recall. The final model exists of several optimized mathematical equations with the corresponding optimized weight factors. These equations determine the relationships between causes and effects. An ANN (i.e. the model) can be transformed into code that can be used as a stand-alone system or can be implemented within existing hardware and software.

POSSIBLE APPLICATIONS

Possible applications of ANN in integrated water management will be briefly explained.

Operational water management and real time control. In operational water management, optimized decisions have to be taken based on much information. Neural networks are especially suitable to determine the best relations between steering parameters and the resulting effects. Using these relations it is possible to adjust management immediately based on actual measurements. Applications neural networks in integrated water management are operational water management, steering of pumps or weirs (ground water steered surface water management) or the flushing of canals. Applications also concern real time control of wastewater treatment plants and sewage systems.

Optimizing and rationalisation of measuring networks. In general, the performance of measurements is expensive. The optimal density and frequency of measurements strongly depends on temporal and spatial variability of those parameters. Using multivariate techniques, these variations can be quantified in order to optimize the measuring networks, the measuring locations, measuring frequency and measuring strategy. Optimizing measuring networks on forehand can reduce the costs of performing measurements substantially.

Prediction models. Many factors play a role in prediction of the weather, stock and currency/exchange rates, energy and drinking/water production, water level discharge relations, flow and sediment transport, et cetera. Mostly, the exact causes and relations are not clear, but potential significant variables are known. An ANN is very suitable to fit the best relations and trace all possible relations based on information in the past.

Simulation models. Deterministic models are common in integrated water management (morphological-, ecological-, ground water-, flow-, or contaminant- models). For deterministic simulations it is required that the nature of the processes is known. An ANN does not require this. Based on information like measurement data, the relations are fitted. It often appears that an ANN performs and predicts better than a deterministic model. Also, an ANN can be used in order to correct errors from deterministic models. In these cases an ANN is used to study and quantify systematic patterns, like trends, in the errors from the output of deterministic models.

Ecology. In common, the nature of physical and chemical processes is known adequately in order to get good results using deterministic simulation models. However, ecological processes are derived from physical and chemical processes, and are therefore very complex. More usual, ecological effects are predicted based on expert-opinions or empirical relations. In these applications, neural networks are especially suitable for finding and describing these relations.
Special applications. Complex processes influence rating curves and duration curves, such as discharge water level relations. These processes disturb the singularity of such a relation and therefore introduce extra dimensions, which should be taken into account during construction of these rating curves. Often it is hardly possible to define these relations adequately. Using an ANN, it is possible to define multi dimensional relations without knowing all the processes in detail.

Trend- and time series analysis. Due to the many degrees of freedom, an ANN is able to trace and quantify both linear and non-linear trends and seasonal effects. Advantage of this method compared to the conventional time series analysis is that it is not necessary to define whether the relation is linear or non-linear. The technology can be used also as an interpolation-tool to estimate intermediate values.

A FEW APPLICATIONS

In the past two years ANN has been used successfully in several integrated water management applications. Four cases will be described shortly.

Case 1: Relating erosion of groyne field sediments to the characteristics of passing vessels

Purpose. Together with the National Institute for Inland Water Management and Waste Water Treatment (RIZA) a study has been carried out to establish a relationship between observed erosion of groyne field sediments and the noted and derived characteristics of passing vessels (Schulze et al, 1998).

Background. The river Waal, which carries about 65% of the total Rhine discharge, connects some of the main industrial areas of Germany, Switzerland and France with the North Sea port of Rotterdam. The shipping density on the river Waal is among the highest of all inland waterways in the world. To protect the mainly sandy riverbanks from erosion and to combat sedimentation of the riverbed, groyne have been build all along the river at intervals of approximately 200 meters perpendicular to the riverbank. At moderate and low river flows the hydrodynamic and sedimentation processes in the area between the groyne are changed by navigation traffic. These changes have become more important as vessel size and power have increased enormously over time. To quantify the impact on erosion of groyne field sediments a series of measurements were carried out. Sediment resuspension between the groyne was measured by pressure devices, electromagnetic flow meters and optical backscatter sensors. At the same time several characteristics of the vessels passing the area were noted, such as length, width, speed, duration, etc. Beside the noted characteristics, also derived characteristics such as volume, wetted cross-section, average return-flow etc. were used. These characteristics are mentioned in literature as potentially significant variables in explaining the erosion of groyne field sediments. Beforehand it is difficult to tell which of the characteristics most significantly affects the erosion of groyne field sediments under conditions as prevailing in the area of research. In addition it is also not known what type of relationship exists between the erosion of groyne field sediments and the various characteristics of the passing vessels.

![Figure 3. ANN architecture for relating erosion of groyne field sediments to the characteristics of passing vessels.](image)
Approach and results. Because of the unknown relationship between the erosion of groyne field sediments and the various characteristics of the passing vessels and inadequacy of conventional methods, an ANN was used for this study (figure 3). First the developed ANN was used to carry out sensitivity analyses to find those characteristics that contribute most significantly to the observed variation in the erosion of groyne field sediments. The results were compared with those found in literature. Secondly the ANN was successfully used as a tool to predict (future) erosion of groyne field sediments as a function of (changing) vessel characteristics and traffic increase (figure 4). It learned successfully the general transfer function between the erosion of groyne field sediments and the various characteristics of the passing vessels.

![predicted erosion by ANN versus measured erosion](image)

Figure 4. ANN prediction of erosion of groyne field sediments due to the various characteristics of the passing vessels.

Conclusion. Thus far the ANN was the only tool capable of relating the erosion of groyne field sediments to the various characteristics of passing vessels adequate enough, at relatively low cost and at short notice. The latest developed ANN has been transformed into an operating system for the prediction of erosion of groyne field sediments.

Case 2: Analysis of time dependent rating curves on the river Bovenrijn near Lobith

Purpose. Together with the Ministry of Transport, Public works and Water Management a study has been carried out to establish a multidimensional relationship between the measured discharges and the measured water levels over time for the river Rhine.

Background. When a number of discharges have been measured in one cross-section of a rectangular cross-section of the river, it is possible to determine a relationship between the discharge of the section and the corresponding water level: the unique stage discharge curve or Q-h relationship. A real river however almost never meets the needs of a rectangular riverbed. Complex phenomena such as changes in cross-sectional area over time, changing water levels because of flood waves, changes in vegetation, backwater effects, dredging, etc cause a non-unique Q-h relationship, which is almost impossible to put in mathematical terms. The Bovenrijn near Lobith is such a river.

Approach and results. To establish an accurate relationship between discharge and the water level over time, measured discharges, measured water levels and the corresponding dates (year and month) were used. Since complex, and probably non-linear, processes are of great influence on the relationship between measured discharge and measured water levels over time conventional methods appeared to be inadequate to establish a relationship accurate enough to predict discharges. An ANN was successfully used to establish an accurate relationship between measured discharges and the measured water levels over time (figure 5). The developed ANN was used as a tool or model to predict discharges due to corresponding water levels over time (figure 6).
Conclusion. Because of the complex relationship between measured discharge and measured water levels over time conventional methods appeared to be less adequate. The ANN was capable to relate measured discharges to measured water levels over time adequate enough, at relatively low cost and at short notice.

Case 3: An intelligent prediction system for the on-line prediction of water consumption

Purpose. Together with the Amsterdam Water Supply a study has been carried out to develop an intelligent prediction system. The main purpose of the feasibility study was designing a first version of an on-line and hourly based prediction system of the consumption of drinking water for the city of Amsterdam.

Background. The consumption of drinking water varies in time. The consumption is influenced by many factors like temperature, radiation, type of day during the week, moments during the day, soccer games, (public) holidays, etc. The production of drinking water has to be attuned to these variations so that in all cases sufficient drinking water can be distributed. The drinking water production plants are generally designed on the maximum demand of the day. In ideal situations drinking water plants are working efficient (cost and quality) with constant flow. In practice this ideal situation will never occur because consumption varies during the day, week and year. Therefore to improve quality and minimize costs, an accurate on-line prediction model is of great importance.

Beforehand it is difficult to tell which of the factors most significantly affects the drinking water consumption. In addition it is also not known what type of relationships exists between the observed water consumption and the various factors. Since the unknown relationships, the
complexity of the problem, the wish for on-line prediction and the limitations of traditional models Artificial Neural Networks (ANN) were considered.

Approach and results. For the case in Amsterdam a simple ANN was used to predict the water consumption of Amsterdam due to several factors like climate, historical consumption, holidays, special days like important soccer games, etc. The model exists of several optimized mathematical equations with the corresponding optimized weight factors. These equations determine the relationships between causes and effects. To explore the possibilities of an on-line prediction system the defined ANN (i.e. the model) is transformed into a first version of such a system, named DRIVOS (figure 7). This system predicts the hourly water consumption for the next 24 hours and recalibrates every 4 hours (figure 8).

![Figure 7. DRIVOS](image)

![Figure 8. On-line predicted water consumption for 1 week in February.](image)
Conclusion. The first version of the ‘intelligent’ on-line prediction system, called DRIVOS, is based on a quite simple ANN architecture and has already a prediction accuracy that varies from 75% to 99% and will be improved.

Case 4: Use of artificial neural networks in optimising a water quality monitoring network

Purpose. Together with the National Institute for Inland Water Management and Waste Water Treatment (RIZA) a pilot study has been carried out for the use of artificial neural networks in optimisation of monitoring networks.

Background. The use of artificial neural networks is common in a broad range of applications. It is very common in operational applications and also for predicting it is a very suitable technique. However, for construction and optimisation of a monitoring network it is, until now, less common. The aim of this study is to find out if artificial neural networks can contribute to spatial optimisation of a water quality monitoring network. The optimisation of the water quality monitoring network of the IJsselmeer area is used as a case study. The study can be seen as a pilot study for the use of artificial neural networks in optimisation of monitoring networks. The IJsselmeer area is chosen for this study because of its complex physical character for waterflow direction, because of i.a. changing wind directions and water depth.

Approach and results. For the spatial optimisation of the IJsselmeer area time series of parameters were chosen from different parameter groups, which show a broad range of behaviour. These data are chloride, nitrate, phosphorous, lead and the amount of floating particles. Criteria for these parameters were: a high concentration present, representative for soluble parameters and particular parameters. Criteria for the locations were water flow direction, location of issue, present and future function of the water system and subjects and policy aims. These parameters and locations were chosen by an inventory of information needs under the users of the data of the monitoring network. Time series are from 1992 to 1998, with 12 or 24 measurements per year. Linear modelling (ARX or ARIMA models) was used to find relationships between locations. The models between locations were multivariate transfer models. These models can also be defined by an artificial neural network instead of an ARX model. This is done for the parameter chloride as a pilot to find out the use of artificial neural networks for prediction of these time series. Chloride was chosen because of the best quality of the time series, showing almost no gaps. Figure 9 shows the prediction results by using a ARX model. The left-side of the vertical line is used for defining the model and the right side is used for predicting chloride at the location of Andijk. Figure 10 shows a better result when a neural network is defined and used for predicting chloride at Andijk for the same period.

![Figure 9. Prediction of chloride at Andijk by an ARX Model.](image-url)
Conclusion. Because of the complex and often non-linear relationships between locations in most cases conventional methods appeared to be less adequate. At the same time it is possible to develop a complex monitoring network of the whole measuring network based on ANN for on-line detection of trends and casualties.

STRENGTH AND WEAKNESS ANALYSIS

An ANN is a multi-use-modelling-tool used to describe all kind of relations between cause and effect. Requirements in order to construct or define neural networks are: (1) representative data, (2) software and hardware with enough memory and capacity, (3) close cooperation with experts with respect to the discipline concerned, and (4) knowledge of construction of an ANN and statistics to validate those models. The advantages of using neural networks are: (1) almost every relation between cause and effect can be modelled, (2) no assumptions need to be made considering the nature of the relation, (3) preprocessing of data is minimized, and (4) once calibrated or trained, an ANN can be transformed easily into program code so anyone can use it. Disadvantages that should be mentioned are: (1) the problem is set as a black box instead of studying the processes itself, (2) with respect to the degrees of freedom, there is a risk of over-parameterization. It is therefore necessary to be familiar with statistical techniques that can locate, visualize and quantify over-parameterization, and (3) it is not possible to simulate scenario's with other boundary conditions since the ANN did not learn how to handle these situations. Especially in situations where knowledge of individual processes is meager, but enough representative data are available, an ANN can be an adequate modelling tool. Often, the prediction of an ANN is better compared to deterministic models.

FUTURE

Nowadays, neural networks are in the reclamation stage whereas the applications of neural networks in the several disciplines are tested. An ANN is a powerful tool to get optimized descriptions of relationships between cause (input) and effect (output) irrespective of the
behind laying process. This article has shown that applications of neural networks are widespread and vary from optimization of measuring networks, on-line water management, prediction of drinking water consumption, on-line steering of waste water treatment plants and sewage systems, up to more specific applications as establishing a relationship between the observed erosion of groyne field sediments and the characteristics of passing vessels. Especially where processes are complex neural networks are able to open new possibilities for understanding and modelling these kind of complex processes. It is therefore recommended to work out various case studies. In order to build an ANN it is necessary to be familiar with mathematics and statistics. A fully trained ANN can easily be transformed to an operational model inside or beside existing computer systems. It is expected that neural networks will become a fully accepted mathematical tool in integrated water management within the next few years.

REFERENCES


FUZZY C-MEANS CLUSTERING: A MULTIVARIATE TECHNIQUE FOR THE EVALUATION OF SURFACE-WATER-QUALITY MONITORING NETWORK DATA

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Most point source pollution in surface water in the Netherlands is minimised and there is a growing interest in the diffuse pollution and generic processes that explain surface water quality variation, summarised as the ‘integrated water system approach’. Spatial surface water quality variability may be high, reflecting the geology, soil type, land use and hydrology of the drainage area. Results from monitoring programs are used here to classify water types in the Netherlands and to identify the biogeochemical processes that dominate for each type. This is done by the use of the multivariate technique fuzzy c-means clustering.

This technique groups individual monitoring sites, based on the concentrations of the major ions, into well defined water types. Fuzzy c-means clustering provides a conceptual framework to interpret water quality in terms of both source and biogeochemical processes.

Nine water types seem sufficient to describe almost all sample points of the general surface water quality in the Netherlands. These water types range from unchanged rainwater to brackish water, while pollution and seasonal effects are recognised and accounted for. Surface water chemistry in the Netherlands appears to vary strongly over space. Therefore future monitoring should emphasise more on spatial variations in macro chemistry, than the traditional focus on temporal variations of nutrients and contaminants. More conclusions can be drawn from the spatial distribution of macro chemical compounds than was hitherto realised by use of the conceptual framework developed here.

INTRODUCTION

Surface water quality monitoring is traditionally focussed on the detection of site specific temporal features. At the national scale in a selected number of sites water composition is monitored for water quality parameters on a daily to monthly basis. This high monitoring frequency is used in the international rivers, especially to detect calamities. The lower frequencies, that occur in many more points, are used to test conformance to norms and to investigate trends and effects of measures to improve water quality.

Monthly measurements are usual in regional water monitoring networks. In the beginning of monitoring in the sixties and seventies this was done to control point source pollution like sewage and to monitor the improvement of water quality after treatment plants were completed. Nowadays, point source pollution has been minimised in the Netherlands and most parts of the Western World. The goals of monitoring shift towards the study of diffuse sources of pollution, like acid rain and nitrate leaching. This shift in interest is called the ‘water system’ or ‘integrated water management’ approach. In this approach surface water is considered part of the water cycle. Understanding the water-quality-governing-processes, rather than just to test water composition against water quality standards, is crucial in this approach.

Since traditionally focus was on temporal variation, less is known about spatial variation of water quality and its dominant processes. These processes are pollution by diffuse sources, ground water seepage (for instance brackish or anoxic, iron rich ground water), acid buffering, and biological processes. To describe or detect these processes the macro chemical composition must be evaluated, because macro chemical constituents, the major ions, DOC, plant biomass, pH and oxygen, dominate the chemical equilibria that reflect the main biogeochemical processes in the surface water. However, as yet no conceptual framework to evaluate surface water quality based on these major chemicals exists. Some efforts to establish such a framework have been made, e.g. in the Netherlands the classification by van Wirdum (van Wirdum, 1992). This classification of water types is aimed to interpret the contribution of several sources to the surface water at a specific site, like seawater (thalassocline), groundwater

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(lithocline) and rainwater (atmocline), based on three macrochemical parameters: calcium, chloride and conductivity.

To develop a more sophisticated conceptual framework we used data on the macro-chemical constituents from existing monitoring networks, stored in the STOWA database. This database covers the conterminous Dutch territory and is currently being developed. The multivariate technique fuzzy c-means clustering (FCM) is used to group the sites by their macro chemical composition into homogeneous groups or surface water quality types. These types, detected as multivariate homogeneous groups of monitoring sites, may react differently to external influences and can be regarded as genetic water types.

These water types provide a framework to assess the vulnerability of surface based on its source and on dominant biogeochemical and pollution processes. Another application is to improve the design of monitoring programs.

THE HYDROLOGICAL SETTING OF THE NETHERLANDS

The Netherlands is located in a deltaic plain: from Germany the River Rhine and from Belgium the River Meuse enter the country. The background of figure 1 shows the main geographical features of the Netherlands.

Based on geology the Netherlands can be divided into two major areas. The eastern and southern parts of the country consist of sandy soils above sea level. The landscape here was shaped during the Pleistocene glacial periods and the geology consists of cover sands, ice pushed ridges en till deposits. Intensive agriculture is the main land use here. The area is drained by relatively free flowing lowland streams. The groundwater in these areas ranges from acid to base rich and depend on the mineral content of the soil. Small isolated lakes, recharged by rainwater or groundwater, occur in the area.
The western and northern part of the country is below sea level. These areas exist of polders with a controlled water table, often within a few cm. The soils are fluviatile or marine clays and peat of Holocene origin. Due to the high ground water level land use is often limited to dairy farming. In the coastal zone the groundwater in many places is of marine origin and seeps into the surface water, especially in the deeper polders and in dry periods. Along the North Sea shore sandy dunes store rainwater and so oligotrophic surface water can occur here.

The main rivers form an important part of the surface water system as they are used to supply water deficits in summer. Over 65% of the surface water in the Netherlands originates from these rivers (Zwolsman, 1996). During winter this water mixes with rainwater and surface run-off; it is discharged via ‘boezem’ canals (main transportation canal) to the large rivers and to the sea.

METHODS

The STOWA data set

The STOWA data set contains abiotic surface water quality data from monitoring sites where also biotic monitoring took place. The sites cover the conterminous Dutch territory. The data are collected from regional water quality monitoring networks, managed by regional waterboards. Sampling and chemical analysis are carried out on unfiltered samples, according to standard protocols. Some differences in protocols may exist between the regional monitoring networks, and some differences in parameter set and analytical methods occur. The data set also contains site specific information like soil type, land use and morphology of the water body per sampling site.

The data set contains over 46,000 samples, taken at 2278 location in the Netherlands in the period 1978 to 1991. Many sites were sampled on a monthly basis; 1355 sites were not sampled throughout the year. Measured water quality parameters are ammonium, bicarbonate, calcium, chloride, chlorophyll-a, Secchi visibility, iron, Kjeldahl nitrogen, magnesium, nitrate, potassium, sodium, sulphate, ortho-phosphate, oxygen and oxygen saturation, pH, silica and total phosphorus. Detection limit cases were replaced by 0.7 times the highest detection limit. As only macro chemical constituents are used, less than 3% of the samples for all parameters samples have detection limits. High detection limit values were assigned a missing value.

Exploratory data analysis

All variables, except oxygen, bicarbonate and pH, show a log-normal multimodal distribution. In general chloride, sodium, potassium, magnesium and sulphate show a linear relation described by seawater dilution (see figure 4). However, some deviations from seawater dilution occur, caused by specific processes. These specific processes will be discussed in description of the water types below.

A significant correlation exists between Kjeldahl nitrogen and ammonium, oxygen saturation and oxygen content and between ortho-phosphate and total phosphorus. For this reason Kjeldahl nitrogen, oxygen saturation and ortho-phosphate are not used for further analysis.

To incorporate temporal variation, the time series at each monitoring site was split into a summer and winter average. This was done using data of the whole sampling period of 1978-1991. Summer was defined from April to September; winter from October to April. The number of water quality variables is doubled to try to incorporate differences in yearly dynamics of the water quality between sites that show the same annual average water composition.

Most water quality parameters show higher concentrations in summer than in winter, caused by evaporation in summer and dilution in winter by rainfall. However, nitrates are up to a factor 3 higher in winter in most water types, probably caused by manure leaching. Additionally figure 2 shows that high nitrate concentrations in winter coincide with small difference between summer and winter concentration. This can be explained that these sites have a high load on nitrate over the year. This implicates also that nitrate concentrations vary more in space between sites than in time at individual sites. It is interesting to notice that the average pH values are not lower than 6, indicating that acidification is not a threat to surface water in the Netherlands (box- and whisker plot pH, figure 3), and is explained by the inlet of base rich river water, liming and calcium-carbonate mineral buffering of soils.
Figure 2. Scatter plot of nitrate average concentration in summer vs difference between summer and winter average concentration (mgN/l), indicating leaching in winter for most water types. For the effluent and manure and domestic water type high nitrate concentrations in winter also occur in summer, indicating all year round pollution.

**Multivariate data evaluation method: fuzzy c-means clustering**

Multivariate statistical methods are used to find patterns in the data set. These patterns reflect important differences in water sources, pollution and biogeochemical processes and will be referred to as water types. Fuzzy c-means clustering (FCM) is used to evaluate and interpret surface water quality data.

C- or k-means clustering is a partitioning technique, directed to minimise the maximum distance of a case to its cluster centre. Conventional (hard) c-means clustering assigns each case (water sample) to a unique cluster. Cases that are intermediate between two or more clusters, or outliers that do not belong to any cluster, are forced into a cluster. This may distort the cluster structure and may present the data structure incorrectly. In FCM a different approach allows some ‘fuzziness’ in the allocation of cases to a cluster, and per site memberships are calculated to every cluster. These memberships have values between 0 and 1; 0 for a site that does not belong to a cluster and 1 if it is in the cluster centre. Per sample all memberships sum up to 1.

Here we calculate memberships using the standardised distance, the values of the variable divided by the standard deviation. In case of mixing processes or the occurrence of several processes within a sample memberships are shared to several clusters. This calculation of memberships seems appropriate for water composition data processing since mixing is usually expected and water samples need not fit entirely into a cluster. The theoretical background of FCM is given by Bezdek (1981) and Vriend et al. (1988).

Theoretically each sample belongs by membership to every cluster. For interpretation the clusters must be defuzzified and assigned to a specific cluster. A sample can not be defuzzified if it is a fifty-fifty mixture between two clusters, or more generally has equal memberships on all or most clusters. This happened in about ten percent of the cases.

The choice of the number of clusters is important. The number of clusters is commonly decided heuristically on the basis of spatial distribution, hydrochemical interpretability, and the unimodality of parameter distribution within a defuzzified cluster. FCM cluster analysis has been applied for groundwater quality (Frapporti et al, 1993) and soil quality (McBratney and de Gruijter, 1992).
Data set preparation for FCM cluster analysis

FCM cluster analysis was carried out with the log-transformed variables ammonium, bicarbonate, calcium, chloride, chlorophyll-a, visibility, iron, magnesium, nitrate, oxygen pH, sodium, sulphate and total phosphorus. Potassium and silica are not used because they were measured in an insufficient number of samples. Still, many ‘incomplete’ chemical analyses were present after this selection. If we select sites at which at least chloride, nitrate, ammonium, oxygen, sulphate and total phosphorus are analysed only 363 site remain, mostly in the southern and eastern part of the Netherlands. However, FCM is expected to be robust for missing values and 963 sites remained that will be studied below.

RESULTS: ADOPTED CLUSTER MODEL

A nine-cluster model was chosen to describe the main features of the water chemistry of surface waters in the Netherlands. To facilitate the description a label is assigned to each cluster that characterises the biogeochemistry. In the characterisation also site specific information like soil type, land use and local geomorphology was incorporated. Identified cluster will be referred to as water types by their label. Figure 1 shows the geographic distribution of the defuzzified water types and Figure 3 shows the result of cluster analysis by box- and whisker plots of the most significant parameters ammonium, bicarbonate, calcium, chloride, chlorophyll-a, iron, nitrate, oxygen, pH, sulphate and total phosphorus.

Table 1. Cluster centres or typical water compositions of the nine cluster model. Concentrations in mg/l, except chlorophyll-a (µg/) and depth visibility (m). sum = average summer (April to September), win = winter (October to March).
Figure 3. Box- and whiskerplots of water quality parameters per water type and per season (summer or winter). Only defuzzified samples are used (see figure 1).
Figure 3. Box- and whiskerplots of water quality parameters per water type and per season (summer or winter). Only defuzzified samples are used (see figure 1).
Softwater type
The softwater type is characterised by low concentrations for all parameters, including nutrients; both winter and summer concentration levels are low. The bicarbonate concentrations are low, and indicate low buffering capacity for acidification. Nitrate concentrations are higher in winter: median concentration of 1 mgN/l in summer and 2 mgN/l in winter. The concentration ranges are comparable to rainwater and base content is low. Oxygen concentration levels are around 10 mg/l and pH is neutral between 7 and 8. The water type is found in the central part of the Netherlands, near the main rivers Rhine, IJssel and in the north around the sandy plateau in the up- and midstream parts of lowland streams, flowing from sandy highs.

Lowland stream manure polluted water type
Most parameters in the lowland stream manure polluted water type show concentration levels comparable to the softwater type. Nitrate, calcium and bicarbonate concentrations are higher, that indicates calcium-carbonate buffering, combined with nitrate leaching, intensified by run-off in winter. Oxygen is above 10 mg/l. This water type is interpreted as recent rainwater recharge, affected by manure. The manure is oxidised, whereby ammonium is transformed into nitrate, while acid is produced as a by-product. This acid is partly neutralised by liming, that increases the calcium-carbonate contents. The pH is somewhat lower than most other water types, between 6.5 and 7 (figure 3), and indicates that buffering is not fully achieved. Although in winter the leaching rates are expected to be higher, nitrate concentrations are also relatively high in summer (figure 2). Sulphate concentration levels are also higher than in the softwater type, and are also high relative to seawater diluted by rain (figure 4b). This implies an additional source of sulphate, probably manure. The lowland stream manure polluted water type is mainly found in the sandy areas in the south and east where intensive dairy farming takes place.

Figure 4. Scatterplots of magnesium and chloride average concentration in summer (a) indicating desalinisation in the brackish water type (deviation from the seawater dilution line), and scatterplot of sulphate versus chloride winter (b) indicates surplus sulphate concentration in the effluent in regulated streams, lowland stream domestic and manure polluted, and low brackish water types. This is explained by extra sulphate leaching from respectively sewage, manure and soil sulphide minerals during winter run-off.

Lowland stream domestic polluted water type
This water type resembles the lowland stream manure polluted water type. The main differences with the former manure water type are lower nitrate concentrations, but higher ammonium and phosphate concentrations. This indicates the same processes as were derived from the lowland stream manure polluted water type, but the oxidation of organic matter did not take place to full extent. As this water type occurs near domestic wastewater discharges, the domestic waste water explains the more persistent anaerobic conditions, and consequent high input of phosphate concentrations.

Ground water seepage water type
This water type resembles the soft water type with low concentrations of all parameters, including nutrients. The major difference with the soft water type is that calcium, bicarbonate
and pH are much higher. This indicates rainwater discharge through calcium-carbonate buffered soils and aquifers. This water type seems natural; it is clear water with low nutrient concentrations. The carbonate buffered rainwater is located in the same areas as the soft water type, in the central part near the main rivers and sandy areas of the Netherlands at sites where open water is recharged by ground water. This large influence of ground water is also illustrated by the small differences between summer and winter average concentrations of all parameters.

Peat water type
This water type resembles the lowland stream domestic polluted water type, with the difference that nitrate and oxygen concentration levels are even lower, and the iron concentrations are the highest of all water types, especially in summer. Ammonium and total phosphate concentrations are relatively high, but not the highest of all water types. Chloride concentrations are low, but rise above 100 mg/l in summer. This water mainly consists of rainwater. Relatively high nutrient and low oxygen contents are explained by mineralisation of organic matter in the peat, because the water type is mainly found in Holocene peat and clayey areas in the western and northern coastal regions of the Netherlands.

Effluent in regulated streams water type
The effluent in regulated streams water type is characterised by the highest ammonium and phosphate concentrations (median 4 mgN/l respectively 2 mgP/l) and high nitrate concentrations, comparable to the lowland stream manure polluted water type. This indicates high levels of nutrients that originate from cattle manure. Oxygen concentrations are low, and comparable to the ground water type. The water type is found in the same regions as the lowland stream manure polluted water type and lowland stream domestic polluted water type in the sandy areas of the southern and eastern parts of the Netherlands at known effluent discharge sites.

Rhine River inlet water type
Typical for this water type are chloride concentrations comparable to the River Rhine, especially in summer. Nutrient concentration levels are quite low and the water chemistry indicates calcium-carbonate buffering: calcium and bicarbonate concentrations are high. This water type seems to reflect the highest level of biological activity. The pH values are highest above 8, due to consumption of CO$_2$, oxygen levels are above 10 mg/l, and chlorophyll-a concentrations are high (median above 50 µg/l in winter and above 100 µg/l in summer). The water type is mainly found in Lake IJssel and the Holocene western and northern part of the Netherlands. The Rhine River discharges directly into Lake IJssel, while in the other areas Rhine water is let in during periods of water shortage.

Low brackish water type
This water type is characterised by higher chloride concentrations than in the Rhine River inlet water type. Nutrient concentration levels are relatively high and comparable to the peat water type. The differences between summer and winter concentrations are highest in this water type, although these differences are within a factor 2. Magnesium, potassium and sodium concentrations positively deviate from the seawater dilution line and this indicates dilution by fresh water of an originally marine environment (figure 4a). The adsorption complex of clay minerals was enriched with these cations while marine conditions existed, and exchanged by calcium ions in later fresh water. Excess sulphate concentrations, relative to seawater dilution, indicate an additional sulphate source (figure 4b). This extra source may be dissolved sulphide-minerals from sea clay soils. Higher concentrations of nutrients may be caused by mineralisation of organic matter in marine clays and peat. Oxygen concentrations are low in summer, and are between 5 to 8 mg/l. The brackish water type is mainly found in the near coast regions.

High brackish water type
The high brackish water type has the highest chloride concentrations, above 1,000 mg/l both in summer and winter, and some sites up to 15,000 mg/l. The cations magnesium, potassium and sodium, as well as sulphate, are the highest in the high brackish water type. However, unlike the low brackish water type, the magnesium, potassium and sodium to chloride ratio follows the seawater dilution line (figure 4a and 4b). Nutrients show elevated concentration, but are moderately high and comparable to the peat water type. These nutrients probably have leached from soils rich in organic matter: marine clay and peat. Total phosphate concentrations are relatively high, up to 2 mgP/l, chlorophyll-a concentrations are extremely high, while also bicarbonate concentrations are relatively high, especially in summer; these features indicate
biological activity. Oxygen concentration levels are around 10 mg/l. The high brackish water type is found in the south-western part of coastal areas.

DISCUSSION ON PRACTICAL APPLICATION OF WATER TYPES

The ‘integrated water system approach’ will benefit from the macro-chemical water typology. This typology will provide insight in the spatial variability at different scale levels (national, regional or even within catchments) as it comprises the dominant biogeochemical processes and water sources. In addition the vulnerability of surface water can be assessed, while also prioritisation of water quality measures can be made by use of these methods. Finally the regionalisation of water quality standards can be based on the water typology. A summarising statistic of the water quality parameters per natural water type is calculated and can be used as a guideline for acceptable water quality.

In the Netherlands the spatial variation of water quality is high at every scale. On a national scale level the water quality ranges from brackish to fresh water, while on a local to regional scale the water composition varies by water source (ground water or rainwater), soil type or geology (peat, clay or sand), pollution (domestic or agriculture), and biological activity. The surface waters recharged by recent rainwater and already influenced by pollution in sandy areas may turn out to be the most vulnerable water types, while highly brackish or ground water types may be less vulnerable.

The water types also illustrate that temporal differences between summer and winter concentrations are less pronounced than spatial variations. For instance, nitrate concentrations in the polluted water types (lowland stream manure, domestic and effluent) are always high, in winter somewhat higher than in summer (figure 2). The water types with higher chloride concentration show this dilution effect clearly with lower concentrations in winter, caused by rain. The period of six month that is used in this investigation to average the water composition is too large to discern more detailed effects like those of wet and dry years, shorter dry or wet periods, etc.

RECOMMENDATIONS FOR SURFACE WATER QUALITY MONITORING SYSTEMS

Given the relative high spatial variation the evaluation of monitoring data should be directed more towards spatial variations. As a dense monitoring network with high frequency sites in most cases is financially and logistically unachievable, a balance must be found between spatial and temporal resolution. We proposed in an earlier paper to expand surface water quality monitoring networks with more sites that are less frequently monitored (Frapporti et al., 1999). As an example: if a network operates with 20 sites, that are sampled 12 times a year, we suggested to keep a few high frequency sites operational (e.g. 5 down stream sites to monitor area in- and outlet average). The remaining 15 sites times 12 samples (180 samples) were reallocated over 90 randomly chosen sites, visited once in summer and once in winter. Temporal trends can accurately be monitored in the five remaining high frequency sites, and also by the (summer and winter) yearly average of 90 sites.

Given the non-ideal (STOWA) data set for multivariate statistical analysis (incomplete macro chemical analysis per sample, different types of water bodies) the results are encouraging to use this approach on an improved data set. For this purpose monitoring networks need adjustments, such as routine analysis of all macro-chemical constituents, and relocating or adding monitoring sites in different hydrological setting (both large and small water bodies or streams) to study spatial variations at different scales.

CONCLUSION

New developments in water quality management, the ‘integrated water system approach’, focuses on diffuse pollution and on processes that control the water quality. Macro chemical constituents play a key role to assess these new aspects. Present day surface water quality monitoring networks already include some of these macro chemical constituents, but in most cases at a much lower frequency, while not all macrochemical constituents are analysed. Multivariate statistical techniques are most useful to evaluate the macro chemical constituents and should become a standard evaluation tool in surface water monitoring networks.
We demonstrated one way to evaluate macrochemical composition data on a national scale. From combined regional surface water quality monitoring networks and subsequent cluster analysis of water compositions with fuzzy c-means clustering, clusters with comparable water compositions were defined. These clusters or water types provide a conceptual interpretational framework of the surface water quality in the Netherlands. Nine water types were defined, that range from very fresh (virtually unaffected rainwater) to highly brackish water (seawater). The salinity decreases from clay and peat coastal areas to the sandy inland areas. Within the coastal areas the salinity depends on recharge by ground water, water inlet in summer from the Rhine River and biological processes. In the inland areas domestic or agricultural effluents, water source (ground water or rainwater) and acid buffering by soils or aquifer sediments cause spatial variations. Recognition of these processes in surface water may help to prioritise surface water streams for vulnerability and/or to implement water quality measures.

The emphasis on macro chemistry and the need for a denser spatial coverage of water quality samples also may have impact on the monitoring system design. A shift from relative few sites with high frequency sampling towards more sites with lower frequency is recommended. Subsequently multivariate evaluation techniques are added to the interpretation tools of the monitoring network, in addition to the traditional univariate testing of pollutant against standards.

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WATER MANAGEMENT INFORMATION SYSTEM - A PILOT PROJECT IN THE REGION OF DUESSELDORF

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In the region of Duesseldorf different water authorities and waterworks collect, analyse and store data for the purpose of water management. The various groups have similar information needs and use similar monitoring strategies - but the use of different systems for data handling, analysis and reporting prohibit data exchange between different users.

About one third of the area of the city of Duesseldorf is covered by protection zones for public water supply. Therefore the different users are interested in avoiding unnecessary double efforts in collecting data, inconsistent data and mediaeval data exchange procedures.

In order to develop a system which enables a smooth exchange and use of water management data a study group consisting of three partners (water authority Duesseldorf; waterworks Duesseldorf and Duisburg) and the ahu GmbH conducted in 1997 a pre-feasibility study. The study showed, that internet technology together with technologies like distributed databases are an useful means for solving the problems mentioned above. In 1998 the study group decided to conduct a feasibility study in order to develop a detailed technological concept for an integrated water management information system. The results of this study concerning technological, organisational and legal aspects are the objective of this article.

INTRODUCTION

The city of Duesseldorf is the capital of the federal state of North-Rhine-Westphalia (Germany). A characteristic of this area is the combination of dense habitation are as and large industrial zones. Nevertheless more than 95 % of the public drinking water demand is supplied by water resources which are situated in the entire area of the city of Duesseldorf or very nearby. About one third of the area of the city of Duesseldorf is covered by protection zones for public water supply (see figure 1). The importance of this region for public water supply is therefore evident. The water authority of Duesseldorf and the waterworks Duesseldorf and Duisburg co-operate to conduct several efforts to allow both, the intensive economically usage of the area and the sustainable use of local water resources.

Efficient water management in this region requires a smooth exchange and use of water management data produced by different institutions with different IT -systems (see above). In future the requirements of the water framework directive of the European Union will enforce this need for a dynamic exchange of data between different institutions in a river basin.

Most of the institutions involved in water management have build up their own information systems during the last few years. The different databases contain specific water management data, being collected by each of the institutions. The information (for instance reports, maps, diagrams) produced by each of the different institutions, based upon their own databases, show that in comparison better and detailed information can be produced using a common database, which contains consistent data of different institutions rather than using their own proprietary databases. This problem is well known - therefore several interfaces have been produced in order to exchange data between different database management systems (DBMS). Yet, the physically exchange of data between different DBMS by standard interfaces doesn’t solve the problem which occurs, when data are stored in different logical structures. Unfortunately in most cases data which are stored in physically different DBMS do also have a different logical structure. This is especially true for geographical data, stared in geographic information systems (GIS).
TASKS AND QUESTIONS

Tasks

In the region of Duesseldorf the water authority of the city of Duesseldorf and the waterworks Duesseldorf and Duisburg have built up a study group in order to develop an integrated water management information system (IWIS). IWIS should produce the following benefits:

- Avoiding double efforts in connecting data (saving money).
- Past and smooth access to distributed and consistent data in order to make decisions faster and more transparent (dynamic water management).
- Saving money by using scaleable applications which provide tailor-made functionality to different classes of users.
- Providing the technological framework for the European water framework directive.

Questions

As a first step a pre-feasibility study was conducted in 1997. It showed that there are information technologies which seem to enable an easier exchange and common use of water management data. Therefore from 1998 to 1999 as a second step a feasibility study was conducted, which should try to answer the following questions:

- Which is the technological framework used by the different partners?
- What kind of user-profiles and data-provider-profiles have to be considered?
- What are the suitable technologies required to realise IWIS?
- What kind of organisational framework is the best for IWIS?
Some of the results of the feasibility study will be presented here.

Realisation

On the basis of the feasibility study recently IWIS is to be realised as a pilot project including the above mentioned three partners. After IWIS win have been established between the three partners, and benefits as well as risks of the technology are evaluated, additional institutions producing and/or using water management data in the region of Duesseldorf will be integrated as new users and data providers into IWIS.

TECHNOLOGICAL FRAMEWORK OF THE PARTNERS

The analysis and description of the existing technological framework of the partners is a basic condition in order to develop a practicable concept for IWIS. Therefore

• the relevant business processes concerning connection, handling, storage and transfer of water management data,
• the information systems (software) in use and
• existing as well as planned communication infrastructure for data transfer in a local or wide area network

were analysed and described. The results are typically for the situation of German institutions in the field of water management and can be summarised as follows.

Dataflow

Each of the three partners receive water management data in heterogeneous digital and analogous formats (for instance water table data, hydrochemical data). The data is manually integrated into the proprietary systems. In lack of suitable interfaces a great amount of data, which exists already as digital data in the information systems of one of the partners, has to be integrated manually into the database of another partner.

Information systems in use

Geographical and alphanumerical data are computed and stored in different DBMS (for instance Oracle or DB2), GIS (for instance Smallworld GIS, GeoLIS) and other systems (for instance groundwater modelling system Spring). There are extreme differences between the logical organisation of data in each of the systems. Interfaces exist for data exchange between different systems within the organisations -but there is a lack of suitable interfaces for data exchange between the organisations.

Communication infrastructure for data transfer

Each of the three partners has realised the technical infrastructure for data transfer and communication in a wide area network. The partners can choose between the Internet as communication medium or a virtual private network (VPN) via point-to-point-ISDN. Recently this infrastructure is used for services like electronic mail or internet browsing. Until now an online access to databases containing water management data with local interest is not yet realised.

USER AND DATA PROVIDER PROFILES

The type of data and information which is provided and required by each of the partners as well as the conditions of their usage is also an important framework for realising IWIS. The existing data as well as the requirements of the three partners concerning the data they wish to use, were analysed and described. The results can be summarised as follows.

Existing and required data

Two different profiles of data providers can be distinguished. The two waterworks provide basic water management data like

• core data of observation and production wells (e.g. type, co-ordinates, dimensions),
• observation data of wells (e.g. production rate, water table data, hydrochemical data),
• geological data of aquifers as results of in-situ-measurements or model computations (e.g. layers, permeability).

The water authority provides the same basic water management data plus additional information like
• and use data and data on town-planning in catchment areas,
• legal discharge in groundwater and surface water in catchment areas,
• data on groundwater and/or soil pollution.

The two data provider profiles are complementary to each other. In combination the data of the two data provider profiles allow more sophisticated analysis of water management data than is possible on either one of the data collections.

Conditions of data usage and providing data

The providing of raw water management data (e.g. hydrochemical data) can be entailed with restrictions concerning the following items:
• data protection requirements: data with concern to personal interests (e.g. a groundwater or soil pollution which can be spatially related to a real estate reduces the value of the real estate),
• aspects concerning license agreements (e.g. restricting resale of data) and copyrights,
• aspects concerning the manner how data are interpreted and visualised.

On the basis of these items the following terms of data usage and data providing were elaborated:
• Data users and providers commit on not to violate any copyrights or license agreements.
• Dependent from the user profile data access may be restricted spatially or related to data content. An example: waterworks own two different user profiles, the first one as an owner of real estate and the second one as an enterprise which supplies drinking water. As an owner of real estate waterworks may access data concerning punctual soil and groundwater pollution as well as hydrochemical data derived from monitoring measurements without restrictions in content but spatially restricted to the area of their real estate. In their role as an enterprise, waterworks have access to hydrochemical data which has been derived from monitoring measurements within their catchment area, but will have no access to data from punctual groundwater pollution.
• Data users commit on a good practice concerning the interpretation and visualisation of data. For instance: hydrochemical data has to be interpreted and visualised within their spatial and chronological context. The membership of single observation points within observation grids has to be taken into account (e.g. data derived from local observation grids which are set up to control the remediation of a groundwater pollution cannot be combined with data derived from regional observation grids set up in order to monitor the long term development of groundwater quality).
• Data providers guarantee the quality assurance of data and commit on agreements for defined data update frequencies. They commit on the documentation of the state of quality assurance and will build up a defined stock of meta-information about their data.

IWIS TECHNOLOGY

Based upon the user and data provider profiles as well as the technological framework of the three partners a technological concept for IWIS was developed. The following items provide the framework for the technological concept of IWIS:
• Existing systems to collect, handle, analyse and visualise water management data (e.g. GIS, DBMS, modelling systems) will be maintained.
• Every partner provides geographical and alphanumerical data.
• The existing communication infrastructure will be maintained and used.
• Client access to water management data provided by IWIS will be enabled by means of freeware internet browsers.
• Provided data (geographical as well as alphanumerical data) have to be integrated seamless into the systems in use (see above).
Within the feasibility study two different technical concepts are developed for IWIS. The first concept can be accomplished in short-term, the second one describes a future prospect, taking the future development of the IT-systems of the three partners into account.

**Concept #1 -interface orientated solution**

Figure 2 shows the principle of concept #1. It is based upon an interface orientated technology. There are three different roles and five application layers. The different roles are

- the data provider: the three partners as the producers or owners of water management data.
- the service provider: third party which operates the server and provides consistent data and functionality as well as rules for access control.
- the user: the three partners as the users of water management data, consisting of the data of all partners (data providers).

The data providers collect water management data with their own proprietary systems. Geographical and alphanumerical data, which are to be provided by IWIS, are transferred by automated interface and import services within two steps to the service provider (see figure 2, thin arrows):

1. Within the **interface layer** XML is supported as a standard exchange format for alphanumerical data, while MapInfo interchange format and DXF-format is supported to exchange geographical data.
2. Within the **data import service layer** the data in standard exchange format is imported into the database of the service provider. An important feature of the import service layer is the assignment of the different logical data models of the data providers to the logical data model of the service provider.

The service provider makes the complete stock of water management data of all data providers available in a consistent manner. Both geographical as well as alphanumerical data are stored as records in an object-relational database management system. This enables a direct logical link between data, user profiles and user accounts and or trustees. Moreover the service provider offers services e.g. enabling access to data and supplying additional functionality via http.

User access to the service provider is accomplished by a simple freeware internet browser with access to the internet or a VPN. After authentication of the user the internet browser is completed by user profile specific applets, providing different functionality for data exploration or visualisation e.g. by means of forms, reports, diagrams or maps. Using the import service layer and the interface layer in the opposite direction enables the user to export data into his own proprietary applications (see figure 2, thick arrows).
Concept #2 - knowledge based solution

Figure 3 shows the principle of concept #2. It is based upon a knowledge based technology. As an important, but recently not yet realised, framework for concept #2 it is assumed, that in future the common storage of geographical and alphanumerical data in standard DBMS is accomplished (recently most of the big GIS- and DBMS-producers are involved in the OpenGIS-Consortium OGC, where common standards for the storage of geographical data in DBMS are defined).

In comparison to concept #1 the interface and import service layers are replaced by a sophisticated set of rules providing a direct user access to the data provider via standard database communication protocol. The service provider offers the required services for authentication and access control as well as servlets and applets, which provide additional user profile specific functionality (see above). Therefore concept #2 does not need a redundancy of data as in concept #1 is still necessary. The service provider’s tasks are reduced to connecting users to data providers and controlling authentication and user profiles.

ORGANISATIONAL AND LEGAL ASPECTS

In prospect to the future operation of IWIS there are some important items concerning the organisational relations between the above mentioned roles (user, data provider, service provider) and legal questions.

Organisational aspects

There are three possible organisational forms to operate IWIS. First of all, one of the three partners can take the role of the service provider. As a second possibility the three partners can establish an utility for the service providing. As a third alternative, comparable to the second, the role of the service provider can be taken over by a third party. All mentioned alternatives have different advantages and disadvantages. For instance: one disadvantage of the first alternative is, that technical service providing is of none of the three partners the core competence. As an advantage of the second alternative the complete control over data remains at all three partners. The third alternative provides an optimum concerning the technical competence of a third party provider.

Legal questions

Within the feasibility study relevant legal questions were collected and documented. The relevant questions touch the following themes:

- Data security concerning data with regard to personal rights.
• License agreements and copyrights.
• Law of contract.

As a next step in the project the legal questions will be answered by a legal adviser.

CONCLUSIONS

The feasibility study has led to the following conclusions:
• The results of the study prove that the technical concept provides suitable means in order to accomplish the requirements of the water framework directive of the European Union concerning the need for a dynamic exchange of data between different institutions in a river basin. Recent years most water authorities and water suppliers collect and manage data within a specific spatial framework which depends not on the spatial extent of river basins but on their special tasks (e.g. the tasks of the water authority of Duesseldorf are related to the spatial extent of the area of the city of Duesseldorf). IWIS provides the means to merge data from different databases in order to build up a consistent database for river basin management.
• Furthermore the study shows, that IWIS is a powerful tool in order to avoid double efforts in monitoring measurements. In combination with IWIS’s strength in optimising data and information as a basis for decision making processes it supports users in realising a tailor-made monitoring.
• IWIS will be realised as a pilot project on the basis of the technological concept #1 (interface orientated solution). The technological concept #2 is an appropriate prospect to the future.
• IWIS will be operated on the basis of the third alternative (third party as service provider).
• After the implementation of the system and an evaluation of benefits and restrictions additional users and data providers will be integrated within IWIS.
Without a recognized, focused effort to ensure consistent and comparable water quality data/information, there is considerable redundancy and inefficiency in water assessments. The intent of this paper is to outline a voluntary effort that has been initiated in the United States to begin the process of providing relevant, comparable hydrologic information. The National Water Quality Monitoring Council (Council) was created in 1997 to provide a national forum to coordinate consistent and scientifically defensible water quality monitoring methods and strategies. The work being done by the Council has considerable relevancy to international water quality data and management issues. The strategies being used in the United States can be used to enhance international cooperation in order to optimize monitoring efficiency and data/information sharing.

INTRODUCTION

Water is the core consideration for almost all of the global environmental, economic and human health issues. However water is not, in general, managed in an integrated, international manner. Consistent, long-term data sets that are comparable are necessary in order to formulate ideas regarding regional and global trends in water quantity and quality. Just as important, comparable methods of analysis and interpretation are necessary in order to generate water quality information that has value and relevance to the management community and the public.

Ward (1966) points out that water-quality monitoring and its attendant data analysis processes, are the primary means through which information about water is developed. However, the manner in which monitoring is conducted and data analyzed has been the subject of considerable discussion, debate and criticism over the years.

Watersheds are boundaries shaped by nature. When a watershed approach is used for managing a region’s water resources, the land is divided into units that reflect these natural boundaries, regardless of whether or not they incorporate several different political boundaries. Watershed boundaries, and the complex set of natural and human interactions that take place within them, are then used as the basis for studying and managing water resources. However, in the absence of an organized approach to assure consistent data and methods for assessing transboundary water resources to ensure comparable information, contention and bitter friction may ensue. Clearly if there is an organized effort to overcome these data and information deficiencies, binational watersheds can be the arena in which forums and collaborations take place to ensure common goals and pathways to achieve these goal. Consistent, transboundary watershed based water monitoring and data analysis methods enable the development of pathways that achieve a preferred environmental state and quality of life.

Early Hawaiian’s recognized the value of a watershed approach to the quality of their lives. The Ahupua’a management system is documented as early as 1879, and can best be described in the following quote:

Ahupua’a – “a resource management system. A principle very largely obtaining in these divisions of territory was that a land should run from the sea to the mountains, thus affording to the chief and his people a fishery residence at the warm seaside, together with the products of the high lands such as fuel, canoe timber, mountain birds, and the right of way to the same, and all the varied products of the intermediate land as might be suitable to the soil and climate of the different altitudes from sea soil to mountainside or top”

Today, water quality practitioners look at rivers as the environmental indicators of the condition of the landscape, an integration of impacts and effects from the entire watershed. There is again
the realization that a watershed runs from the mountains to the sea, and the data we collect can lead to scientific insights on the impacts of climate variability, land use practices, and water use on important environmental and economic variables such as water availability and quality, floods and drought severity, and the health of aquatic and riparian biological communities.

Because rivers integrate multiple landscape impacts, and involve many decision-makers rather than one chief, they are often the focus of resource conflicts. In river basins world wide, water management has evolved in response to the changing demands that various governments and societies have placed on that river. Thus the conflicts are diverse and variable. Watersheds become the focus for debate and litigation rather than the forum for collaboration and cooperation. As watersheds develop, the quality of the water is increasingly of concern to the individual management agencies and the demand for water quality information climbs dramatically. If the concerns are also binational, the added complexity of dealing between and with other governments is exponential. Adjacent countries that share a major watershed(s) often experience evolving management systems that are not comparable, water information systems are not comparable, and water data generation procedures are not comparable. Even within individual countries, these inequities exist.

Thus, without a recognized, focused effort on the part of the impacted entities to ensure consistent and comparable water quality data/information, there is considerable redundancy and inefficiency in water assessments. The intent of this paper is to outline a voluntary effort that has been initiated in the United States to begin the process of providing relevant, comparable hydrologic information. The process is not easy; it requires a change of thinking, and the cooperation of many levels of water quality practitioners. Such a collaborative process applied on a global scale could easily save developing countries millions of dollars as well as provide the knowledge to better understand global hydrology.

**WHAT IS THE NATIONAL WATER QUALITY MONITORING COUNCIL?**

True collaboration among programs is possible if there is both the technical and the institutional framework to promote data comparability and data of known quality (ITFM, 1995a). In order to promote this philosophy, the U.S. National Water Quality Monitoring Council (Council) was created in 1997 to provide a national forum to coordinate consistent and scientifically defensible water quality monitoring methods and strategies. The Council was formed as the permanent successor to the Intergovernmental Task Force on Monitoring Water Quality (ITFM, 1992-1995). The Council is comprised of a balanced membership of 35 representatives from Federal, interstate, state, tribal, local and municipal governments, watershed groups, universities, and the private sector, including volunteer monitoring. The Council provides the major national forum for the coordination of consistent and scientifically defensible federal and state water quality monitoring methods and strategies. Such strategies are intended to improve understanding of different impacts, such as polluted runoff and habitat alteration, on water quality and to define a national agenda of needed monitoring, research, and assessment models and tools. The Council is co-chaired by the U.S. Geological Survey (the major U.S. Government water data collection agency) and the U.S. Environmental Protection Agency (a major U.S. Government water quality regulatory agency). The Council is chartered under the Federal Advisory Committee Act and meets a minimum of three times a year at locations throughout the country.

**Challenge to the Council**

Monitoring is a more important element of management than ever before. In the United States, beginning in 1972, progress in water quality restoration under the Clean Water Act was accomplished by controlling the relatively easily identified and regulated sources of point-source pollution. However, today these sources comprise a relatively small percentage of the total pollution load to America’s waters. Recent inventories of the causes of degraded waters identify sources that result from land uses and management practices known as non-point sources, those that discharge intermittently and are dispersed across the landscape. They contribute silt, bacteria, and elevated temperatures and nutrients. They are not easily regulated. However because of monitoring, the subtlety and extent of these impacts is better understood. Monitoring is essential to identify these sources, prove a further understanding of their impacts, and guide control efforts. Monitoring ultimately proves the value of the controls that are implemented.
The Clean Water Action Plan (CWAP) (1998) provides a blueprint for restoring and protecting the nation's water resources. Significantly, the plan calls for a cooperative approach to watershed protection in which state, tribal, federal, and local governments as well as the public identify those watersheds with the most critical problems and then work together to focus resources and implement effective strategies to solve those problems. The Clean Water Action Plan assigned to the Council the responsibility for several of the key action items that relate to guidance on water quality monitoring.

In addition, the Council has designated key priority tasks/goals for itself following the recommendations of the early work accomplished by the ITFM and the known need by Council members for a national forum to coordinate a national process that provides a foundation for data and information consistency. Obviously some of the Council's priority tasks/goals and the CWAP assignments overlap since there is broad recognition of the need for this process to focus on collaboration and comparability. The goals have been developed with accompanying objectives and specific tasks, to meet important program needs, with special emphasis on supporting the needs of watershed management. It is the hope of the Council that the resulting assessments, summaries, and guidance that evolves, provide fundamental information useful to the monitoring community at many levels and at the same time addresses many of the needs for environmental restoration as defined in the CWAP.

Though progress is being made within the United States, considerable institutional and political barriers, including data and information remain. Inherently, when watersheds cross local, state, tribal and especially, national jurisdictions, a host of political, policy, resource, and budgetary complications come into play. Researchers and practitioners alike can benefit from the recommendations evolving from the Council strategies in order to help form new partners, to become more fully educated and build coalitions of institutional support to minimize or avoid the possibility of conflicts over water resources.

COUNCIL APPROACH

The Council promotes an integrated watershed management approach to address water issues and related natural resource concerns. In many cases, existing watershed management and monitoring strategies are not systematically designed in an integrated fashion and thus are not capable of addressing water quality impairments and related natural resource threats that originate upstream or culminate downstream from diverse activities throughout the watershed. However, it is an objective of the Council, working in cooperation with a wide number of organizations such as the Universities Council of Water Resources, and the National Institutes for Water Resources, to encourage monitoring system design and operation that provides comparable data and information throughout a watershed. Such information permits a more integrated approach to watershed management than is now feasible.

The Council's efforts are designed to provide information needed for program implementation at many levels and help foster local watershed/state water quality monitoring council organizational partnerships. The national goals of the council are formulated and carried out through Council meetings and work groups. Results are shared at biennial conferences, local and/or state councils, fact sheets, web sites, CWAP results reporting, and publications. The conferences and other meeting forums are designed to provide feedback from constituents and stakeholders. This feedback is in turn used to advance the goals of the Council. Local organizations likewise benefit from these reality checks and adjust accordingly, based on their own local organizational needs and priorities.

The following goals, associated objectives, and tasks were developed by the Council work groups to support important needs of watershed management. Validation of these goals has been accomplished at two national conferences sponsored by the Council. The work of the Council and the guidance that evolves from the accomplishment of the following tasks will help guide the monitoring community toward a more cohesive approach to monitoring and to collect data using comparable methods.

- Water Information Strategies

After a water quality law is passed and is being implemented, the question is eventually asked, "Are we achieving the water quality goal(s)?" The 'goals' in the minds of policy
makers are generally those at the beginning of the law, such as "providing for the protection and propagation of fish, shellfish, and wildlife, and for recreation in and on the water….wherever attainable". The 'goals' in the minds of technical staff, are generally those associated with implementation of a specific program in their locality or area of influence. As the law is implemented and amended other, often more detailed questions are asked in order to guide program implementation and to adjust approaches. If the monitoring system has not been designed to meet these information goals, it is unlikely that a satisfactory answer will be abstracted from existing water quality data.

**Approach**

The Water Information Strategies Work Group is working to provide a framework and forum for defining and developing a more information goal-oriented approach to monitoring system design. The framework will permit a direct connection between information needs/expectations of water quality managers and the public and the ability of monitoring to produce information.

The approach focuses on bringing experts together with work group members to

1. Define and specify water information goals;
2. Benchmark current monitoring system design efforts;
3. Coordinate with other groups working toward similar objectives (such as the efforts of this conference, other European efforts, and the information focused monitoring system design efforts in New Zealand and Australia); and
4. Develop guidance for the monitoring community.

**Methods and Data Comparability**

The Methods and Data Comparability Board was established by the Council to promote and coordinate voluntary participation of the monitoring community in the use of collection and analysis methods that produce water-quality-monitoring data of known and documented quality. Methods used for collection and analyses of physical, chemical, biological, and microbiological data from surface water, ground water and the unsaturated zone must include sufficient meta-data and quality assurance and control information to allow comparison of data sets collected and analyzed by different protocols.

**Approach**

The Methods Board is providing a framework and a forum for comparing, evaluating, and promoting methods that produce data that can be compared among water quality monitoring programs. This approach will focus on bringing experts together with Board members to define and specify goals, benchmark current associated efforts, coordinate with other groups working toward similar objectives, and develop guidance for the monitoring community. Where appropriate and feasible, pilot studies and work agreements with the private sector and other collaborative efforts will be encouraged to enhance communication and use of comparable monitoring methods.

**Institutional Collaboration**

Many water quality monitoring organizations are needed to collect and distribute data and information required for informal decision-making. Collaboration between institutions, at all levels, is essential as no single agency can afford to gather the diverse information needed for informed decision-making. Through coordination and collaboration, integration of the many organization efforts can be accomplished by establishing partnerships of multi-organizational efforts at numerous levels. The Council feels very strongly about the importance of this goal and believes that regional, state, or local partnerships are particularly critical to the Council’s success.

**Approach**

The Council encourages and supports the development of regional and state monitoring councils with representation from appropriate stakeholders. A major emphasis of these councils is inter-and intra-state coordination and collaboration. For example, the Council
has participated in and supported the development of regional and state monitoring Councils such as in Texas, Maryland, Oklahoma, Virginia, and Colorado. In addition, efforts are underway to establish a Lake Michigan Council and an Upper Mississippi River Council.

In order to continue to encourage the development of these groups, the Council plans to publish an information circular outlining the benefits of such councils, and provide a blueprint for their formation and operation. This will be accomplished with Web links, clearinghouse referrals, newsletter stories, fact sheets, or information brochures, in order to encourage and connect those pursuing collaborative programs with others who have successfully attained them.

- **Data Management**

  The Council believes it is necessary to promote data and information sharing between the various entities involved in the water quality monitoring community by providing guidance on common reference tables and data standards.

  **Approach**

  Many of the issues and tasks regarding the availability and management of data are at this time being coordinated in the ongoing work elements of goals 1, 2, and 3. For example, information strategies (goal 1) likewise provides the forum for the development of data management processes that provide consistency over a wide spectrum of water quality practitioners. The objectives of the work taking place in goal group 2 (methods and data comparability) relate to the development and recommendation for a consensus set of core water quality data elements (information about the data) that would allow data to be compared regardless of, but recognizing, the purpose of the monitoring activity. Institutional collaboration (goal group 3) is an ideal forum in which to inventory the available database reference tables and data base standards, and determine their currency, adequacy, completeness, geographic coverage, degree of use, and degree of support and maintenance by the caretaker organization.

- **Public Awareness and Stakeholder Outreach**

  The early work by the ITFM recognized the need to increase the level of public awareness and stakeholder involvement in water quality monitoring issues in order to meet the full range of requirements for society. Polls conducted in 1995, and 1997, and an ever-increasing number of grassroots watershed activities, clearly show that the American people are concerned about water quality and water pollution were identified as one of the most significant environmental issues. The Clean Water Action Plan recognized the need to bridge the gap between science and society, and thus requires agencies to expand efforts to facilitate the sharing of monitoring information among stakeholders of all kinds.

  **Approach**

  The Council agrees it is important to promote public awareness of the field of water quality monitoring and educate stakeholders on water monitoring issues. Thus this workgroup will strive to communicate Council activities and products to a side group of interested people and organizations. The use of educational web sites such as http://water.usgs.gov/wicp/, and http://wi.water.usgs.gov/pmethods/, fact sheets, posters, and conference presentations are being used to disseminate the findings, guidance, products, and recommendations of the Council. In addition, the workgroup will facilitate the linking of other relevant websites and provide a clearinghouse of information as appropriate.

- **Monitoring Interactions Among Watershed Components**

  Within the context of watershed coalitions, the Council recognizes the need for coordination of consistent data for all aspects of the hydrologic cycle in order to generate the holistic understanding to make responsible water quality management decisions. It thus follows that the role of the Council is to provide forums in which the tools and recommendations necessary to improve and enhance coordination and the use of consistent
and scientifically defensible methods and strategies for the monitoring of the various watershed hydrologic components. In addition, the increased understanding of the interactions between ground-water/surface-water and the significance of the coastal watersheds will be emphasized.

INTERNATIONAL SIGNIFICANCE

The work being conducted by the Council has relevance to international water quality data and management issues. The need for water-quality information and conditions on a global basis are not new, and they are increasing. As various earth summits deal with issues of ecosystem sustainability while trying to maintain new levels of environmental protection and maintain or increase the economic infrastructure, the need for comparable water-quality information on a binational/international basis will continue to increase. Borders of ecosystems don’t coincide with administrative or political boundaries. Often, sources of one country’s water supply, are upstream countries, a condition that often leads to disagreements. International agreements, such as the Helsinki Convention on the Protection and Use of Transboundary Watercourses and International Lakes, are thus very important (de Jong and Timmerman, 1996). United Nations Global Environmental Monitoring Systems have been collecting water-quality data for many years and now produce technical assessments of global water-quality conditions. But these efforts are not holistic, and would be of much greater benefit if there were controls and efforts in place to ensure joint monitoring and consistent data and information that is fed into a comparable and accessible database.

New Zealand has designed and operates a national water-quality monitoring program that is being used to track and document national water-quality issues. This network of 77 surface water-monitoring sites has been operated consistently over a 10-year period and is known for its high quality and accessibility. The data set is accompanied by meta-data that describes the monitoring sites, how the samples are collected and analyzed, and all other ancillary data that is of use to data analysts. (Martin, L.M., 1999, Smith et al., 1996).

Many of the European Countries appear to have implemented data networks that are designed to meet both internal and external needs. Buzas (1996) describes agreements with 6 of the 7 countries that surround Hungary utilizing Joint Commissions on Transboundary Waters. The agreements, which are staged, are designed on a binational watershed basis in order to deal with water quality questions and related ground water issues.

In negotiations between the United States and Mexico regarding water-quality monitoring and data comparability, the work of the Council has been the starting point for binational deliberations to draft an interagency bilingual water-quality field manual. Along the U.S.-Mexico border, numerous federal, state, and local agencies, nongovernmental organizations, and academics, collect water-quality data for a variety of purposes, including assessing transboundary water resources. The water community uses a variety of documented and nondocumented procedures, some of which have specific data-quality and data-information objectives. This mix of procedures results in uncertainties by data users as to data validity and quality that in turn limits the uses of the data by the two countries in treaty-related negotiations and decisions.

The results of the effort to develop a binational field methods manual will have direct applications not only across state boundaries, but also specifically for the international reach of the Rio Grande where language and jurisdictional issues are especially complicated.

The proceedings from Monitoring Tailor-Made II (MTM-II) (1996) contain numerous papers and arguments in favor of integrated disciplines and international cooperation and participation in order to optimize monitoring and data/information sharing. Jong (1996) states that while collective alliances exist, the point is to make these effective. Guidelines, directives, or codes of practice are needed to implement cooperation. It is hoped by the authors of this paper that the strategies being used in the United States by the Council have relevance to the issues raised in MTM-II and can be incorporated in some fashion to achieve similar goals.

As we have determined in the United States, there is no one solution, but numerous methods, that if combined and integrated through forums and collaborative partnerships can lead to comparability of information and agreement.
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UNITED STATES WATER QUALITY METHODS AND DATA COMPARABILITY BOARD: CREATING A FRAMEWORK FOR COLLABORATION AND COMPARABILITY

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Each year, government agencies, industry, academic researchers, and a wide variety of private organizations in the United States devote enormous time and several billion dollars to the monitoring, protection, and restoration of water resources and watersheds. Critical differences in project design, methods, data analysis, and data management have often made it difficult for monitoring information to be shared by other potential data users. In the absence of a focused effort to ensure consistent and comparable data, and a means to determine the utility of a data set for a particular use, there is considerable redundancy and inefficiency in water assessments. The Methods and Data Comparability Board is a partnership of water quality experts from federal agencies, states, tribes, municipalities, industry, academia, and private organizations. It is chartered under the National Water Quality Monitoring Council, whose mission is to coordinate, and provide guidance on, implementation of the voluntary, integrated, nationwide monitoring strategy developed by its predecessor - the Intergovernmental Task Force on Water Quality Monitoring (ITFM, 1995a). The Board provides the framework and the forum for comparing, evaluating, and promoting monitoring approaches that can be implemented in all appropriate water quality monitoring programs.

INTRODUCTION

Each year, government agencies (local, state, tribal, and federal), industry, academic researchers, and a wide variety of private organizations in the United States devote enormous amounts of time and several billion dollars to the monitoring, protection, and restoration of water resources and watersheds.

This work includes:
- monitoring the status and trends in water quality
- identifying existing and emerging water-quality issues
- designing and implementing resource management programs
- determining compliance with regulatory programs

The information gathered through these activities is certainly useful to the data collectors themselves, however, critical differences in project design, methods, data analysis, and data management often make it difficult for monitoring information to be shared by other potential data users. Accurate, cost-effective and efficient assessment of the nation’s water resources—within and among watersheds—requires that monitoring entities work collaboratively and strive for comparability in methods and data management. The design and implementation of assessment and management programs should be a cooperative product of the various monitoring agencies and organizations active in any given watershed. Knowing whether the water quality monitoring methods used and the data they produce are comparable is essential to achieving the goals of the Clean Water Action Plan (USEPA/USDA, 1998).

METHODS AND DATA COMPARABILITY FRAMEWORK

Producing comparable data is dependent on the use of a coordinated monitoring framework that includes four primary elements: developing data and measurement quality objectives for each monitoring activity, determining the performance of sample collection protocols, determining the performance of field and laboratory methods, and utilizing a standard data reporting approach to provide methods information (figure 1). The National Method and Data Comparability Board (Board) activities described in this report have been developed to address specific method and data comparability issues for each of the primary elements of the overall
monitoring process. The Board’s strategic philosophy is to complete ongoing products in the short term, while continuing to develop its longer-term goals and objectives. The short-term emphasis is to marshal Board resources to successfully complete projects, so that accomplishments and success stories can be demonstrated. A longer-term view includes the development of tools for the monitoring community through the use of pilot tests and other studies.

![Methods and Data Comparability Framework](image)

**DQO/MQO Development**

Data Quality Objectives (DQOs) and Measurement Quality Objectives (MQOs) are or should be the foundation of all monitoring studies, as these define the questions needing answers and the data quality needed to answer those questions (USEPA 1994; ITFM 1995b; MDCB 1999a and b). DQOs are statements that define the confidence required in conclusions drawn from data produced by a project (USEPA 1994). The USEPA’s DQO process is a 7-step strategic planning approach that is used to define what, how, when, and where data are collected and analyzed to ensure that the type, quantity, and quality of environmental data used in decision making will be appropriate for the intended application (USEPA 1994). For example, the USEPA’s Office of Ground Water and Drinking Water used the DQO process to help ensure that water quality measurement data and engineering information gathered under their Information Collection Rule (ICR) were adequate to support development of a series of drinking water regulations regarding surface water treatment requirements and disinfectant and disinfection byproduct controls (RTI, 1995). MQOs are statements that contain specific units of measure such as percent recovery, percent relative standard deviation, or detection level. They should be thoroughly specified to allow specific comparisons of data to a MQO.

**Field Sampling and Monitoring**

Monitoring data are obtained by field sampling, direct field measurements, and remote sensing. The Board recognizes that field method performance is an area in need of attention, as sampling-induced error or biases can be larger than those associated with laboratory analysis (ITFM 1995c). One of the key concepts inherent in the Board’s mission is the use of a performance-based system (PBMS) approach. The Board defines a PBMS as a framework that permits the use of any appropriate sampling or analytical technology that demonstrates the ability to meet established performance criteria and complies with specified DQOs and MQOs of the project.
Laboratory Analysis

Most monitoring samples are currently analyzed using prescriptive methods. The widespread use of prescriptive methods will undoubtedly continue in the future, however, a performance-based systems approach, if effectively designed and implemented, could provide the framework for ensuring better, faster, safer, or less expensive analytical methods that provide comparable data. A PBMS approach recognizes the need for well-defined DQOs and MQOs, an adequate supply of reference materials, the need for validated or reference methods that meet the stated MQOs, and the need for adequate training. Additionally, to ensure comparable data of known quality, it is critical to standardize laboratory quality systems. Standardization of quality systems is particularly important for nation-wide monitoring programs in which data from a variety of organizations are used to make large-scale assessments. A key component for ensuring quality systems is third party accreditation of laboratories that provide environmental data. Accreditation is defined as a formal recognition of competence that a laboratory can perform specific tests, or types of tests.

Reporting Data to Include Methods Information

The ability for a user to judge the comparability of monitoring data is dependent on the availability of information about which methods were used to obtain the data, information about the performance of those methods, and the inclusion of metadata and data quality documentation reported with the data. The Board recognizes that better documentation of existing methods, and more complete and consistent metadata reporting by monitoring organizations, will improve our ability to define and use comparable data for a variety of purposes.

METHODS AND DATA COMPARABILITY BOARD APPROACH

The Board includes 5 federal, 5 state or tribal, and 5 private sector delegates and an equal number of alternates from each of those sectors. The Board accomplishes its objectives through Workgroups that include Board delegates, alternates and other volunteer experts in the water-quality field. These Workgroups define and specify goals, benchmark associated efforts, coordinate with other groups working toward similar objectives, and develop guidance for the monitoring community. Where appropriate and feasible, pilot studies and work agreements with the private sector and other collaborative efforts are encouraged to enhance communication and use of comparable monitoring methods.

Seven Workgroups have been developed to focus on specific methods and data comparability issues. Some of the Workgroups have a more inherent product focus (NEMI, WQDEs, Outreach), others have both a product and a process focus (Accreditation, PBMS), while still others have a primarily discipline-based focus (Nutrients, Biology). The discipline focused Workgroups are also involved in product development, however, much of their effort will go towards providing information to the product and process focused Workgroups. The Outreach workgroup integrates across all work groups using a liaison member approach. In addition to these more formalized Workgroups the Board has also been available to review and participate in the development of other related technical activities in a more ad hoc fashion.

The Board meets quarterly, however, the Workgroups conduct much of their business between the Board meetings via conference calls and electronic mail. A steering committee meets via conference call between Board meetings to coordinate funding, conference calls, integration efforts, and to provide overall focus and direction to the Workgroups. The full Board meets via conference calls between Board meetings to provide progress reports on product task force efforts and to discuss operational issues.

Current Workgroups and their specific objectives are:

Performance Based Systems (PBMS)

Inform the Board and the National Water Quality Monitoring Council (Council) on technical matters pertaining to the implementation of a PBMS. Define the dimensions of a PBMS for field chemical, microbiological, and biological protocols and laboratory analyses and prepare
guidelines to implement PBMS in ambient and compliance monitoring. The workgroup provides peer review of and coordination with PBMS development efforts being undertaken by other organizations.

**National Environmental Methods Index (NEMI)**

Provide a web-based searchable compendium containing chemical, physical, radiochemical, microbiological and biological field and laboratory methods, including protocol summaries and information. It will allow the rapid communication and comparison of critical parameters of methods for use with methods selection and (or) methods modification and data comparability.

**Laboratory and Field Accreditation**

Promote laboratory accreditation and develop a Board position on federal laboratory accreditation and pre-laboratory certification in order to establish a uniform national accreditation process including the use of performance-based systems (PBMS). Coordinate the accreditation related activities of the various Board workgroups and communicate those efforts to the corresponding workgroups in the National Environmental Laboratory Accreditation Conference (NELAC) in order to avoid duplication of effort. NELAC’s purpose is to establish and promote mutually acceptable performance standards for the operation of environmental laboratories.

**Water Quality Data Elements (WQDE)**

Develop and recommend a "core" set of data elements for reporting water quality monitoring results, to be voluntarily implemented, that would allow data to be compared regardless of, but recognizing, the purpose of the monitoring activity.

**Biology Measurement**

Identify, compile and develop a framework for characterizing and comparing water monitoring efforts that diagnose environmental conditions using either: a) whole organisms; b) biomolecular materials; c) populations in the field; or d) microbiology. Develop pilot tests of the PBMS position guidelines and information for the NEMI database.

**Nutrient Measurement**

Identify, compile, and develop a framework for characterizing and comparing nutrient methods and data. Develop nutrient pilot tests of the PBMS position guidelines and information for the NEMI database. Coordinate these efforts with the EPA regional nutrient criteria strategy.

**Outreach and Publicity**

Develop and implement means to inform and solicit input from the water resources community regarding the efforts of the Board. Employ various media, including: the internet, brochures, fact sheets, reports, posters, and conference presentations. Develop methods comparability sessions and workshops for the biannual National Water Quality Monitoring Conference. Develop a coordinated outreach approach for the Board Workgroups and with the Council.

**COMPLETED BOARD EFFORTS**

**PBMS Workgroup**

One of the key concepts inherent in the Board’s mission is the use of a performance-based system approach. Key aspects of a PBMS were identified in an issue paper developed by the Board entitled "Towards a Definition of a Performance Based Approach to Laboratory Methods."
Important elements include: a) the need to establish concise measurement quality objectives (MQOs) and data quality objectives (DQOs) for each parameter reported; b) the need for demonstrated methods capable of meeting these MQOs or DQOs; c) the need for adequate reference materials to assist laboratories in demonstrating the appropriateness of a given method (prescriptive or performance-based); d) the need for laboratories to adequately document method performance, and e) the successful completion of a pilot program to demonstrate the advantages and viability of a performance-based approach. The paper addresses the issues and concerns regarding the use of PBMS, and defines the dimensions of a PBMS – focusing on laboratory, but also addressing field aspects. It includes chemical, microbiological and biological protocols used in compliance and ambient monitoring programs. The Board believes that setting data-quality objectives and using PBMS to meet these objectives will promote and enhance innovative technologies without compromising data quality. Data users can determine data comparability and make their own decisions concerning the applicability of data to fit a particular need. Performance criteria, such as precision, bias, sensitivity, specificity, and detection limits, used in conjunction with reference methods, will be integral to the implementation of PBMS and will provide the user with "judgment" factors for decision-making. If implemented successfully, the use of PBMS is a move towards the production of valid, scientific procedures and evaluations, and away from a prescriptive regulatory mandate.

NEMI Workgroup

The National Environmental Methods Index (NEMI) will ensure that the consideration of analytic methods is a more active part of the planning and implementation of monitoring programs. NEMI will include data fields for comparison such as, relative cost, sample preparation, instrumentation required, method detection level, sampling information, sample preservation and storage conditions, bias, precision, and other QA/QC requirements. Typical users of NEMI are expected to include regulators, regulated parties, scientists, volunteer monitoring groups, and watershed planning organizations. The Workgroup developed a list of critical methods parameters that relate to analytes, instrumentation, matrices, interference, sampling, sample handling, and data quality. These critical parameters will be developed for a list of 100 methods that were chosen to represent a wide range of organizations as well as method type.

Nutrients Workgroup

The Board completed a study to determine if the information available on nutrient methods from four sources (APHA 1995, ASTM 1989 and 1999, and USGS (Fishman), 1993) were sufficient to populate the NEMI data fields. Descriptive information for 25 desired NEMI fields was compiled from 26 nutrient methods from these 4 sources. Findings included: Information is readily available for 15 of the fields, information is predominantly not available for 6 fields, and information availability for the other 4 fields is variable.

Outreach and Publicity Workgroup

The Board developed sessions and workshops at each of the first two National Water Quality Monitoring Conferences - Monitoring: Critical Foundations to Protect Our Waters (NWQMC, 1998) and Monitoring for the Millennium (NWQMC, 2000). Sessions included: QA/QC for monitoring programs; data comparability and collection methods; and inorganic, organic, and biological methods comparability. Workshops were held on PBMS and Water Quality Data Elements. Additionally, as a part of its outreach effort, the Board has developed and published two Fact Sheets, entitled "The National Methods and Data Comparability Board: Collaboration and Comparability" (MDCB, 1998) and "Why is a National Environmental Methods Index Needed?" (MDCB, 2000a); established an internet site (MDCB, 1999b); prepared a portable display that provides an overview of the Boards efforts (MDCB, 2000b); prepared articles for a variety of newsletters; and made presentations and prepared papers (Brass 1998, 1999, 2000) for numerous conferences and meetings.

The Board provided oversight and guidance for the development and implementation of methods comparability projects. These projects included: "A Comparison of Water-Quality
Sample Collection Methods Used by the U.S. Geological Survey and the Wisconsin Department of Natural Resources (Kammerer, 1998); "A Reconnaissance for Sulfonlurea Herbicides in Waters of the Midwestern USA" (Battaglin, 1998a and 1998b, Scribner 1998); and a review of the USGS-EPA Drinking Water Initiative (Patterson, 1997).

CURRENT BOARD EFFORTS

PBMS Workgroup
One of the technical challenges in implementing a PBMS is defining a rigorous, yet cost-effective framework by which laboratories can document that method performance has been achieved. To address this challenge, the MDCB has designed and is coordinating a pilot study that evaluates the method verification process, within a PBMS framework. The pilot study examines a new method for analyzing carbonaceous oxygen demand (COD) that does not generate hazardous waste (mercury), evaluating with respect to the current prescriptive COD method. The study is testing and evaluating an analytical and statistical protocol that will reasonably and efficiently demonstrate: (a) laboratory competence with the method; (b) that the new method performance is appropriate for the matrices of interest; and (c) that laboratory performance is maintained. The pilot makes use of two different approaches that have been advocated for PBMS: (1) a DQO approach that relies on data quality objectives as reference criteria upon which to judge method comparability and (2) a reference method approach in which method performance and comparability is judged relative to the performance of an already established and validated method (Eaton, 1999).

NEMI Workgroup
The database is being developed in three phases. The first phase is expected to be completed by December 2000 and will include the 100 methods chosen to test the approach to developing a data dictionary, business rules, user requirement rules, and design using an ORACLE database structure. The second phase, expected to be completed July 2001, will incorporate reviewer comments of Phase 1, include about 250 additional methods, and will create the functional, web-enabled NEMI database. Phase 3 includes updating methods in the database and adding additional laboratory methods and field protocols on an ongoing basis.

Accreditation Workgroup
Many Federal agencies have laboratories that evaluate water quality. The samples are collected for a variety of reasons and are analyzed by a variety of methods. There is considerable variability in the quality control standards implemented by the laboratories, based on the data quality objectives. An issue paper is being developed to assist federal laboratories in making an informed decision regarding the relevance of national accreditation for various water testing objectives. The paper summarizes the range of purposes for which various federal laboratories conduct water testing and delineates the analytical services in which these laboratories are engaged. The paper reviews several key elements in considering which accreditation program best suits the accreditation needs of federal laboratories. Key elements considered in this evaluation include national or international authority, reliance on generally accepted accreditation standards such as ISO 25 or ISO 17025, and state recognized reciprocity. The issue paper also explores current laboratory accreditation standards available for consideration as candidates for the implementation of a national accreditation program. The Workgroup is also providing a review of the proposed privatization of the USEPA performance testing program.

Biology Workgroup
Assessment and documentation of method performance characteristics is essential for appropriate application of environmental sampling and analysis methods, and interpretation of results. One of these characteristics, method precision, is important for establishing and evaluating measurement quality objectives (MQOs). Further, quantification of method precision is necessary to develop data quality objectives (DQOs) for program design and assessment. Similarly, the sensitivity of a method provides an indication of the responsiveness of an indicator to the stressor or stressors of concern. There are a variety of statistical methods that
can be used to assess the precision and sensitivity of a method. The Board has developed a draft issue paper (MDCB, in press) describing procedures for documenting precision of field collection methods for stream benthic macroinvertebrates. This paper specifies several ways in which the precision of a given field collection method can be determined and uses case study data derived from several areas of the United States. Many of these field methods are currently used in the U.S., however, their performance is largely unknown or undocumented (Barbour, 1999). The lack of known data quality from these field methods has hampered attempts to assess biological quality on a national scale (e.g., state 305(b) assessments; USEPA/USDA 1998).

**WQDE Workgroup**

Core data element lists for reporting chemical and microbiological water quality monitoring results have been developed. The core data elements were developed to be compatible with the major existing water quality databases. The core lists are continuing to be reviewed by the water quality monitoring community. The lists will be discussed during regional "public meetings" that will be announced in the Federal Register.

**Nutrients Workgroup**

The Board is providing review of two activities: 1) an appendix on nutrient methods that is a part of the Regional Nutrient Criteria Strategy for Streams and Rivers (USEPA in review); 2) an intercomparison exercise for nutrients in seawater that is being conducted by The National Research Council (NRC) of Canada (NOAA 2000).

**PLANNED FUTURE BOARD EFFORTS**

**PBMS Workgroup**

The Board will develop a framework to characterize the performance and determine the comparability of field sampling methods. Pilot studies will be developed to examine field sampling methods for nutrients and stream benthic macroinvertebrates. These pilots will represent collaborations among several organizations (federal, state, and private) and will rely on the DQO process to establish appropriate methods and measurement quality needed. The Board will collaborate with the Water Environment Research Foundation (WERF) and EPA to provide peer review on the development of DQOs and the selection of appropriate methods for validating laboratory effluent toxicity tests using field biological assessments.

The Board will develop consensus issue papers on other aspects related to PBMS implementation. These may include: status and needs for reference material availability, liability issues, a national framework for accrediting laboratories, training required to implement PBMS, and develop a "how to" guide for conducting pilot tests. To ensure comparable implementation of a PBMS, the Board will study the need for a national program to accredit field personnel who provide environmental data.

**NEMI Workgroup**

The Board will work to develop a NEMI biological methods database that is searchable and provides summaries of performance characteristics for biological methods and that could be used to determine comparability of different biological methods. The biological methods database will include laboratory and field methods that span taxonomic levels. Types of methods currently being considered include: (1) cellular and molecular (e.g., immunoassays), (2) whole organism (e.g., Microtox, algae, invertebrates), (3) microbiological activity/count (e.g., fecal coliform, Cryptosporidia), and (4) field population/community sampling and assessment (e.g., benthic invertebrates). Methods will be searchable by: type of method, source of method, type of 'matrix' (e.g., freshwater, saltwater, tissue), type of organism (and species if appropriate), and by type of quality control requirements. The biological methods NEMI will be linked with other appropriate on-line databases provided by consensus organizations, and federal and state agencies to ensure that current method updates are included. The information obtained during the development of the NEMI biological methods database will be useful for developing the
business rules and data dictionary needed to include chemical and physical sampling protocols in NEMI.

The Board will compile relevant information produced by several agencies to develop concrete, hands-on guidance on developing DQOs, translating those DQOs into MQOs, and choosing appropriate methods using real-world examples from the water quality monitoring field. A "expert-system based" front-end user interface will be developed for NEMI to guide a user to the appropriate field protocol or analytical laboratory method to satisfy user defined DQOs and MQOs. The user interface will be developed so that it can be incorporated into any "on-line expert system" that the National Water Quality Monitoring Council may develop.

WQDE Workgroup

The Board will develop core data element lists for reporting monitoring data from biological, chemical, and physical field sampling, field measurements, and remote sensing.

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WATER QUALITY MONITORING AND GIS: A STEP FURTHER

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This paper focuses on the problems of quantifying the impact from nonpoint sources pollution on the water quality in rivers. The case study was carried out on the Piray River in Bolivia. The river is in the upper catchments of the Amazon River. The sources of the river are the mountainous areas in the western part of the Santa Cruz province in Bolivia. Most of the catchment of the river system is situated in the flat flood plain, where most point-sources are concentrated. At the same time the flood plain is an intensive agricultural area, which leads to non-point pollution. The project used GIS, digital elevation models and full dynamic hydraulic and water quality models to investigate the effects from the different sources of pollution.

INTRODUCTION

River monitoring with the purpose of getting data for water quality assessment and control has been a hot topic for the last decades. The traditional monitoring has been focused on the point-sources like industries, sewage plants etc whereas the monitoring of non-point sources has been more or less neglected and the loads from such sources have been given as the deficit between the total load and the point-sources. However, separate and more precise estimates of non-point pollution have become necessary, prompted by the need to assess causes and effects of pollution and by the fact that concurrently with the significant reduction of point source pollution in many places the non-point pollution loads become increasingly important.

When data are collected for water quality modelling, the constraint is often the limited financial resources for the collection and analysis of samples. The collection has to be optimised and concentrated to the most important locations in the river network. Data on point sources of pollution/load are relatively easy to get. However, the non-point sources can be very difficult to monitor, but they may be the biggest contributors to the load of organic matter, nitrogen and phosphorus. In the present case an analysis of the monitoring data between two stations, 100 km apart, showed concentration of e.g. organic matter and nutrients at the same level. There are no point sources between the two stations and naturally occurring degradation should have reduced the levels considerably between the two stations. The only answer to the analysed levels at the downstream station was load from non-point sources.

Non-point source pollution can be estimated based on information on landuse, population, geology and other basic data. One of the tools for handling such data is GIS. GIS provides the possibility to map a watershed and estimate area-based load for variables such as BOD, nitrogen and phosphorus. Access to relatively cheap images from e.g. the LANDSAT 7 satellite provides the possibility to identify the different types of land-use and even makes it possible to identify the different types of crops. The loads from the different land-uses can to a certain extent be calibrated through a comparison with the concentration of the variables found at the water quality monitoring stations. Several studies on the load from different types of crops are also available and all this information can be efficiently handled and presented through the use of GIS.

Satellite images from LANDSAT 7 will be used in the coming phases to provide better information on the agricultural land-use and thus main types of crops.

Methods and models

Water quality:
In a study of the River Pirai in Bolivia (approx. 10,000 km²) with the purpose of investigating the different impacts caused by point-source and non-point pollution, eight monitoring stations were set up. As the river runs through areas with dense jungle and swamps, the freedom to set up the stations optimally was limited to setting up stations in connection with roads and bridges. A limited number of samples were taken on the eight stations through July to October 1999.
The samples were analysed for organic matter and nutrients. Besides, information about the point-source pollution was obtained from the authorities and factories.

**GIS:**
A GIS system (ArcView, 3.1, incl. Spatial Analyst) was used to classify and delineate the river system and its sub-catchments on the basis of a digital elevation model, DEM, downloaded free of charge from USGS on the Internet.

It was transformed into the local geo reference system. A digitised version of the river was made, based on local geographical. The digitised river was added to the DEM and "burned" into the DEM with an altitude difference of 100 m. The catchment delineation was performed through the use of the "Eight pour-point" procedure in ArcView, where the altitude of each cell in a GIS map is analysed to find slope.

Precipitation for the catchment is used to calculate the actual run-off through a simple precipitation/run-off model and the distributed run-off is then added to each of the sub-catchments.

Information on precipitation, planned land-use, populations in the different municipalities, sewered areas etc. was delivered by the local authorities, the Santa Cruz Prefectura, who have also made land-use plans for the whole catchment. Literature studies reviewed by Beneman et al (1996) on loads from different landuse gave initial values for the non-point loads and these values were assigned to the different types of landuse-areas. Literature values were used, when no actual measurements could be applied.

An overview of the standard values used is given in the tables 1 and 2.

**Modelling systems:**
A fully hydrodynamic water quality model, MIKE 11, was used for the study and the loads from non-point sources was calculated and added to the model through the GIS-based model LOAD. The modelling software MIKE11 is described in DHI (1999 A) and LOAD is described in DHI (1999 B).

The MIKE model is a fully hydrodynamic water quality model, where the flow and degradation can be calibrated to simulate the existing conditions. The model has been developed and upgraded continuously by DHI Water and Environment through the last decade. Is has been used world-wide as flood warning system and to investigate the impacts from point sources.

LOAD is an extension to ArcView, designed and developed at DHI Water & Environment to make use of the GIS information and calculate total load from the different non-point sources (agricultural run-off and run-off from un-sewered villages, based on run-off coefficients). LOAD is able to route and degrade the non-point loads from their origin and into the river in each sub-catchment. For the variables Total Nitrogen, Total Phosphorus and BOD it is possible to set standard loads and degradation levels, resulting in a reduction of the loads, when they reach the river. There is a direct interface between the LOAD model and the MIKE model, making it possible to perform iterations of the non-point pollution to fit the measured levels of load found in the river system.

The data created through the use of GIS was then used to provide better information on the different sources of pollution. Through water quality modelling it is thus possible to split the monitoring data into loads deriving from the point sources and the non-point sources.

<table>
<thead>
<tr>
<th>Land Use</th>
<th>Run-off coefficient</th>
</tr>
</thead>
<tbody>
<tr>
<td>Urban</td>
<td>0.89</td>
</tr>
<tr>
<td>Open</td>
<td>0.22</td>
</tr>
<tr>
<td>Agriculture</td>
<td>0.24</td>
</tr>
<tr>
<td>Barren</td>
<td>0.22</td>
</tr>
<tr>
<td>Wetlands</td>
<td>0.80</td>
</tr>
<tr>
<td>Residential</td>
<td>0.34</td>
</tr>
<tr>
<td>Water</td>
<td>1.00</td>
</tr>
<tr>
<td>Forest</td>
<td>0.15</td>
</tr>
</tbody>
</table>

Table 1. Run-off coefficients used in the LOAD model. From Beneman et al.(1996).
Results and Discussion:

The preliminary results from the first water quality modelling of the Piray River clearly showed that the levels of organic matter and nutrients, measured at some of the stations, were much higher than modelling results, using standardised degradation. An example is shown below. The distance to nearest known point source was nearly 70 km, so the levels of BOD must come from internal sources as production of alga, bacteria and other biological sources of BOD and from external sources as non-point loads from the human activity in the catchment. A similar picture was observed on the main part of the river, where the distance to the nearest source was more than 100 km and the measured levels were higher than the model results.

The recognition of non-point sources of river pollution as having a substantial role in the water quality has led to investigations in ways of quantifying this load. In the present study we used the MIKE 11 model, which has been used globally to simulate the effects of different impacts on the river water quality. In river systems, where the modelling area has covered relatively short distances it has been possible to make simulations, where the simulations and the measured levels were of the same magnitude. In River Piray in Bolivia the river system simulated in the model covered a distance of 240 km, where the point sources were located in the first 100 km.

<table>
<thead>
<tr>
<th>Land-use</th>
<th>Total Nitrogen</th>
<th>Total Phosphorus</th>
<th>BOD</th>
</tr>
</thead>
<tbody>
<tr>
<td>High density urban</td>
<td>2.10</td>
<td>0.37</td>
<td>11</td>
</tr>
<tr>
<td>Residential</td>
<td>3.41</td>
<td>0.79</td>
<td>18</td>
</tr>
<tr>
<td>Agriculture</td>
<td>1.56</td>
<td>0.36</td>
<td>5</td>
</tr>
<tr>
<td>Open/Pasture</td>
<td>1.51</td>
<td>0.12</td>
<td>7</td>
</tr>
<tr>
<td>Forest</td>
<td>0.83</td>
<td>0.06</td>
<td>7</td>
</tr>
<tr>
<td>Wetlands</td>
<td>0.83</td>
<td>0.06</td>
<td>7</td>
</tr>
<tr>
<td>Water</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Barren</td>
<td>5.20</td>
<td>0.59</td>
<td>16</td>
</tr>
</tbody>
</table>

Table 2. Mean run-off concentrations used in the non-point model (mg/l) From Benaman et al. (1996).

Each of the polygons in the GIS map was then divided into the different land-use types and thus different run-off coefficients and loads. For further details in the use of the LOAD model, please refer to DHI (1999 A).

Figure 1 The Piray River drawn on a Digital Elevation Model. The red triangles are the monitoring stations.
of the river. The downstream area of the river was without point sources, but with extensive agricultural areas and jungle. The loads from these sources are expected to be the main loads into the river in the downstream area. However, the internally produced biomass will also contribute to the total load of organic matter and nutrients.

The combination of a fully hydrodynamic water quality model and GIS provided the possibilities to investigate the impact from both the point and the non point sources and made it possible to quantify the different loads through an iterative process. Initially it was expected that the massive fish kills, occurring at irregular intervals in the downstream area of the river, were caused by discharge from sugar mills in the catchment. The modelling combined with the surveys showed that this is probably the truth, but at the same time it was shown that the non-point loads were quite substantial.

The development of GIS systems that can handle spatial data on landuse and precipitation will enable the authorities to focus their effort on the best way to control the effects from point- and non-point sources. It will provide them with an efficient tool in helping to identify where monitoring stations can be established. As the knowledge about the loads from different types of crops is continuously growing, it will be possible for the authorities to make further calculations, where they can simulate different types of crops in different areas of the catchment and then investigate the impact on the water quality.

![Figure 2. An example of the modelling results, compared with the measured levels of BOD (Indicated by crosses).](image)

In the present study, the identification of the different sources and their loads will provide the authorities with a tool to plan for the reduction of the different sources in order to reach the objectives set for the Pirai River. It also provides them with the possibility to perform Environmental Impact Assessments with focus on the aquatic environment through modelling.

However, it is important to realise that the present study used "table-created" levels of the land-use, since no precise information is available. In the continuation of the project it is expected to make use of LANDSAT 7 satellite images, where several types of crops can be identified through their absorbency spectre. But even with more precise land-use data, more investigations on the load from different types of landuse in different climates and under different geological conditions are needed.

The study has shown that the development of GIS systems will provide planners and investigators with very powerful tools when monitoring is in focus. At present most GIS-users are only utilising the "Viewing" possibilities and not the possibilities to integrate models and other decision support systems that can support monitoring, when this has to be tailor-made.

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Access to aggregated information at a European level concerning the marine and coastal environment is of increasing importance for different European institutes (such as EEA, EU, OSPAR and JRC). The CoastBase project, supported by the European Commission within the fifth Framework Programme (DG IST), aims to improve the information flow for the assessments of the marine and coastal environments. CoastBase (URL http://www.coastbase.org) aims to support the provision of timely, targeted, relevant and reliable information on an European scale. This information is needed for a sound preparation of European policies. Access to aggregated information also contributes to the harmonisation of the marine and coastal policy implementation at a national or sub-national level.

The CoastBase project focuses on the construction of a technical architecture and the related communication process among European authorities at different levels, which is essential to put techniques in to practice.

Users of coastal and marine data and information on different aggregation levels are represented within the CoastBase consortium. Their input was used in an information definition and user requirement study as a first step in the development of CoastBase. Matching of user requirements and data availability resulted in the proposal for two case studies, on eutrophication and on planning.

The CoastBase-server will be the virtual ‘door’ to information sources, this will be the interface for both the querying and the display of results. Through this interface connections and applications are activated by the user. CoastBase will also provide access to European level aggregated Marine and Coastal information.

**INTRODUCTION**

Coastal areas and their natural resources play a strategic role in meeting the needs and aspirations of the European population. The large variety of environmental conditions and gradients creates highly productive systems for all kind of sectors. Management authorities consider these uses and interactions between actors from different sectors and policy levels for integrated coastal zone management.

The National Institute for Marine and Coastal Management is a partner within the Topic Centre for Marine and Coastal Environment, the consortium providing information to the European Environment Agency.

The information gathering and manipulation process becomes more and more relevant for the policy support throughout the EU, which has to connect well to the lower level policy makers to assure adequate implementation of European regulations. The quality of products used for policy making is highly determined by the data and information used as well as by how data are manipulated. Efforts and time needed to obtain good information at an European level costs at this moment a significant amount of work, and the results do not reflect the amount of available knowledge and information.

**PROBLEM DEFINITION**

The process of aggregating and interpreting information is an essential, but time and money consuming process. This information is dispersed among coastal institutes and organisations and neither readily available nor accessible.

The technical and communication processes and the interaction between many European institutes regarding the information flow presently is not functioning effectively. This lack of availability and accessibility of information influences effective planning and decision making. In addition, no account is taken of the level of comparability and the quality of data or information from different sources.
OBJECTIVES

CoastBase's main objective is to improve the availability of data and information through a technically innovative design, and contributing to improvement of the context, analysis and understanding through pan-European participation and incorporation of a feedback and communication structure.

The CoastBase-server provides the virtual ‘door’ to information sources through interfaces for querying and displaying results. Information will be accessed either from:
1) Dispersed information sources via metadata and data links and/or
2) European level aggregated Marine and Coastal information via an internet accessible database.

CoastBase focuses on the construction of a multilingual technical architecture for search, access and aggregation of coastal and marine information. It also focuses on facilitating communication among different levels of authority in European organisations, as such communication is essential to put the newly developed techniques into practice and make the system operational.

INFORMATION INTERPRETATION

Information is a collection of data that is relevant to a recipient at a given moment in time. Information is data in their context: it has meaning, relevance and purpose. An important consequence of this definition is that we see information as one element of a hierarchy through which sustainable management of the coast can be achieved, thus:

\[
\text{Data + Context} = \text{Information} \\
\text{Information} + \text{Analysis} = \text{Understanding} \\
\text{Understanding} + \text{Management} = \text{Possibility of sustainable action}
\]

Most of the marine and coastal information is generated with a national or regional scope, directed towards the demand for that level of policy. In order to use this information for assessments on a supra national level the information will be aggregated. Within the process of aggregation a good understanding of the information and the methodology is essential. In order to obtain this understanding good communication between the provider and the end-user is needed.

The structure of the system can be divided into three main blocks (Kazakos et al., 2000):
The process of interpreting information to a European level is complicated. Both themes and related measures required are different; good co-operation between institutes functioning at different levels is essential. 

*Information to assess the state of the environment and support policy measures for an aggregated level and a larger geographical area needs validation possibilities from the original information source.*

**TECHNOLOGY**

A generic system will be developed connecting and automatically updating the local metadata and CoastBase metadata into one CDS (Catalogue of Data Sources)-based virtual frame, allowing dynamic querying of real information of these information sources. Separate modules will be developed to aggregate, convert and interpret and check the quality of the information.

- **The Virtual Coastal & Marine Catalogue Services**
  It contains the metadata necessary to identify the data and information and their sources. From a single point of entry presented in different languages, it is possible to search for data and information spread around different data banks, in different formats and at different aggregation levels.

- **Data Access & Manipulation Services**
  After finding the requested information, the system can access them directly. Raw data, maps, documents and charts can be downloaded, processed and manipulated. Proper tools to do so will be provided within the system. Finally it is possible to upload the achieved results and share them with other users.

- **Feedback Module**
  A module provides automatic feedback to the information source so that providers can track what happens with their information - for example, by sending them back the obtained aggregated information (a map, report, graph etc.). At the same time they will be more engaged in the European policy making and monitoring; communication among these authorities will be intensified, ultimately enhancing pan-European collaboration.

The Communication and Feedback procedures focus on the commitment and structural support of information sources and a good tuning of required and offered information by the different actors.
COASTBASE PARTICIPANTS

Institutes related to different aspects in the storing, searching, accessing and manipulation of information are present within the CoastBase consortium. This includes IT developers, administrations holding information sources at different European levels, governmental organisations experienced in collecting and aggregating information and an organisation specialised in the European meta-data developments. The European Environment Agency (EEA) is co-operating actively with the project, assuring good harmonisation with EEA developments in this field.

INFORMATION DEFINITION AND USER REQUIREMENTS

In the previous sections the project mission, its goals and history have been presented. As a first step in the actual development of CoastBase, an information definition and user requirements study was performed. This study aimed to get a clear image of the users of CoastBase, the data providers, and the type of data that can be made available for the CoastBase project. The following steps were taken in this information definition and user requirement study (Eleveld et al., 2000):

• Define the audience;
• Determine user goals and expectations;
• Describe the available data banks;
• Match user requirements with data availability;
• Determine target areas and topics.

One of the final objectives of the CoastBase project is to provide authorities, research institutes, universities, private sector and public easy Internet access to CoastBase. For the development of a working prototype CoastBase partners representing users were defined. Institutional users interested in data on various levels are represented within the consortium, from European, to regional (countries and states involved in the OSPAR and HELCOM Commissions), to national and local data.

RESULTS FROM THE INFORMATION DEFINITION AND USER REQUIREMENTS

Four groups were distinguished in the identification of institutional users, users working on a European, regional, national and local level. They have the following in common:
• They are all dealing with spatio-temporal issues;
• They are all somehow involved in, or supporting, policy-making.

Institutes working on 3 different (of the 4) levels provided examples demonstrating how CoastBase could be used by actual users in their organisation. These examples had in common that disparate data are used, compared and/or integrated.

<table>
<thead>
<tr>
<th>Category</th>
<th>Institute</th>
</tr>
</thead>
</table>
| European | European Topic Centre on Marine and Coastal Environment (ETC/MCE)  
Joint Research Centre (JRC) |
| Regional | International Council for the Exploration of the Sea (ICES) |
| National | National Institute for Marine and Coastal Management (RIKZ)  
Italian National Agency for New Technology, Energy and Environment (ENEA)  
Institute for Marine Research (IMR)  
Information Training Local Development Ltd. (PETA) in co-operation with the Greek Ministry of Environment  
Marine Institute Gdansk (MIG) |
| Local | European Union for Coastal Conservation (EUCC), representing local users |

Overview of institutes per level
CoastBase also aims to satisfy a variety of individual users, it seems necessary to see if certain groups of users are distinctly different, and if they have very different wishes, so that a certain group is not unintentionally excluded. Based on the input from CoastBase users four groups were distinguished in the identification of individual users: decision & policy maker, policy advisor & project manager, researcher, and database administrator & programmer. They all have the following in common:

- They are all dealing with spatio-temporal issues;
- They all use a PC;
- They all produce output in the form of reports.

**DISCUSSION**

To enable an 'user focus' in setting up the system (Doody et al., 1998), the user perspectives were analysed to see if differences in user groups occurred. The institutional users were grouped based on their focus (European, regional, national, local). The individual users were grouped according to their function.

Individual user groups prefer data of different interpretation levels. In general, the researchers have detailed knowledge about a limited area of interest, while the decision makers have to have broader an overview over different topics and areas. This difference in focus results in different data needs.

<table>
<thead>
<tr>
<th>Interested in raw data</th>
<th>Interested in interpreted data</th>
</tr>
</thead>
<tbody>
<tr>
<td>'General Public'</td>
<td>Decision Makers &amp; Policy Makers</td>
</tr>
<tr>
<td></td>
<td>Policy Advisors &amp; Project Managers</td>
</tr>
<tr>
<td></td>
<td>Researchers &amp; Database Administrators &amp; Programmers</td>
</tr>
</tbody>
</table>

The database administrators & programmers and researchers at the right hand side of the figure are mainly interested in raw data while the decision and policy makers at the left hand side of the figure are interested in interpreted data. Policy advisors and project managers occupy an intermediate position. The work of some policy advisors is closely related to the work of the researchers while others operate more closely at the side of the decision makers.

Both institutional and individual users had different preferences for area and topic. Matching of user requirements and data provision results in a proposal for geographic areas (2) and topics / thematic fields (2) to be covered by CoastBase.

During a plenary project meeting, an agreement was reached on the areas and topics to be covered in the CoastBase prototype:

1) Both a Mediterranean and a North Sea coastal area, focusing on Integrated Coastal Zone Management (ICZM) and planning. Data for planning projects is often very disparate. There is a clear interest from both institutional and individual users.

2) The North Sea region, focusing on water quality / pollution & eutrophication. Here the main interest is on the process to derive European level indicators for policy making from raw data.

**CONCLUSIONS**

The challenge of CoastBase is to realise a technological architecture that will contribute to improvement of the assessment of marine and coastal environments. Tailor made information towards the end-users is essential in the project. Within the two coming years a prototype showing functionality and long term viability will be developed.

The end-user requirements of all actors in this process are being gathered, technological and financial limitations will determine the realisation of the final product.

The generic nature of CoastBase allows the use of the technical architecture in other fields in which access to multiple, distributed information sources is required. The CoastBase system foresees a minimal effort of the information sources considering the IT investments and a fully automatic linkage at metadata and database level reducing maintenance costs and efforts.
ACKNOWLEDGEMENTS

We thank all volunteers and CoastBase partners for their input, and Ardy Siegert for the technical analyses in the information definition and user requirements study. The European Commission, DG Information Society is acknowledged for financial support of the CoastBase project (IST-1999-11406).

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THE USE OF SCIENCE IN MONITORING

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As I've listened to the papers presented during Monitoring Tailor-made III, I've heard a number of calls for better monitoring through the use of sound science. There is no doubt that good monitoring involves good science. However the means by which we formally employ 'sound science' to improving water quality monitoring is not clear to me. In general, bringing sound science into monitoring will involve some form of peer review.

There are examples of monitoring programs that are currently using peer review to insure that the information produced is scientifically sound. The South Florida Water Management District currently has its annual water quality report peer reviewed by a panel of scientists. The U.S. Geological Survey's National Water Quality Assessment Program (NAWQA Program) has its monitoring program's design peer reviewed by the National Academy of Sciences.

Thus, as I reflect upon the calls for sound science I've heard during MTM-III and the examples with which I'm familiar in the U.S., I want to share with you my thoughts on questions we need to confront if we are to employ peer review in the design and operation of our water monitoring efforts.

With regard to MTM-III, I sense that monitoring system design is becoming more accountable for the information it produces as well as the methods used to produce it. Here are several of my observations during MTM-III:

1. Since MTM-I, considerable attention has been given to understanding and documenting the information we expect from water quality monitoring;
2. Monitoring designers and managers are exploring a wide variety of ways of producing water quality information and relating the information to decision-making and public understanding;
3. Monitoring methods, practices, and strategies are being tested in greater number today than six years ago with the result that considerable refinement in monitoring systems is occurring as more experience is gained; and
4. Not all monitoring problems and issues have been resolved nor are monitoring system refinements to the level many of the speakers at MTM-III are seeking.

During the meeting, I've heard calls for:

1. Improved science in the monitoring toolbox;
2. The need to better match management instruments with monitoring capabilities;
3. Involving policy makers in efforts to peer review monitoring systems;
4. The need to listen carefully to the 'science' questions facing water quality management;
5. More comparable and consistent information from monitoring – it is necessary to build confidence in monitoring;
6. Better definitions of 'compliance' monitoring as a precursor to improved compliance monitoring; and
7. Improved and accepted water quality indices for management purposes.

I also heard references to 'The good, the bad and ... the unknown' as a description of the information landscape in water quality monitoring. It was acknowledged that the multi-disciplinary and multi-agency approach being called for in integrated water management today will demand much more science rigor.

On the first day of the meeting, I heard the question: "Is monitoring a scientific activity?" It definitely involves science disciplines, from biology, chemistry, physics, and statistics to hydrology, engineering, computer science, and journalism (reporting). Adding science to monitoring will require an interdisciplinary approach.

In some ways, we are getting mixed signals: There is a lack of public confidence in science, and yet there are laws being passed that require 'peer review' of the results of monitoring.
The 1994 Everglades Forever Act, passed by the Florida legislature, states in section 373.4592(4)(d)(6):

"Beginning January 1, 2000, the district and the department shall annually issue a peer-reviewed report regarding the research and monitoring program that summarizes all data and findings."

Implementation of this legal requirement requires the South Florida Water Management District and the Florida Department of Environmental Protection to prepare an annual 'overview of the status of compliance with water quality' standards in the Everglades Protection Area. The report is then peer reviewed by a panel of scientists from around the world. How is the annual report of a water quality monitoring program to be peer reviewed? What are reviewers to consider relative to the state-of-the-art? Having been a member of this panel over the past three years, I can assure you that these questions are regularly confronted when the panel meets to review the Everglades annual report.

As a sidelight, it is interesting for designers and operators of water quality monitoring programs to reflect upon the question: How would my monitoring program's reports hold up under intense scientific peer review as required in the Everglades?

The USGS's National Water Quality Assessment Program introduces peer review into its water quality monitoring in a different manner. Rather than have the annual report peer reviewed, the USGS has the design of its monitoring program peer reviewed by the National Academy of Sciences. Once the design has been peer reviewed, the monitoring program operates producing water quality information that is based peer reviewed science.

Thus the question arises: What is the appropriate way to bring sound science into water quality monitoring systems producing information for management and public understanding? In this context, how is peer review defined?

The U.S. Environmental Protection Agency (EPA) published a Peer Review Handbook that expresses a policy that its "technically based work products related to Agency decisions normally should be peer reviewed." EPA defines peer review, for its purposes, as follows:

"Peer review is a documented critical review of a specific Agency major scientific and/or technical work product. The peer review is conducted by qualified individuals (or organizations) who are independent of those who performed the work, but who are collectively equivalent in technical expertise (i.e., peers) to those who performed the original work. The peer review is conducted to ensure that activities are technically adequate, competently performed, properly documented, and satisfy established quality requirements. The peer review is an in-depth assessment of the assumptions, calculations, extrapolations, alternate interpretations, methodology, acceptance criteria, and conclusions pertaining to the specific major scientific and/or technical work product and of the documentation that supports them. …. Peer review is usually characterized by a one-time interaction or limited number of interactions by independent peer reviewers. Peer review can occur during the early stages of the project or methods selection, or as typically used, as part of the culmination of the work product, ensuring that the final product is technically sound."

EPA continues in its Peer Review Handbook to state that peer review is not free; however, not doing peer review can be costly. To be most effective, EPA notes that peer review of a major scientific and/or technical work product needs to be incorporated into the up-front planning of any action based on the work product – this includes obtaining the proper resource commitments (people and money) and establishing realistic schedules.

**Obtaining a ‘Sound Science’ Label for Water Quality Monitoring**

When water quality monitoring, used to produce management and public water quality information, seeks peer review, there are a number of key issues to be resolved. Questions, such as those below, must be answered.

1. Who are the peers that should serve as ‘peer’ reviewers of water quality monitoring? Should the peers be biological, chemical, statistical and hydrological scientists? Should a user of the resulting information (i.e., a manager, policy maker and/or the public) be considered a
‘peer’? Are there water quality management information specialists that can be considered ‘peers’ for monitoring system peer review?

2. What should actually be peer reviewed – the final report each year or the initial monitoring program design?

3. If the annual report is to be peer reviewed, is the report peer reviewed only for the science it contains? Or is it also reviewed for ‘understandability’ and relevance to its management information purpose? Should it be reviewed for management’s accountability to the public paying for the management programs?

4. How do we build peer review into monitoring? In other words, when is it scheduled (annually, on a regular basis as information is produced, or every five years)?

5. How much will monitoring peer review cost and how should it be funded?

These questions raise a number of related questions. If the current system of peer review is used to insure that sound science is employed to produce water quality information, does each person collecting the data and analyzing it have the option to choose any ‘scientifically acceptable’ method in searching for the scientific ‘truth’ regarding the status and trends in water quality? This use of science, while moving scientists toward a better understanding of the ‘truth’ of water quality conditions, may not be best understood nor used by managers and the public because it may not comparable over space, time or management programs. What obligation does science have to agree on ‘standard’ scientific methods for producing management relevant information? Is such agreement counter to the concept of peer review?

While these are not easy questions, they do reflect a field of scientific endeavor (producing management information) that is maturing. It is only recently that such peer review questions, relative to management information purposes, have been posed. The two National Monitoring Conferences held in the U.S. and the three Monitoring Tailor-made workshops held in the Netherlands over the past six years are creating a literature against which peer review of monitoring systems for management information is feasible. The experiences of peer reviewing SFWMD annual water quality reports and USGS National Water Quality Assessment Program further establish the use of peer review in monitoring.

What are other disciplines doing to establish sound science under broad information reporting systems, such as economic and weather reporting?

**Lessons from Other Disciplines**

Economists have established a number of statistics that are reported regularly to define the health of our economy. For example, the Dow Jones Index in the U.S. is carefully defined and regularly updated to maintain its ability to convey relevant information to the public. The Gross National Product, unemployment statistics, and housing starts are other examples used by economists to describe our economy’s behavior.

Meteorologists have developed a blend of graphic, numeric and narrative information that is regularly published on the back page of most U.S. newspapers. The information is readily accepted as being correct in describing past weather behavior and the best available in predicting future conditions. There is a level of standardization that permits people from all over the U.S. to readily understand what is happening in other parts of the country.

Air quality and water level information are included in some newspapers where it is used to inform the public about air pollution levels and potential flooding and drought conditions.

Unfortunately, when information about the U.S.’s water quality is presented in the newspaper, it appears on the front page as an expose of poor government, usually presented by a special interest group that has analyzed existing data, using its own methods, to draw dire conclusions. The means to measure water quality is not standardized nor is it presented in a form the public can readily understand. We in the water quality monitoring field have failed to agree on ways to produce water quality information for the public. We lack a standard, peer reviewed means of obtaining water quality information that can find a regular place in newspapers, along side economic, weather, and air quality information.

Economists and meteorologists have applied science to the design of publicly oriented information systems in ways the water quality community has not. There are lessons to be learned.
Concluding Remarks

While introducing strong, peer reviewed, science into water quality monitoring will not be easy nor straightforward, we are seeing a field of endeavor, water quality monitoring, that is maturing and producing a body of knowledge that will lead to a better scientific basis in the future. In 1994, at the first MTM, it did not seem possible to talk about ‘peer reviewing’ a water quality monitoring system. The context of producing tailor-made information was just beginning to be defined in terms of supporting water quality management decision-making. The very healthy and enlightening dialogue during MTM-III is strengthening the case for a stronger scientific foundation for monitoring.

Where will better science in monitoring take us in the future? I have a vision that in 20 or 30 years from now, society will be operating mature water quality information systems whose designs are peer reviewed. Such information (monitoring) systems will permit the production of comparable and consistent water quality information over time and space. The information will be regularly published for use by management agencies, and the same data will be converted into information readily understood by policy makers and the public. The water quality information will appear in the regular news sources of the time. The water quality information system designs will be evaluated (and peer reviewed) at 5-year intervals to keep them scientifically sound and relevant to public information needs. A large body of knowledge on which to base the peer review of the water quality information systems (much of it initiated during the MTM workshops) will have been produced over the years. The monitoring specialists of the day will meet periodically, under the auspices of a Bureau of Environmental Statistics, to discuss design approaches and the science that underpins them. The public will come to expect regular updates on the water quality conditions in their community, nation, and world. And the comparable and consistent information produced will have developed considerable public confidence in the information (monitoring) system’s design and operation.

Before closing, I want to thank our hosts for organizing and conducting an outstanding meeting on the production of information for sustainable water management. The meeting has been both professionally and personally enjoyable. The enthusiasm for the topic seems to be growing with each MTM workshop and the interest is spreading to more and more countries. We are well on the path to the above vision and MTM is providing the scientific stepping stones to get there. Thank you.
INTRODUCTION

Monitoring Tailor-Made III is the third international workshop on strategies and practices to design, implement and report monitoring programmes which render information on aquatic resources. This third workshop put emphasis on information for sustainable, integrated water management. Integrated water management takes economic, ecological and societal issues into account and requires thus information on the status of aquatic resources in relation to the economic and societal issues. Frequently, many data are available for all of the issues. Nevertheless, policy-makers perceive the information available as inadequate. A framework is required for the aggregation of data from the different disciplines and for information which can underpin integrated water management. Such a framework information-architecture is needed which enables meaningful aggregation of data of the different viewpoints to be made and which allows trade-offs between economic, societal, and ecological dimensions to be evaluated. The conceptual technical elaboration and implementation of such a framework is a challenge facing all environmental managers today.

Exchange of knowledge and experiences is essential to further develop the quality and accessibility of information. The workshop Monitoring Tailor-Made provides a platform for an active and effective interchange of ideas and practices. The first and second Monitoring Tailor-Made workshop focused on information and the role of monitoring in it. Monitoring Tailor-Made III continued this general theme with a specific theme of: how to assemble integrated information to support sustainable water management. Four levels of integrated information are distinguished: integration of science, policy-makers and the public; integration of different scientific disciplines in the domains of natural and socio-economic sciences; integration on a spatial scale; and integration of measurement and data-treatment methodologies within the domain of the natural sciences. The main findings from the presentations and discussions during the workshop Monitoring Tailor-Made III are presented in this paper.

INTEGRATED WATER ASSESSMENT

Monitoring is used to assess the quantity and quality of water and is often linked to legal instruments to regulate the use of the water body. The growing importance of diffuse pollution makes common monitoring practices less suitable as a policy instrument, as Russell shows us in his paper; but also makes a good classification of waters far more complex than a single-sample exercise, as Ward shows in his paper. All in all, an integrated approach to assessing water bodies is needed.

A wide range of definitions for integrated assessment have been used during the workshop. From the various presentations one can conclude that integrated assessment implies integration of at least the following:
- Information from different countries and authorities;
- Different disciplines such as biology, chemistry, economics, etc.;
- Different types of information. An example of this is the D-P-S-I-R (Driving force - Pressure - State - Impact - societal Response) approach.
- Models, remote sensing & monitoring;
- Data (through aggregation): basic data are transformed into indicators.

This range of definitions indicates that the integrated assessment concept itself and/or the use of this concept needs to be sharpened. However, various characteristics of integrated assessment have come to the fore. Integrated assessments should include:
• Multidisciplinary teams with scientists applying holistic approaches. Scientists from the social, economic and natural sciences should work together on the issues, each being aware of having blinkers that may hinder the full view of the issue.

• Communication. Special emphasis on communication within and between parties and for dialogue with policy-makers. This communication is essential in information production process and specification of information needs can prove to be crucial basis for this communication. The paper by Timmerman and others provides a framework for the specification of information needs.

• Creation of understanding of, and support for, approaches chosen. The people involved in the process must have an open mind and show interest in each others' concerns and working methods. Only understanding can lead to support.

• Orientation on the process rather than on the project. There is a common tendency to focus on 'technical' details of a project, meanwhile avoiding more sensitive, political issues, which usually in the end are crucial to the success of a project. By focusing on the process, a more realistic view of the requirements for the success of the project can be obtained, thus avoiding unnecessary pit-falls.

• Special attention for the level of aggregation. For example:
  → aggregation across spatial and temporal scales. As e.g. the social and economic disciplines normally work on larger scales then the natural disciplines, integration of information of different disciplines can be a tough task.
  → type of models used. Only a truly independent institute can develop widely accepted high level models.

• Setting of clear priorities. Having a clear objective can be very helpful in choosing the right measures. For example, a flat-rate emission reduction approach can be much less cost-effective compared to an effects-based emission reduction approach

• Uncertainty issues. Uncertainty can be used to delay action, therefore it can be advantageous to direct the research agenda to the most uncertain parts of a complex problem.

• Multiple information sources. Information from different disciplines is needed for integrated assessments. There is however no necessity for each project to provide all the data needed for an integrated assessment. Suitable information can nowadays be obtained from various organisations through various tools like internet, models, and decision support systems.

• Realistic ambitions. A simple start can provide proof that the approach chosen really works.

The following sections examine these issues in more detail.

THE MULTIDISCIPLINARY APPROACH

Focusing on the multidisciplinary approach within integrated water assessment, it appears that there is a need for translation across disciplines. The term 'compliance monitoring', for example, is a way of keeping up with the progress made in implementing policy actions in the socio-economic sciences. On the other hand, in the natural sciences 'compliance monitoring' is used for testing against standards. Specialists in different fields also have naive expectations of work from other disciplines. In water science, for example, environmental values and environmental damage functions are often incomplete or missing. Such different perceptions hamper communication between disciplines. Another issue that hinders communication between disciplines is the mismatch in spatial and temporal scales. Monitoring in the natural sciences is performed on a rather detailed scale in time and space (monthly in a river stretch), whereas the socio-economic sciences focus on larger scales (yearly in a province).

To overcome communication mismatches, closer, and regular, contact is needed. Working towards sustainable solutions calls for intensive co-operation and communication between policy-makers and scientists in a multi-disciplinary, and, where relevant, transboundary setting. Through the exchange of ideas, such co-operation enhances the mutual understanding, and, consequently the support of solutions. Sharing of a common problem in a 'joint learning curve' eventually leads to a better utilisation of knowledge and available information. One crucial condition to make such co-operation work is a strong notion of joining in the effort by involved stakeholders. Pollution problems often have multiple causes and cannot be connected to only one of the parties involved. Furthermore, solving one pollution problem may cause another. The paper of Harremoës and Turner provides examples of this. When the aspect of blaming each other enters the discussion, the effort will go into denial and counter-blaming while any possibilities for solving the problem will be delayed or even blocked. A basis for
communication may be found in the use of indicators, where, from a natural science point of view, the D-P-S-I-R framework may assist in bridging the gap between disciplines. However, the multidisciplinary approach will require us to change our present-day organisations.

INDICATORS

Indicators may help in simplifying communication. Aggregation to the desired scale can in particular be an important issue that is supported by the use of indicators. Next to this, the use of indicators is frequently linked to the presentation of data and information. Figures provide good possibilities to present the aggregated data in a condensed format. The Cyprus example (Michaelidou and Maro) shows condensed information in nine quality/effects indices, which when grouped in one AMOEBE type figure, provide a powerful communication tool. The Dutch Vechte example (Verhallen and others) shows that it is possible to communicate uncertainties through indicators. The UK-example (Seager) shows that the indicator of ‘percentage of rivers with good quality’ provides useful information for policy-makers. However, the indicator is an aggregation of a multitude of data, which leads to the conclusion that the use indicators does not necessarily imply a reduction of the monitoring effort.

During the workshop it appeared that the D-P-S-I-R framework has wide support. There are, however, few, if any, examples of full implementation of this framework. Reasons for this may be that the information ‘system’ is not yet embedded in an integrated assessment framework or in a multidisciplinary approach. But the D-P-S-I-R framework is not the last step in the multidisciplinary approach. Communication should be tailor-made. To illustrate, for politicians there is a need for indicators that describe the progress in the policy process. Also, decision-makers require information on a high level of aggregation. This once again stresses that the dialogue between policy-makers and scientists is imperative. Consequently, also in the field of indicators there are no universal solutions, and there is still much work to be done.

MONITORING PRACTICES

Contributions to Monitoring Tailor-Made III provide many examples of initiatives for introducing tailor-made practices. The issues dealt with in previous MTM workshops reappear; quality assurance for instance, is not limited to laboratory analysis, but is now extending to sampling procedures and biological measurements. Statistics, agreed methods (protocols), and knowledge of chemical, hydrological and biological processes are applied in the development of monitoring networks. Information and communication technology (ICT) plays a major role in data-management and the monitoring cycle is implemented. All this signifies that improvement in monitoring is a continuous process.

The need to establish a dialogue with policy-makers is generally acknowledged and linked to the importance of specification of information needs. In the process of specifying information needs, practices from the social sciences prove to be useful, such as the ‘devil’s advocate’ technique. Many initiatives to improve or organise co-operation have been shown during MTM-III. Examples are the US National Water Quality Monitoring Council, ECE guidelines on transboundary waters, the Lake Peipsi example (which explicitly includes the important aspect of capacity building), and various examples on regional scales.

One important aspect of monitoring discussed was the need to reserve time / capacity in the monitoring network design for special surveys. There should be slack in the monitoring network to enable the performance of survey activities on subjects that are not yet known at the time planning is done. In this way, "surprise" issues can be quickly addressed with appropriate monitoring programs.

In the case of pollution incidents, emergency monitoring and rapid assessment of the hazard and likely exposures are needed to determine the immediate actions. This should then be followed by long-term monitoring and assessment of the progress of rehabilitation of the contaminated area. In preventing accidental pollution, an inventory and risk assessment of potentially hazardous sites should be made.

It may be concluded that progress in improving monitoring systems is being made, but there is no reason to be satisfied. The numerous capabilities offered by modern technology are rarely
used and the major problems we are struggling with are not addressed from an information technology point-of-view.

**MONITORING AND COMMUNICATION**

As people become more and more politically aware, they demand more information. This leads to a situation where information is provided outside the long-established sphere of legislation, governments and industries. Accidental situations are significant events in this respect. During pollution incidences, for instance, communication of information to the public plays a major role, especially in avoiding misunderstandings of the actual state of affairs. To be prepared for incidences, a plan should be ready for co-operation between the involved institutions, communication between the institutions and to the ‘outside world’, and monitoring. In developing such communication plans much can be learned from social scientists. For example, ‘the public’, as a uniform entity, does not exist. There are different groups of stakeholders, each having their own interests and concerns. In communicating, each group should be addressed in a different way. The media (newspapers, television) play a significant role in dissemination of the information to ‘the public’. This role can be supportive yet also misleading. In any case, the role of the media is unpredictable.

Three interconnected levels of communication can be distinguished:

1. Provision of information as a means to increase awareness: this type of communication is currently used most frequently; the situation is described together with possible consequences.
2. Appeal to ethical standards to change behaviour: this type of communication appeals to the citizen’s conscience. One example is a government message that drinking water is a scarce resource and the public should be careful not to waste water.
3. Involvement of ‘public stakeholders’ through a genuine dialogue, which will require institutional reforms. In many countries, public participation is used to inform the public, but also to improve plans (i.e. "true" inclusion).

The first two levels of communication are largely one-way options whereas the third level assumes a two-way dialogue. The challenge for the water science people will be to think beyond a technical approach and prepare for better communication.

**INFORMATION AND INTERNET**

More and more, the internet is seen as the medium of choice to disseminate information. Reporting is done on the web in real time, open GIS is very popular to unlock information and, also, the internet is an important source of information, for instance for downloading remote sensing images. As more data are available through the internet it becomes more apparent that data cannot easily be compared. Consequently, the internet is considered to be an instrument to enforce comparability. The full potential of the internet however is not yet exploited. The internet is still seen as an easy to distribute, up-to-date ‘modern brochure’ with real time, changing data. The possibilities of the internet, for instance, as a medium to interact with stakeholders is not yet fully developed.

**RECOMMENDATIONS**

Providing information for integrated water management requires major changes in the customary method of monitoring. It demands intensive co-operation between policy-makers and scientists. It also requires a multidisciplinary approach that utilises the expertise of various scientific disciplines. Such changes cannot be achieved without changing the present-day organisation and an development of a willingness to think beyond the narrow technical approach.

The D-P-S-I-R framework provides a setting in which a multidisciplinary approach can be stimulated. Driving forces and societal Responses require interpretation from the socio-economic disciplines, whereas Pressures, State and Impact can be approached from the natural disciplines. Indicators within this framework can be powerful communication tools for co-operation between policy-makers and scientists.
Communicating water science information to ‘the public’ demands insights from the social sciences. True communication with and involvement of the public can only be achieved by a genuine dialogue.

Rapid developments in technology provide numerous opportunities to improve monitoring and dissemination of information. Effort is needed to explore such opportunities and exploit the existing technological possibilities.

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DESIGN OF MONITORING NETWORKS BASED ON FRACTION CALCULATIONS. AN APPLICATION IN SOME POLDERS IN THE NETHERLANDS

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The analysis of water quantity results mainly covers the analysis of flows and water levels as a function of time and location. From a water quality management and planning point of view it is important to know from which source or area a particular amount of water arriving at a specific location is originating. This type of analysis is often referred to as ‘Source analysis’ or ‘Fraction calculation’. The word ‘fraction’ is related to the fraction or percentage of water originating from user-specified sources of water. Source analysis improves the understanding the behaviour of water system and is used to study the dynamics of mixing of water in networks of ditches in several Dutch polders in the administration of DWR, the Netherlands.

Source analysis or fraction calculations indicate from which source(s) the available water originally comes from at any location in the water system, and furthermore which percentage of water originates from each of the specified sources. The power of such an analysis is manifold, one aspect is that fraction calculations enable a first estimate of the water quality situation at any location in the system based on the water quality of each of the identified sources without taking into account information on waste loads or water quality processes inside the simulated water system.

The results of water quantity computations (or measurements) are the only required input information for fraction calculations. A fraction calculation is performed by means of a water quality model, each source of water that has to be analysed has to be included as a separate water quality variable in the water quality model. The fraction calculation can start if the boundary, or any other inflow of water to the system, concentrations are specified correctly: for the specific boundary the concentration (fraction) has to be set to 1.0 (100%) and for all other boundaries (sources) to 0.0 (0%).

The fraction calculation will have the same characteristics as the water quantity calculation: if the water quantity simulation is time-variable, then the fraction simulation will be time-variable in exactly the same way. In most graphs presenting the results of fraction calculations there is indicated a source called ‘Initial’ (coloured red in the graph). This represents the water that is present in the water system at the starting time of the calculation: by definition the fraction composition is unknown at that moment. The rate of removal or replacement of this initial water illustrates the residence time of water in that specific part of the system. Other fractions which are of interest for polder systems are the points of inlet (sluices, intake pipes etc.), precipitation and salt water intrusion. For ecological development in polders, the water originating from within the polder itself, so called ‘area-hold’ water, is considered of importance.
Fraction calculations will assist the water manager in the analysis of the pathways of water from specific sources and the dynamics of mixing in networks. For the design of the monitoring networks this study geographical overviews of several fractions are presented.

An example of the result of a fraction calculation is included in the next figure, indicating the contribution of the various sources of water in the polder Groot Wilnis/Vinkeveen and their variability, as several locations, within six hydrological years.
INNOVATION OF MONITORING: INFORMATION CYCLE AND PROCESS CHANGES

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In 1996 a programme was initiated for innovation of the monitoring sector of the Dutch Directorate General of Public Works and Water Management (Rijkswaterstaat). This innovation included the use of new technology like remote sensing and modelling, but also the specification of needs for information and a vision for the future of the sector.

The programme started with the development of a method to specify information needs and demonstration of new technologies to users. During the programme it became clear that not technology was a major incentive for innovation and adaptation of the organisation, but specification of information needs and future scenarios. This resulted in a change of the programme towards definition and organisation of work processes for water management (the information circle), process driven change and the introduction of new elements to the programme.

Some of the new elements were:
• information strategy: a method is developed to determine what strategies are available, if it is known what information is needed.
• not technology but demand driven research was initiated: for the field of sediment characterisation the information needs were determined, new technology made available and pilot projects started. The organisational implementation is a major part of this innovation.
• internal cooperation is stimulated by networking; cooperation between the governmental organisation and businesses will be adapted by discussion core competencies; new strategies will be drawn and implemented.

Innovation by this programme has resulted in major changes in the information sector of Rijkswaterstaat. Especially the mindset has changed towards customers, satisfying customer needs and use of new technology. This change was essential for innovation of this sector and now a fruitful environment was created for practical follow up. In the continuation of the programme this will be translated in introduction of new technologies and new working practices.

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Good water management requires operational information. Especially ecological system information is often fragmented, incompatible and incomplete. At best, data is available in databases but its use hindered by lack of integrated analysis- and geographical display tools. The North Sea Information System is an Internet based knowledge system that includes tools for integration of information on aquatic ecosystems and biodiversity. Knowledge will be made applicable using decision support systems, graphical analysis of data, including GIS, and models. The North Sea Information System is designed for the North Sea but its software can be applied to any ecosystem. It aims at developing tools for statistical analysis and graphical display of ecologically relevant data. Fields of application are biodiversity, eco-toxicology, impact assessment, early-warning systems and bio-monitoring.

The system consists of the following components:

- Multimedia taxonomic database with ecological information on aquatic species;
- Computer assisted identification system;
- Database with monitoring data, including species distributions, biological, chemical and physical parameters;
- Meta-database with relevant ecological data, Internet links and information on relevant research data and publications;
- Interactive and dynamic combined spatial and temporal analysis software.

The North Sea Information has been designed to contain both data from on-line measurement systems and off-line data, including historical data sets. The database will include biological monitoring data on a wide range of organisms and analysis techniques, including flow cytometry. Flow cytometry provides quantitative particle analysis that can be translated into detailed taxonomic information, thus permitting the precise, rapid and repetitive description of the population structure of primary producers, bacterioplankton and zooplankton. Results of the EC-project AIMS will be presented.

The existence of many data sets, which are not directly compatible, hinders the use of monitoring data by decision makers. A specific software module on biological monitoring data together with a module on chemical and physical parameters will allow making various data sets compatible and enables the automated import of validated data. Examples for this system are the use of data from taxonomists who have different taxonomic concepts and the inter-calibration of data from various flow cytometers.

The North Sea Information System will be presented with emphasis on use, application and portability to other aquatic ecosystems.
REVIEWED RIVER GUIDELINES: MAJOR CHANGES AFTER 4 YEARS OF EXPERIENCE WITH PILOT PROJECTS UNDER THE UN/ECE WATER CONVENTION

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Under the UN/ECE Water Convention the "Guidelines on water-quality monitoring and assessment of transboundary rivers" were adopted in 1996. After a period of four years, substantial additions to the guidelines were necessary. Reasons include important international strategic and scientific developments during the past few years. There was also considerable experience gained on best practices to implement monitoring and assessment activities under the Convention. Besides, particular attention was given to the further development of recommendations for monitoring and assessment of water-quantity and water and health aspects.

The findings of pilot projects on monitoring and assessment of transboundary rivers were included in this revision. In 1997, the task force on monitoring and assessment under the Convention started the implementation of the 1996 Guidelines in a pilot project programme which covers eight river basins in the UN/ECE region. Although the completion of this programme will take two more years, the findings of the various projects have already led to substantial modifications of, and additions to, the guidelines.

More attention has been given to the analysis of water management issues in river basins, as the outcome of this analysis defines the scope of environmental information which is relevant for the respective transboundary river and its catchment area. In transboundary river basins, there is generally an urgent need for good practices to identify problems and to find the cause-and-effect relations concerning pressures and transboundary impact. The role of inventories and preliminary surveys has been further elaborated, as these activities may form excellent tools in problem analysis.

Special attention has been given to the evaluation of legislation. In the UN/ECE region, different assessment methods and classification systems are used by riparian countries. Harmonisation of water-quality criteria and targets will lead to a better evaluation of the scientific basis of the different classification systems. Comparison with internationally accepted risk assessment criteria is required. There is also a need to compare the existing national legislative systems with recent developments in EU legislation, for example, the draft Directive establishing a framework for Community action in the field of water policy.

The role of indicators in environmental information has been emphasised. Environmental information should not only focus on the state of the transboundary river, but also on the pressures and the driving forces which constitute the actual and future state of the river and/or its catchment area. In addition, information on the impact of the state and the response of the society form indispensable elements and are politically relevant for decision making. These considerations were brought in line with recent developments by leading international institutions, such as the European Environment Agency (EEA) and the Organisation for Economic Co-operation and Development (OECD).

Recommendations on approximate calculations (estimates) of loads from point sources and diffuse sources have also been incorporated as such estimates are of utmost importance for the receiving waters (lakes, estuaries, seas) as well as for pollution abatement strategies in river basins. Finally, the institutional aspects have been further elaborated, and the links to the relevant provisions of the Convention were more developed.
EGYPTIAN EXPERIENCE IN DEVELOPING WATER QUALITY MONITORING STATIONS

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In the face of rapidly increasing irrigation demand, the available water supply in Egypt is supplemented by the reuse of agricultural drainage water, which is often of lower quality. Around 10% of the water distributed in the Nile Delta is already reused drainage water. It is planned on short term to double this amount after the implementation of the new extension project. This led to the importance of monitoring the quality of drainage water.

The growing concern of water quality monitoring, as a function of society's efforts to manage the environment and the increasing threat of drainage water pollution called for the inclusion of other water quality variables. In 1996, an upgrade of the monitoring program was made. Continuous Water Quality Monitoring Stations have been installed at some strategic locations according to specific criteria. The variables monitored are Temperature, Turbidity, Dissolved Oxygen, pH, and Electrical Conductivity. Currently, DRI is in the process of upgrading the present stations to be on-line by use data collection platform. This paper discusses and evaluates the selection criteria, implementation, and operation of water quality-monitoring stations under the Egyptian conditions.

INTRODUCTION

Since 1983, the Drainage Research Institute (DRI) of Egypt, in co-operation with Dutch experts, has developed, maintained and upgraded a drainage water quality-monitoring program to provide reliable information following standard methods and procedures. The drainage water quality-monitoring network initially consisted of about 20 open locations and 70 pump stations. The instrumentation of the network started in 1984 to enable a continuous and permanent monitoring of discharge and salinity. The station instrumentation includes time and level gauges for the pump stations. Level gauges, EC recorder and velocity/level recorders have been installed at the open locations. A special velocity recorder ENDECO for continuous recording of flow velocity, flow direction, conductivity and temperature of water was installed at the drain outfalls where backwater effects were expected.

Monitoring of drainage water quality is an essential activity to obtain information about the physical, chemical and biological characteristics of drainage water. The usefulness of the information is highly dependent upon a monitoring program, or system, being properly designed and operated. In the Nile Delta, about 140 locations are monitored on a monthly basis through the water quality-monitoring program developed and operated by the Drainage Research Institute (DRI) with the assistance of the Netherlands government.

Four Continuous Water quality Monitoring Stations (CWQMS) have been installed at four strategic locations to have a better understanding of daily variation of the selected water quality variables. The objective of this paper is to present the Egyptian experience in the installation and operation of the CWQMS.

DESCRIPTION OF THE CWQMS

The continuous water quality monitoring stations are placed at four main open drains at the intake of the existing reuse pump stations. The CWQMS includes the following sensors: pH (acidity), DO (dissolved oxygen), electrical conductivity (soluble salts), turbidity (transparency), and temperature. The measured values are stored in a data logger (digital) and on a recorder (paper). Figures on paper are easy to oversee without a computer; digital values can be downloaded and used in calculations.

All the equipment is mounted in a locked steel container, which contains the measuring unit, two suction pumps and a measuring tank, in which the sensors are placed. The two pumps are operating alternatively, switched by a timer, to prevent overheating of the pump as shown in Photos 1 to 4.
SELECTION CRITERIA FOR THE EQUIPED SITES

Due to the limited number of stations compared to the total number of monitoring sites of the drainage water-monitoring network, selection criteria were used to select the most important strategic locations. The selection criteria of implementation of CWQMS were as follows:

- Strategic point with a potential for reuse;
- Degree of variation of water quality in the drain; and
- The capacity of the reuse project.

Using the above criteria the most four strategic locations (Figure 1) have been chosen:

- Intake of Salam 3 pump station at Bahr Hadus drain;
- Intake of Hanout Irr. pump station at Bahr Hadus drain;
- Intake of Hamul Irr. pump station at Gharbia main drain; and
- Intake of Edko Irr. pump station at Edko drain.

PREPARATION PHASE

Bench testing is generally the most time-consuming component of the deployment routine. However, this is an essential step to ensure that all programming and connections are functioning prior to moving to the field. Bench testing equipment involves:

- Checks whether the equipment is properly connected;
- Tests for equipment functionality; and
- Assessment of the operation of software programming.

It is imperative that all hardware is appropriately programmed, wired and prepared. The time taken to ensure the functionality of equipment and software will save time and frustration in the field (Photo 5 and 6).
5. CONSTRUCTION PHASE

Through the agreement between the Drainage Research Institute and the Mechanical & Electrical Department, the electric supply is provided to the CWQMS from the reuse pump stations. The steel container is placed on a concrete foundation plate and equipped with the twin suction pumps, intake and off take pipes, measuring tank, water quality sensors, and the station itself. The steps of construction are shown in Photos 7 to 12.
OPERATION PHASE

Inspection, maintenance, calibration and collection of data need to be done on a regular basis. Inspection is done on a daily basis to keep the sensors in a good condition. The calibration, maintenance, and collecting data are done every two weeks. The operational team must have experience to evaluate the following points:

- Is the calibration frequency for pH and oxygen sensor sufficient;
- Are sensors dirty every time and are the stored data not right;
- Are the digitally obtained data sufficient;
- Are the values in phase with processing needs; and
- Is the paper roll sufficient for the desired period.

Service and Maintenance Cycle

A maintenance schedule is planned for the monitoring program. Establishing a schedule for regular station and instrument service as part of the planned quality assurance and quality control measures is essential. Exact timing depends on site-specific factors, but the plans are
initially based on a daily service schedule for the sensors. Service functions include:

- Physical cleaning of instruments (Photos 13,14); and
- Inspection for fouling, corrosion or damage.

Downloading of data is done on a monthly basis (to lap-top computer).

The following activity is done according to the conditions of each station:

- Replacement of instruments for calibration; and
- Replacement of defective components.

**Field sampling**

As part of the water quality-monitoring program, water samples are taken on a regular basis at the same locations of CWQMS and sent to the laboratory for analysis and/or measurements made with portable meters. The result of the analyses are recorded and entered on a database with, or linked to the automatic sensor results (Photo 15).
Documentation and Data Management

Quality assurance and quality control of automated data depend to a large extend on the quality of the documentation surrounding calibration logs, procedures, field notes, and area events. The following is an outline of the basic information that is kept on file for data management and storage.

Documentation

For each site a file is maintained to provide information on the equipment at the site, the maintenance and service records. At minimum for each site the following are kept on file:
- Equipment and sensors manuals. Complete at the time of set-up of the equipment and sensors.
- Modification or damage form. If any modification is made to a site or damage is noted on of these forms (e.g., if a sensor is replaced) is completed.
- Station log. The log is kept at the site, where each visit is recorded.
- Field form. With each site visit one of the field forms is completed.
- Other relevant notes are included in the file.

Downloading Frequency

Downloading frequency is an integral part of the QA/QC program, because it allows one to identify data gaps, unusual reading and unusual patterns of one or more variable. If gross errors in data exist they may be apparent when plotted, and subsequently problems within the system can be identified. For example, sensors may need field recalibration or manufacturer/ laboratory calibration. Data downloading is done on a monthly basis.

Data Management

All data captured by the automated monitoring station is stored in EXCEL spreadsheet program. This includes sensor readings and graphs. The following procedures are followed.
- Review data graphically
- Define all date and time ranges that are considered to be anomalous data and reasons

The comments about the sensors are entered and the reasons to consider data anomalous include:
- Periods when it was known that sensors were obstructed, or operational requirements were not met, and therefore were returning incorrect data
- Data spikes caused by the operator not instruments off line prior to servicing or instrument performance testing.

OUTPUT SAMPLE OF CWQMS

Around one year data recorded through the WQMS are presented in this paper. The recorded data is covering the seasonal and daily variation of the water quality at different monitoring sites. The water quality is affected by crop rotation especially in the summer with high drainage rate resulting from rice zones. The maintenance of the drainage system can also affect the water quality due to the bed drain sedimentation. The data recorded each hour by the WQMS is collected from twelve measurements.

Temperature

Water bodies undergo temperature variations along with normal climatic fluctuations. The daily and seasonal temperature variations are given in Figure 2. The obtained results from continuous water quality monitoring station (CWQMS) at Edko reveal two points as following:

1. There is daily variation during the monitor period (October 6, 1999 to August 24, 2000). This daily variation is a marked increase in the summer and this variation is around 4°C.
2. The seasonal variation for the same period is higher than the daily variation. The temperature difference between winter and summer is about 15°C.

Dissolved Oxygen (DO)

Oxygen is essential to all forms of aquatic life, including those organisms responsible for the self-purification processes in natural water bodies. The oxygen content of natural water varies with the temperature, salinity, turbulence, the photosynthetic activity of algae and plants and atmospheric pressure.

Figure 2 illustrates the variation in recorded DO throughout the same period. The obtained results reveal several points of interest as following:

1. The peaks shown in figure 2 are due to technical problems;
2. The lowest recorded values are caused by cleaning problems with the DO sensor;
3. Dissolved oxygen values ranged between 0.5 and 2.5 mg/l. The daily variation of DO concentration reveal opposite temperature-oxygen relationship which may be due to different aquatic activity (photosynthesis and respiration for algae and plants); and
4. Seasonal variation in DO cannot be detected over the monitoring period.

pH, acidity and alkalinity

The pH is an important variable in water quality assessment as influences many biological and chemical processes within a water body and all processes associated with water supply and treatment. Data in Figure 3 illustrate the values of pH during the monitoring period. The obtained data reveal that:

1. In general, the variation of pH values ranged between 7 and 8;
2. The pH values are slightly higher in the summer than the winter; and
3. The pH values lower than 7 reflect periods when there was need for physical cleaning of pH sensor and calibrations.
Conductivity

Conductivity, or specific conductance, is a measure of the ability of water to conduct an electric current. It is sensitive to variations in dissolved solids, mostly mineral salts. Figure 4 represents the variations of EC values, and this variation ranged between .94 and .98 dS/m over the monitoring period. This obtained result reveal to the main point, that continuous monitoring of EC at that location is not important.

Figure 3 Typical CWQMS output of ph (Edko reuse pump station from October 99 to August 2000)

Figure 4 Typical CWQMS output of Electrical Conductivity (EC) (Edko reuse pump station from October 99 to August 2000)
Turbidity

Turbidity should be measured in the field because settling during storage, and changes in pH can lead to precipitation and affect the results. This is the reason for the continuous turbidity monitoring in the field.

The variations of turbidity values are given in Figure 5. The obtained data show that:
1. The appearance of the flat peaks was due to the need for physical cleaning of the turbidity sensor;
2. There are dramatic changes in the monitored turbidity values. This finding is probably due to the effect of drain maintenance and the operation of Edko reuse P.S.; and
3. The continuous monitoring of turbidity at this location is important due to the high variation in the turbidity.

![Figure 5 Typical CWQMS output of Turbidity (Edko reuse pump station from October 99 to August 2000)](image)

CONCLUSIONS AND RECOMMENDATIONS

The CWQMS is good tools support the routine monitoring program where outputs of the CWQMS gave a better understanding of the water quality daily variation. Moreover, it can give a good impression about the initial setting for the required sampling frequency in the design phase of the monitoring program.

For the equipped sites, turbidity and temperature show a daily and seasonal variation while the others show more or less variation within very narrow ranges.

The operational results of the CWQMS showed that these stations should be designed in such a way to be movable after good understanding of monitored water quality variation at the equipped site.
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INTEGRATED ASSESSMENT OF RIGA INLAND WATERS

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The location of the Riga City on the estuary of the Daugava River and in the area of the Baltic Sea determines the obligation of the Riga City Council to integrate its water quality protection measures and water resource management activities on both the national and international level. The Riga City Council’s policy, directed at the involvement of both scientists and citizens in the water quality assessment, allows taking decisions about water resources management considered from every point of view. This paper aims to describe the work of the Environmental Department of the Riga City Council, carried out to support the decision-making process of the Riga City Council in the area of the inland waters protection and sustainable management.

INTRODUCTION

The Riga City Council as the main policy-maker in the city has the overall responsibility for the environment. Responsibilities concerning water bodies include the prevention of water pollution, development of water supply and sewerage, promotion of the recreational use of water bodies, monitoring and supervision of water resources.

In 1995, the monitoring network was designed to cover the small tributaries of the Daugava, lakes and artificial water reservoirs within the administrative territory of Riga (Figure 1).

Figure 1: Location of the observation sites

Monitoring of the observation sites from number 3 to number 20 was focused only on the hydrochemical parameters and, mostly because of the limited financial resources, it was limited to a few general variables, specific ions, nutrients and some heavy metals. It was sufficient to have a general overview of the Riga inland water quality and to trace trends in the water quality. To obtain more comprehensive background information on the biological quality of
Lake Kisezers and Lake Jugla, bottom fauna, zooplankton and habitat surveys were also included in the monitoring programme of the observation sites number 1, 2, 21, and 23.

Such broad-scale observations, carried out during 1996-1998, enable us to obtain a very comprehensive view of the occurrence of organic contaminants and biogens and revealed an urgent need to take all appropriate measures to treat municipal sewage and industrial wastewater with a special emphasis on the reduction of discharges of nutrients (Figure 2):

![Figure 2: Annual mean concentrations of total phosphorus and nitrogen in Riga water bodies](image)

Public lakes Jugla and Kisezers were identified as the key area for concern. Lakes are shallow, they suffer from the super-abundance of nutrients and organic substances, and they have considerable biodiversity of aquatic species and high relative abundance of the algal flora, zooplankton and zoobenthos species. The functions like recreation, aquatic wildlife and fisheries are important in both lakes.

**Interdisciplinary investigation of Lake Jugla**

In 1999, the work started including comprehensive scientific studies of Lake Jugla, covering theory, targeted research, monitoring and modelling (Figure 3). Integrated assessment is used in the evaluation of the ecological integrity, in which both the biological condition and the habitat quality are evaluated. The programme contains also the sediment investigation, biological tests and control of bacteria-plankton, phyto-plankton and benthos.

The research project focused on:
- impact studies;
- trend analyses;
- multi-scale appraisal.
This project was conducted by the Environmental Department of the Riga City Council to evaluate the ecological conditions in Lake Jugla and to provide the Riga City Council recommendations for the pollution prevention, sustainable water resources management and economic activity restrictions in the catchment area. Scientific – consulting company Geo-Consultants Ltd, chosen by way of tendering, performed the project. The leading scientific institutions from Latvia were involved.

The Riga Regional Environmental Board the Ministry of Environment and Regional Development of Latvia approves the monitored variables and sampling frequencies within the framework of the Integrated Investigation Programme for Lake Jugla.

Assessment objectives:
- examine the ecological quality of Lake Jugla and adjacent land area
- detect the trends in water quality related to economic activities and to the influence of the leakage basin
- determine the suitability of the Lake Jugla for fisheries and other types of use
- obtain reliable estimates about the effect of certain protection measures
- obtain a representative picture about the extent of effects caused by different types of land use
- gain easy understandable information in a simple and meaningful way using tables, graphs and maps.

The sampling locations are chosen to represent Lake Jugla (Figure 4). The analyses are performed by the Laboratory for the Latvian Hydrometeorological Agency in Riga and by the ACME Laboratory in Ontario (Canada).
Table 1 Comparison of Lake Jugla sediment quality with Quality Standards regarding to EU directive requirements

<table>
<thead>
<tr>
<th>Chemical elements</th>
<th>Unit</th>
<th>Highest measured concentration</th>
<th>Sample Nr.</th>
<th>86/278/EEC normative</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cadmium</td>
<td>mg/kg</td>
<td>0.34</td>
<td>27</td>
<td>1</td>
</tr>
<tr>
<td>Copper</td>
<td>mg/kg</td>
<td>14.1</td>
<td>12</td>
<td>50</td>
</tr>
<tr>
<td>Nickel</td>
<td>mg/kg</td>
<td>16.9</td>
<td>3</td>
<td>30</td>
</tr>
<tr>
<td>Lead</td>
<td>mg/kg</td>
<td>21.3</td>
<td>14</td>
<td>50</td>
</tr>
<tr>
<td>Zinc</td>
<td>mg/kg</td>
<td>64.2</td>
<td>14</td>
<td>150</td>
</tr>
<tr>
<td>Mercury</td>
<td>mg/kg</td>
<td>0.092</td>
<td>27</td>
<td>1</td>
</tr>
</tbody>
</table>

Table 2 Number of zoobenthos organisms in Lake Jugla in 1963 – 1998. (pieces/m²)

<table>
<thead>
<tr>
<th>Years</th>
<th>Chironomidae</th>
<th>Malacostraca</th>
<th>Mollusca</th>
<th>Oligochaeta</th>
<th>Varia</th>
</tr>
</thead>
<tbody>
<tr>
<td>1963</td>
<td>679</td>
<td>260</td>
<td>1136</td>
<td>195</td>
<td>138</td>
</tr>
<tr>
<td>1969</td>
<td>233</td>
<td>109</td>
<td>2736</td>
<td>1454</td>
<td>164</td>
</tr>
<tr>
<td>1978</td>
<td>148</td>
<td>20</td>
<td>20</td>
<td>1348</td>
<td>64</td>
</tr>
<tr>
<td>1982</td>
<td>1278</td>
<td>10</td>
<td>214</td>
<td>1498</td>
<td>272</td>
</tr>
<tr>
<td>1991</td>
<td>6710</td>
<td>90</td>
<td>800</td>
<td>2670</td>
<td>220</td>
</tr>
<tr>
<td>1998</td>
<td>400</td>
<td>1258</td>
<td>883</td>
<td>225</td>
<td>142</td>
</tr>
</tbody>
</table>

Figure 4: Location of the geoecological sampling sites in the Lake Jugla aquatic area

The monitored parameters include eutrophication parameters, heavy metals (table 1), organic pollutants, hydro-biological (table 2) and microbiological parameters. For physico-chemical and biological water quality assessment, a classification system and indices according to Surface Water Quality Requirements in Latvia are used. The categories describe the ecological quality - from good ecological quality to high ecological quality, depending on the defined water quality targets.
The information on the concentration of pollutants in effluents was obtained from reports provided by facilities. The Latvian Environmental Protection Law regulates the effluent monitoring. The obligation for self-monitoring is laid down in the industrial discharge consents. Facility operators are responsible for the self-monitoring of waste water discharge. Variables are selected on the basis of the expected pollution.

A very important aspect to be covered is the information to be gained from hydrological investigations, which includes information about water flow velocities and direction, and water balances. The dynamics in hydrology and morphology, together with spatial differences in geology, relief and soil in the catchment area were also included into the investigation programme because the aquatic area ecosystem is interrelated with the littoral area (banks, the amphibious area) and the terrestrial area (floodplains). Certain natural events such as floods and other changes in the water level and the impact done by the sluice on the Jugla Canal have been considered.

The fishery characteristics of the lake include spawning grounds, feeding basis for fish, fisher areas with stationary nets, migration paths of the migratory and semi-migratory fish.

The transport of the suspended and soluble matter from the land area to Lake Jugla has been investigated in order to study the effects of changes of the silvicultural and agricultural land use upon the material transport in small drainage areas of streams discharging in the lake.

#### Table 3 Concentrations of biogens in Lake Jugla and its catchment area (1999)

<table>
<thead>
<tr>
<th>Parameters</th>
<th>Unit</th>
<th>Concentrations in drainage area</th>
<th>Concentrations in Lake Jugla</th>
</tr>
</thead>
<tbody>
<tr>
<td>BOD5</td>
<td>mg/l</td>
<td>1,3 -1,6; 1,7</td>
<td></td>
</tr>
<tr>
<td>COD</td>
<td>mg/l</td>
<td>41 – 67</td>
<td>48</td>
</tr>
<tr>
<td>N-NH4</td>
<td>mg/l</td>
<td>&lt;0,1</td>
<td>1,09</td>
</tr>
<tr>
<td>N-NO2</td>
<td>mg/l</td>
<td>&lt;0,011 - 0,013</td>
<td>0,29</td>
</tr>
<tr>
<td>N-NO3</td>
<td>mg/l</td>
<td>0,42 - 0,89</td>
<td>2,84</td>
</tr>
<tr>
<td>Oxygen</td>
<td>mg/l</td>
<td>8,2 – 9,9</td>
<td>7,1</td>
</tr>
</tbody>
</table>

The public opinion poll as one of the ways to identify environmental problems was carried out. Results were presented in clear, easy to understand graphs and maps. The visualisation of data allows involving the public, entrepreneurs and interest groups, and letting them take part in the process of the local decision-making.

**RESULTS OBTAINED AND DECISIONS MADE**

As a result of the studies of the landscape-ecological conditions of Lake Jugla, the territorial zoning, based on ecological properties, regarding the degree of economic use and natural protection was conducted.

The zone A: highly limited use, it occupies the areas of the lake with exclusive significance for biological diversity of the lake biocenoses, wetlands and wet forests, water birds nesting and feeding places.

The zone B: lowland meadows and grass-bog, habitat for rare species of flora and fauna, significant for biological diversity, limited recreational activities are allowed.

The zone C: recreational activities and limited economic use is allowed.
On the basis of the information obtained during the geological exploration and the results of sample analyses, the Quaternary Geological Map of the Lake Jugla area with scale 1:25 000 was prepared for the first time. Quaternary deposits form an almost uninterrupted cover of various thickness in the area of Lake Jugla. The base of the lake deposit consists of grey silty sand with the density of approximately 2m. The upper part is covered by a muddy saprophytic deposit with the density of 3 to 5m. The results of the studies of the Quaternary section and the upper part of the bedrock are shown in the chart below:

<table>
<thead>
<tr>
<th>Object to be assessed</th>
<th>Findings</th>
</tr>
</thead>
</table>
| Water quality         | - some heavy metals, nutrients and organic matter exceed the limit values laid down in the Surface Water Quality Requirements in Latvia;  
                      | - the oxygen regime is favourable;  
                      | - the most suitable places for swimming were selected, where water quality is in accordance with the EU Bathing Water Directive 76/160/EEC requirements, bank areas are sandy, dunes covered with pinewoods |
| Bottom sediments      | - consists of muddy sand, saprophytic mud, peat;  
| Leakage basin         | - the impact caused by human activity in the catchment area do not cause significant adverse effects upon flora and fauna, landscape and historic monuments;  
                      | - the input of nutrients (phosphorous) with waste waters exceed the limit;  
                      | - diffuse contamination from urban run-off and agricultural lands caused pollution of the bottom and water body itself |
Fishery
- migratory fish movements are prevented by the sluice on the Jugla Canal;
- the amount and structure of fish population allows professional fishing;
- spawning and feeding places are in satisfactory condition and amount

Water flow and water level
- the sluice on the Jugla Canal has lost its role of protection lake water basin against the inflow of the salty Baltic Sea water, it should be demolished;

Overgrowing degree and dominated water plants
- the estimated overgrowing degree reaches 80% of the total aquatic area;
- water plants are not rich in species – reed, bulrush coltsfoot dominated;

Aquatic and littoral flora and fauna
- several protected plants, birds and animals, included in the Latvian Red Book, and outstanding trees are identified;
- diversity of invertebrate communities (planktonic and bottom-dwelling);
- diversity of the aquatic plant communities;
- diversity of the higher vertebrate community (amphibians, reptiles, birds and mammals)

Table 4: Evaluation scheme to characterise the major impact upon Lake Jugla

<table>
<thead>
<tr>
<th>Decision based on integrated data assessment</th>
<th>industry discharge</th>
<th>building</th>
</tr>
</thead>
<tbody>
<tr>
<td>restriction determination for</td>
<td>recreational activities</td>
<td>fishery</td>
</tr>
<tr>
<td>adoption of the Riga Environmental Strategy</td>
<td>establishing of bank protection belts</td>
<td>establishing of bathing places</td>
</tr>
<tr>
<td>termination of drinking water intake from Lake Jugla</td>
<td>sluice demolition on the Jugla Canal</td>
<td></td>
</tr>
<tr>
<td>sewerage input prohibition in Lake Jugla</td>
<td>arrangement of the protected areas in Lake Jugla</td>
<td></td>
</tr>
</tbody>
</table>

Figure 6: Decisions based on the interdisciplinary investigation of Lake Jugla

CONCLUSIONS
Investigations, done in 1999, provide an integrated picture of the Lake Jugla water quality and ecological condition of the water body itself and the leakage basin of the lake and the discharged small rivers.
For the effluent sampling and testing, the monitoring parameters are determined by legislation and regulation, and by taking into account the presence of any specific pollution. For surface water monitoring, the same factors are also important, but the local aspects, such as the function of the water body, are of significant importance and should be used to adapt the choice of the required monitoring parameters.

The quality of the aquatic ecosystem as a whole, including the water area, water bottom or sediment and the terrestrial areas, and the animal and plant communities represented there, were assessed as valuable in terms of the biological diversity, as appropriate for recreation, including bathing, and as vulnerable against the anthropogenic load.

Based on the large number of different tests described in the assessment report that was prepared by the responsible executor Geo-Consultants Ltd, the draft guidelines for the Lake Jugla management and quality assurance is currently under preparation. Tests that are considered to be effective indicators of water quality conditions will be incorporated into the Riga inland water monitoring programme. The introduction of the monitoring programme is set as the target goal in the Riga Environmental Strategy adapted by the Riga City Council.

The local long-term action plan, namely, the Riga Environmental Strategy, was adapted by the Riga City Council this year with a view of providing guidance to planners and decision makers in the formulation of municipal policies for the sustainable use of nature resources. The information about the ecosystem performance under natural conditions will have to be used to define the management targets. The water management targets will have to include both the ecological targets and functional (or use related) targets, that are linked to each other in a logical and coherent way to avoid conflicting management.

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INTRODUCTION

Since 1993, the Regional Directorate of Rijkswaterstaat IJsselmeergebied (RDII) conducts an ambient monitoring programme additional to the national ambient monitoring programme which is conducted by RIZA. In 1998 the RDII has asked RIZA to start an evaluation- and optimisation study of their monitoring programme. The objective of the study was to obtain an optimised monitoring program which provides the necessary information for water management.

In this study the "monitoring cycle" has been used as a guidance for the different steps in the study (figure 1). The cycle describes the process from specifying information needs and a monitoring strategy to collecting information through data/sample collection and data analysis up to utilisation of the resulting information. The cycle was developed in the preparation of the "Guidelines on water-quality monitoring and assessment of transboundary rivers" by the UN/ECE Task Force on Monitoring and Assessment in 1996 [UN/ECE, 1996].

![Figure 1. The monitoring cycle (UN/ECE, 1996)](image)

More information and less costs

The monitoring process of the RDII has been assessed following the different steps of the monitoring cycle and recommendations have been given for each step. This resulted in a monitoring program which provides more information at lower costs.

Recommendations from the study

Information needs
- The methods used for the inventory of the information needs (towards water management) are useful. However, more possibilities for the translation from information needs to network lay-out are needed.

Monitoring strategy
- Stop the routine monitoring of micropollutants on regional locations, but execute specific surveys for the specified information needs for micropollutants here. The surveys will function as an important state-indicator supplementary to the trend-indication from the remaining national locations.
Network design
- Statistical research showed that higher sampling frequency at less locations provides more information. It was advised to bring down the number of locations with almost 50% and at the same time increase the sample frequency to at least 12 times a year at every location (figure 2)

Laboratory analysis
- It is cost-effective to select parameters which can be analysed with the same analytical method, in one run.

Data management (Data handling, Data analysis)
Most of the conclusions and recommendations for this step are institutional and organisational:
- Monthly monitor the execution and progress of the programme. In addition a yearly verification of completeness and likeliness of the data is recommended.
- Start a regular users-meeting to list problems and wishes and as a tool for communication on progress and changes in the programme.
- Describe the competence's, methods and procedures used in the process and lay them down in a so-called "procedure-book".

Figure 2: IJsselmeer Region map
? = the study has tried to predict the water quality at the locations marked with an '?' from the other locations
VROUWZD (IJ23) = name of sampling location
= the correlation between the locations within this area was studied
Reporting and information utilisation
The study revealed that the data were insufficiently used for reporting and information supply. It was recommended to report on the data on a regular (yearly) basis in an independent report on the status of the aquatic environment. This would strongly improve the information supply but also function as a means of controlling and managing the monitoring programme.

CONCLUSIONS

- The cost-reduction for the network, through this optimisation is estimated at about 35-50%.
- The monitoring cycle is an effective tool to structure evaluation and optimisation studies of monitoring programmes.
- Most difficult phase in such a study is the specification of information needs. The methods applicable for this step need further development.

REFERENCES

WATER QUALITY MONITORING IN THE DANUBE DELTA

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The Danube Delta, one of the most important wetland in Europe, is stretching over a huge area of more than 5,000 square kilometres, channelling the water of the whole Danube Basin into the Black Sea. The network of canals, lakes, reed swamps and forests, meadows, sandy grasslands, dunes as well, form an unique mosaic of terrestrial and aquatic habitats, where the biodiversity is high. The necessity of monitoring the water quality in the Danube Delta arises from the main goal for which this area was designed as a Biosphere Reserve by the Romanian Government in 1990: protection and conservation of this natural heritage. In this context, the water quality monitoring programme in Danube Delta is part of the Environmental Programme for the River Danube Basin. The water quality monitoring in Danube Delta has the following functions: determine the fluctuations and the long-term trends in the ecosystem's status and test for compliance with standards. Two main problems have been emphasised: discharge of different pollutants and eutrophication. Chemical and biological indicators relevant to these aspects have been selected, based on consideration of ecological relevance. The monitoring programme was carried out in different sites, with the sampling frequency depending on the programme goals. All the information obtained from the monitoring system has maximum importance in the management of the Danube Delta Biosphere Reserve. On the basis of knowledge gained since started in 1990, the monitoring programme should be optimised. The results of three years monitoring (1997-1999) and the necessary activities to improve the quality of data for the benefits of water management actions are presented.

Keywords: Danube Delta, monitoring, water quality assessment

INTRODUCTION

The Danube Delta is one of the most important wetlands in Europe, comprising 23 types of ecosystems that are natural or partly modified by man and 7 types of man-made ecosystems (Gâstescu et al.1999). All these ecosystems shelter a large variety of habitats for flora and fauna. Due to the high species richness, the Danube Delta Biosphere Reserve (DDBR) occupies the third place in the world, after the Amazon and the Nile Delta. Since 1990 the Danube Delta is included in the World Natural Heritage List, in the RAMSAR Convention List and in the ,,Man and Biosphere,, UNESCO Programme. Even a great part of the Danube Delta's territory is under a ,,natural,, status, human intervention often caused habitat deterioration and loss of species, especially over the last decades. The main factors which have changed the natural status are: hydrotechnical works within the Danube Delta for improving water circulation and navigation, dams construction upstream which limited flooding areas, increasing of nutrient levels in water (eutrophication), creation of agricultural and fish polders.

The water quality monitoring in the Danube Delta starts in 1977 by the Environment Protection Agency and includes 5 stations located on the Danube branches (Chilia, Sulina, Sf. Gheorghe). Regular measurements of suspended solids and dissolved nutrients for lakes started in 1980 and from 1996 to 1998 the number of sampled lakes increased to 18. The main goal of the water quality monitoring in the Danube Delta is to provide the required information for the water management, aiming the conservation and protection of DDBR. The water system is analyzed not only as water resource, but also as a complex life support system providing many renewable resources.

METHODS

The monitoring network includes 31 sampling points, 9 of that are located on the Danube branches, 4 on main channels within the Danube Delta and 18 are sampling points in lakes. The selection of sampling points was based on the types of aquatic ecosystems, aiming their ecological significance (figure 1). The indicator variables were selected in accordance with the
Table 1 Parameters monitored in the Danube Delta

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Danube branches and channels</th>
<th>Lakes</th>
<th>Water column</th>
<th>frequency</th>
<th>Water column</th>
<th>frequency</th>
<th>Bottom</th>
<th>frequency</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. general (physical and chemical)</td>
<td>Temperature pH, EC, DO</td>
<td>monthly</td>
<td>Depth, Secchi depth, Temperature pH, EC, DO</td>
<td>monthly</td>
<td>pH, EC</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2. nutrients</td>
<td>inorganic-N Total - P</td>
<td>monthly</td>
<td>inorganic-N Total - P</td>
<td>monthly</td>
<td>inorganic-N Total - P</td>
<td>yearly</td>
<td></td>
<td></td>
</tr>
<tr>
<td>3. inorganic</td>
<td>Heavy metals Anion Cations</td>
<td>Three times/year</td>
<td>Heavy metals Anions Cations</td>
<td>Three times/year</td>
<td>Heavy metals Anion Cations</td>
<td>yearly</td>
<td></td>
<td></td>
</tr>
<tr>
<td>4. organic</td>
<td>COD organochloric pesticides petroleum hydrocarbons</td>
<td>Three times/year</td>
<td>COD organochloric pesticides petroleum hydrocarbons</td>
<td>Three times/year</td>
<td>pesticides organic C</td>
<td>yearly</td>
<td></td>
<td></td>
</tr>
<tr>
<td>6. biological</td>
<td>Chlorophyll a Phytoplankton Zooplankton Macrophytes</td>
<td>monthly</td>
<td>Macrozoobenthos</td>
<td>monthly</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Figure 1
monitoring objectives. Sampling methods and laboratory analysis are standardized. The sampling frequency of the selected parameters depends on the type of ecosystem (Danube River, channels, and lakes) - table 1. The physical variables selected to assess the water quality of aquatic ecosystems are: temperature, water depth, Secchi depth. These parameters are monitored monthly and have a high seasonal variability, but are very useful in assessing the ecosystem trophic state. Chemical monitoring is used for identifying and quantifying the major contaminants, for testing the compliance with water quality standards or for monitoring levels of parameters (e.g. nutrients) of importance for ecological assessment. Beside the main chemical parameters (pH, EC, DO, COD-Mn), a set of other chemical variables are selected to be monitored with different frequency in Danube, channels and lakes. Nutrients (mineral nitrogen, total phosphorous) are analysed monthly from the water column samples in all the ecosystems. The nutrient content of sediments is analysed once per year. The inorganic compounds are monitored three times per year in all the stations in water and once per year in sediments. Since it was designed to obtain a comprehensive view on the occurrence of those organic compound which may contribute to the general toxicity of water, even at low concentration, non-volatile chlorinated pesticides (DDT, HCH) are analysed in all the sampling points for water (three times/year) and sediments (once/year).

RESULTS AND DISCUSSION

Programme structure

The water quality monitoring in the Danube Delta was developed to meet the objectives from the DDBR management strategy: 1) protect and maintain populations of species and habitats with high ecological values; 2) management of the water circulation in the DDBR in order to improve the ecological conditions in lacustrine zones and adjacent areas; 3) ensure the

![Figure 2](image-url)
ecological integrity of strict protected areas. Therefore, the main objectives of the water quality monitoring in Danube Delta are: assessment of the actual status of aquatic ecosystems, regular testing for the compliance with standards, identification of the long-term trends of the ecosystem status. Two main problems have been emphasized by the water quality assessment: discharge of different pollutants and eutrophication. The changes in water quality are effects of drainage, damming and canalisation that have occurred in the last few decades upstream or within the Danube Delta (Vadineanu, 1994).

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. For this reason it is of major importance to sample both the water and sediments. The criteria used to select the key variables to monitor are: proprieties and concentration of chemical substances (such aspects as adverse effects on biodiversity, bioaccumulation, persistence, toxic proprieties) and availability of reliable and affordable analytical methods.

The water quality monitoring in the Danube Delta, is based on physico-chemical and biological monitoring.

**Physico – chemical monitoring**

The hydrology of the Danube Delta is complex and shows a seasonal variation within the year. Maximum water level is recorded from April through June, a minimum in September through November and a small peak in December through January. The hydrological data are incorporated in the hydraulic model using SOBEK programme (Anonymous, 1999) a one-dimensional open-channel dynamic numerical modeling system. SOBEK is also capable to solve equation that describe unsteady water flow, salt intrusion, sediment transport, morphology and water quality.

The Danube discharge follows the same pattern as the levels within the year, but there are differences between the three years (1997-1999) – Figure 2. The year 1997 is characterised by an unusual flood in August, 1998 started dry in May-June and showed a high water peak in October, and 1999 is somewhat similar to 1997, with the highest water peak in May.

![Figure 3](image)

**Figure 3**

![Figure 4](image)

**Figure 4**
hydrology of the lakes within the delta is determined by the hydrology of the Danube river and the connecting network between the lakes and river branches. There are differences between the nutrient concentration (DIN – dissolved inorganic nitrogen, and TRP – total reactive phosphorous) input and output in the three years (Figure 2a and 2b). High negative values of the input/output ratio are explained by the water level fluctuation. During the flood pulse, the nutrient concentration input is decreasing, and the output is increasing by the flushing process. As a general tendency, the concentration of DIN in the Danube river is greater than TRP. Damming the river had as main effect the lost of floodplains together with their buffer capacity (Vadineanu, 1994). As a consequence, the hydrological regimen is controlling directly or indirectly the behaviour of pollutants in abiotic compartments. The nutrient input in the Danube Delta increased considerably, due to the combined effect of the increase in water input and nutrient levels. Based on morphometry, substrate type, hydrological distance from the river, the Danube Delta lakes, are classified in three main categories (Oosterberg, 2000): 1) large and deep lakes, with sand-silt substrate, turbid, without aquatic vegetation, with a low seasonal variation of water depth, (e.g. Rosu lake, figure 4 a, b, c); 2) clear lakes with clay substrate, situated close to the river, with a short retention time and high variability of water level, with abundant aquatic vegetation (Nebunu lake, figure 3 a, b, c), and isolated lakes, surrounded by reedbeds, clear, with abundant aquatic vegetation (Cuibul cu lebede, figure 5 a, b, c). There is a difference between these three types of lakes in nutrient content, as expressed by DIN/TRP ratio (figure 3, 4, 5). The nutrient dynamic in these lakes is connected with the flood pulse in spring and late autumn. During the flood pulse in spring, the DIN/TRP ratio shows high values, explained by the high amount of nitrogen transported by the river. During the summer time, a significant release of phosphorous from sediments is the mechanism that can explain the low values of the DIN/TRP ratio. The chlorophyll a dynamic in the three types of lakes (figure 3a, 4a, 5a), shows that all of them are clear until early June, then the lakes with scarce or without aquatic vegetation become turbid. In the isolated lakes, the reedbeds are efficient removers of nitrogen in spring, but may release significant amount of phosphorus during the summer.
allowing the development of phytoplankton. The heavy metal dynamic has different patterns depending on water level fluctuation. For Cd, the Admissible Romanian concentrations for water (CMA – Ord. Of Ministry of Environment 756/1997) is often exceeded (figure 3b, 4b, 5b). Chlorinated organic pollutants (DDT and HCH) shows a high seasonal variability depending on the input from Danube river (figure 3c, 4c, 5c). In the isolated lakes, with low water exchange, the accumulation of pollutants is higher (figure 5b, 5c).

Biological monitoring

The aim of biological monitoring is to assess the impact of environment on the biotic compartment of aquatic ecosystems. Biological monitoring in the Danube Delta has been focused only in lakes, mostly because of limited financial resources. Phytoplankton, Zooplankton and Macrozoobenthos are sampled monthly from all 18 lakes. For each compartment, community structure indices (abundance, diversity and similarity indices to assess organic, toxic or physical pollution) and indices based on selected species (saprobic indices) are analyzed. Phytoplankton was included in the monitoring programme because algae are the principal primary producers in the open water, making energy available for higher trophic levels. The increase of chlorophyll a shows that an important nutrient quantity (especially nitrogen and phosphorous) is available for phytoplankton growth. In lakes where phytoplankton is abundant, the zooplankton biomass is high as well. Aquatic vegetation is also important to understand the trophic state of lakes. The benthic macroinvertebrates community is considered a good practical tool to assess water quality because the community consist of many representatives from a wide range of faunal orders, spatial and temporal mobility is quite restricted, organisms integrate environmental condition over a long period of time (De Pauw & Hawkes, 1993). Macroinvertebrate community in the Danube Delta is quite scarce in the turbid, eutrophied lakes, without aquatic vegetation, and consists of greater number of species in the vegetated lakes.

Quality assurance:

The investigation quality is achieved by:

- Training laboratory staff
- Using of standards methods for chemical analysis
- Inter-calibration of programmes

CONCLUSIONS

The main processes affecting the water quality in the Danube Delta emphasised by the monitoring system are the eutrophication and pollution with heavy metals, pesticides or other pollutants. These changes affect the water as a complex life support system. Therefore, the assessment of water quality in the Danube Delta calls for an integrated monitoring system, in which the impact evaluation, cause-effect relationship identification through hypothesis testing should be the main requirements.

The identification of the sources of pollutants and loads could exclude the non-significant pollutants, leading to time and costs savings. Improvement of the analytical techniques, and particularly the sampling procedures is needed in the future to obtain information rich monitoring in the Danube Delta at acceptable costs. Higher efficiency in the network design may be achieved by a more specific analysis of information needs. There is a need to develop a closer relationship between water quality monitoring and hydrological monitoring (the station network should be harmonized and hydrological models should be more efficiently used in data analysis). Much more attention must be given to improve reporting and assessment of monitoring data. An early warning monitoring system should be developed in order to prevent the risks of accidental pollution. The extension of biological monitoring for the river branches and channels is also necessary for understanding the complex processes, which take place in the water. Another problem is the lack of clearly formulated goals and objectives of the monitoring system reflecting its close relationship with the environment management system.
REFERENCES


The main objective of the PIONEER project - preparing and integrating presently available technology and methodology in data management, geostatistical and dynamical data assimilation and numerical modelling for simulation and prediction of nutrients in estuaries of European rivers - is presented. A system for the forecast of nutrients in the Odra Estuary - consisting of the Odra Lagoon and the Lower Odra - is described. The idea of this system is to predict changes of nutrient distribution in the Odra Lagoon on a basis of nutrient loads discharged by the Lower Odra, being estimated from the numerical simulation of flows and transport processes in the river network.

INTRODUCTION

PIONEER - Preparation and Integration of Analysis Tools towards Operational Forecast of Nutrients in Estuaries of European Rivers – is one of the MAST III projects founded within EU’s 4th Framework Programme.

The idea of the PIONEER project is to integrate analysis and data management tools in one system being fit for simulating and forecasting of nutrient distribution. Then to apply, to calibrate and to validate this system for chosen European estuaries: for the Odra Estuary - consisting of the Odra Lagoon and the Lower Odra - in the Polish-German border area and for the Ebro Delta in Spain (von Storch, 1999).

The PIONEER consortium consists of GKSS Research Center (Germany), Universitat Politecnica de Catalunya and Universidad Politécnica de Valencia (Spain), Geographical Institute of Copenhagen and Water Quality Institute of Danisch Hydraulic Institute (Denmark), Maritime Research Institute, Szczecin Branch and Technical University of Szczecin (Poland), Netherlands Institute for Sea Research (the Netherlands), Nansen Environmental and Remote Sensing Center (Norway) and Centre de Geostatistique of Ecole de Mines de Paris (France). The scientific coordinator of the project is GKSS Research Center (Germany). The overall management of the project is taken care of by Ocean Sense Ware (Germany).

The project started on September 1, 1998 and will run until August 31, 2001.

THE SYSTEM FOR THE FORECAST OF NUTRIENTS IN THE Odra ESTUARY

Within the frame of the PIONEER project the system for the forecast of nutrients in the Odra Estuary has been proposed and developed. The main objective of this system is to predict changes of nutrient distribution in the Odra Lagoon (Zalew Szczecinski) on a basis of nutrient loads discharged by the Lower Odra, being estimated from the numerical simulation of flows and transport processes in the river network. The simulation is performed for forecasted hydro-meteorological conditions and forecasted nutrient loads coming from the river upstream. In this simulation inflows of nutrients to the Lower Odra from the sources located along the river are taken into account. The idea of the system for the forecast of nutrients in the Odra Estuary is presented in figure1.

In the system as analysis tools have been set up available numerical models of the nutrient dynamics, the hydrodynamics and the advective and dispersive transport of dissolved and suspended matter.

In the system modelling of nutrient dynamics in the Odra Lagoon is performed using the European Regional Seas Ecosystem Model ERSEM, developed by Netherlands Institute for Sea Research (Baretta et al, 1995). This model consists of a pelagic module, a benthic regeneration module and sediment resuspension module. ERSEM has a coarse spatial resolution into boxes. At open boundaries of the model, time series of dissolved and suspended constituents are prescribed. Horizontal transport of dissolved and suspended constituents is calculated on the
basis of the exchange volumes across the box boundaries compiled from flow field produced by a hydrodynamic model. Vertical transport of these constituents is in the form of sinking and sedimentation for suspended and in the form of turbulent diffusion for dissolved ones. ERSEM has been implemented for the Odra Lagoon with spatial resolution into 3, 8 and 12 boxes. Horizontal transport in the model has been driven with the output generated by three-dimensional hydrodynamic model TRIM3D, utilised in GKSS Research Center. This model includes transport of dissolved and suspended matter. For PIONEER purposes in the model the subroutine for calculation of sedimentation and resuspension processes, prepared by Danish Hydraulic Institute, has been incorporated.

In the system modelling of nutrient dynamics in the Lower Odra is not foreseen. It has been accepted that the system simulates only transport of nutrients in the river in order to estimate their loads discharged into the Odra Lagoon.

For the Lower Odra one-dimensional model MODRIM, developed and utilised in Maritime Research Institute, Szczecin Branch (MRI) is used. This model consists of hydrodynamic module, that simulates steady and unsteady flow in the river network, advection-diffusion module and suspended matter module, that is based on the subroutine for calculation of sedimentation and resuspension processes, mentioned above.

MODRIM has been implemented for the Lower Odra from Widuchowa (701.8 km of Odra) to Trzebiez (36.5 km of Swinoujscie-Szczecin waterway) – figure 1. In the model the river network is presented schematically by 51 sections and 36 nodes. Particular sections of the computational network relate to the natural ones or are artificially separated from longer sections of the river. The nodes relate to the connections of particular sections. The computational network is composed of 350 cross-sections. The whole river network is considered as a planar digraph (Ewertowski, 1998).

The boundary conditions of the model are respectively:
- hydrograph of water stages in lower node of the river network (Trzebiez),
- hydrograph of flows or water stages and time series of concentration and transport of constituents in upper node of the river network (Widuchowa),
- wind field over the area of the Lower Odra.

Figure 1
In any internal node or at any cross-section of the river network can be prescribed – as auxiliary conditions – inflows/outflows and concentration of constituents, changeable in time.

MODRIM produces as results:
- water levels,
- flows,
- velocities,
- concentration and transport of nutrients and
- concentration of suspended matter at every cross-sections of the lower Odra river network. The results are stored in data files (ASCII type) and in the tailor-made binary exchange file, via that MODRIM passes them to the models of Odra Lagoon.

The system is foreseen to be supplied with measured and forecasted data stored in constructed for PIONEER purposes database - the MRI PIONEER DataBase. This database consists of Upper Layer with definition of measurement stations and five Lower Layers with data of following types:
- Point, as one value per time (Layer_1),
- Vector, as several values per time (Layer_2),
- Matrix, as several vectors per time (Layer_3),
- Moveable_Point, as \((x, y, z, val_x, val_y, val_z)\) taken in different time and places along the ship route (Layer_4),
- Moveable_Vector, as vertical profiles taken in different time and places along the ship route (Layer_5).

At present the following measuring data - coming from the period 1997-2000 - have been stored in the MRI PIONEER DataBase:
- results of hydro-meteorological observations carried out by Maritime Research Institute Szczecin (MRI) within the scope of the Odra Estuary monitoring network; this network consists of automatic stations with sensors and limnigraphic stations, located in 17 places of the Lower Odra and around Odra Lagoon (Dybkowska-Stefek, Pluta, 1996),
- long time series of hydro-meteorological data obtained from Polish governmental Institute of Meteorology and Water Management,
- water quality parameters obtained from Voivodeship Inspectorate of Environmental Protection in Szczecin, responsible for surface waters monitoring,
- time series of water quality parameters noted at automatic monitoring station Widuchowa,
- results of hydro-meteorological and water quality observations carried out by GKSS Research Center within western part of the Odra Lagoon (Rosenthal et al., 1997),
- results of hydro-meteorological and water quality measurements done during Polish-German campaigns on the Odra Lagoon.

Output software of the MRI PIONEER DataBase offers possibilities of:
- preparing of text and graphical reports for the parameters specified by end users,
- extracting the data as ASCII files in chosen format,
- producing necessary ASCII file with initial and boundary conditions for the operational models of the system in user-friendly way.

The database is located on the MRI server estua.im.man.szczecin.pl and available via WWW.

In the system results of measurements and simulations have been processed using geostatistical and dynamical data assimilation techniques (Bertino et al, 2000)

The developed system for the Odra Estuary will be applied to assess scenarios of possible in the future changes of nutrient load and its distribution in the Odra Lagoon. Results of performed simulations will be stored in the database and conveyed to possible end users.

CONCLUSIONS

The system for the forecast of nutrients in the Odra Estuary, proposed and developed in the frame of the PIONEER project, can be utilised in the future as operational, user-friendly tool supporting sustainable management of the coastal zones. The experience gained in the project can be used for the construction of similar systems for estuaries of other European rivers.
REFERENCES


RWSR: AGGREGATION, INTEGRATION AND PRESENTATION OF MONITORING DATA OF DUTCH REGIONAL WATER SYSTEMS, A TOOL FOR EVALUATION OF WATER POLICY AND MANAGEMENT

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The development, technique and practical use of RWSR, a new method to aggregate and integrate wholesale monitoring results of Dutch regional water systems, surface water as well as groundwater, is described. This method will be the basic instrument used to describe and analyse (regional) water system. It is a building stone to evaluate water management and policy in The Netherlands.

INTRODUCTION

By the publication of the Dutch Third National Water Policy Scheme (State Ministry of Transport, Public Works and Water Management, 1989) a new type of water management was introduced in the Netherlands: an integrated approach of water quality and water quantity of surface water as well as groundwater on basis of water catchment areas. This new "water system approach" required new monitoring programs and new tools for water policy and water management evaluation.

Therefore a new method for uniform assessment, aggregation, integration and presentation of monitoring results of Dutch regional water systems, surface water as well as groundwater, has been developed since 1994. This development was carried out by the twelve provinces of the Netherlands in co-operation with waterboards and the State Ministry of Transport, Public Works and Water Management. The main responsibility of the development of this method known as "Regional Water System Reporting" (RWSR), falls to the General Advising Group Water of the InterProvincial Office (IPO).

The main aims of the method developed in the joint project are:
- To obtain mutually comparable results of provincial monitoring reports to get a consistent survey of regional water systems at a national level;
- To obtain a tool that can be used not only by provinces, but also by waterboards as well as national ministries to draft annual evaluating reports about regional water policy and management in a (semi-)computerised way
- The level of reporting should be adjustable to the various purposes of water policy and management: The lowest level for operational water management by waterboards, the highest level for strategic evaluations on the national scale or on the scale of whole river-basins. Methods used for upscaling of information are geographical aggregation and thematical integration.

A joint development with respect to water policy and water management in the Netherlands

On basis of the Dutch Third National Water Policy Scheme (1989) and the Dutch Law on Water Management (1990) the provinces have to draft Water Policy Schemes¹ every four to eight years. In these schemes functions for use and conservation of water systems must be defined. This enables a better insight into the future water management based on quantified aims. This must be elaborated by the waterboards in Water Management Schemes², also with a working period of four to eight years. In the schemes of the provinces and the waterboards the views for long-term (20 to 25 years) as well as short term (four to eight years) aims must be worked out as quantitatively as possible.

¹Water Policy Schemes of the provinces include a merge of various themes:
1. new opinions about developments desired;
2. translation of national and European water policy to regional level;
3. working out the existing starting-points for water management in relation to the environmental policy, the regional planning aims and possibilities and special fields of policy such as agriculture, nature conservation and shipping.

²Water Management Schemes: More or less operational integral schemes on the basis of the provincial Water Policy Schemes for the whole of the management area of each waterboard.
For the elaboration of views on water management and to ascertain the degree to which aims of water policy and water management are obtained, a joint development of methods by provinces, state and waterboards has been chosen. This ensured the less expensive development of a uniform system to describe and evaluate monitoring data with mutually comparable results and enough general acceptance to use the method.

**Technique of the RWSR method**

In figure 1 the stepwise procedure of the RWSR method is presented roughly and schematically.

**Figure 1. Procedure used in the RWSR method**

The technique of the RWSR method developed conforms to the following uniform stepwise sequence:
1. boundaries of water systems are determined by a specially developed working method;
2. for each water system the functional aims are derived from the provincial Water Policy Schemes and the Water Management Schemes of the waterboards;
3. functional aims are presented by relevant standards for water quality as well as water quantity and morphology of water courses;
4. basic data of quality, quantity and management of ground and surface water are transformed to indicator values for the derived relevant aims;
5. indicator values are tested for the measuring scale of standards;
6. aggregation and/or integration of indicator values;
7. the results of the analyses can be described by a rather uniform reporting method (consequent use of five comparable score classes with fixed colours, standard lay-out for tables, standard GIS-application);
8. the results can be used to evaluate water policy and water management.
Ad. 1. The methods of determination of the boundaries of water systems have been described by CIW (1998), Rijkswaterstaat (1995), TNO-GG (1996) and IWACO (1996). Applying the CIW-method is recommended.

Ad. 2. In the provincial Water Policy Schemes and the Water Management Schemes of the waterboards functional aims are described for water systems and/or water courses ("policy" in figure 1). CIW (2001b) describes a standard set of functions for Dutch water systems (basic level, water for drinking or industrial use, agriculture, nature, shipping, municipal water, swimming, high water storage). These functions are related to guarantees for use or protection of water systems, expressed in standards as objective as possible. In the RWSR method over one hundred indicators have been drafted for the eight functions mentioned above and for some water themes (such as pollution sources).

Ad. 3. The relevant standards for functional aims (water quality, water quantity as well as morphology of water courses) are derived from European directives (European Union, 2000), National Water Policy Schemes (State Ministry of Transport, Public Works and Water Management, 1989, 1999), provincial policy schemes and management schemes of waterboards.

Ad. 4, 5, 6. The method of transformation of basic data of water quality, water quantity and management of ground and surface water into indicator values for the derived relevant aims (the "confrontation" procedure presented in figure 1) has been derived from the "Water Dialogue" method, developed earlier as an evaluation instrument for the water management of Dutch state managed waters (Stutterheim, 1996; Latour et al., 1998). Table 1 schematically shows how to obtain the final water index starting from the measurement data. In Annex 1 this method -which has been roughly adopted in the RWSR method-, is described more profoundly.

<table>
<thead>
<tr>
<th>Measurement: basic data</th>
<th>Test value (for example average, 90-percentile)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Test value</td>
<td>Distance to standard (below or exceeding)</td>
</tr>
<tr>
<td></td>
<td>Test ratio = below value / test value</td>
</tr>
<tr>
<td></td>
<td>Test ratio = test value / exceeding value</td>
</tr>
<tr>
<td>Test ratio (1 - 11)</td>
<td>Water index = 110 - (10 * Test ratio)</td>
</tr>
<tr>
<td>Individual water index</td>
<td>Aggregation / integration</td>
</tr>
<tr>
<td>Water index</td>
<td>Classification (five scores / colours)</td>
</tr>
</tbody>
</table>

Ad. 7. In the RWSR method the results of the analyses described above are represented by a uniform reporting method. This uniformity is mainly the result of a standardised relation between deviation of standards, index values, score classification and a fixed colour scheme for the five resulting score classes (see table 2). A consequent use of the five colours in graphs and GIS-maps enables an easy comparison between RWSR-reports about various regions and for very different levels of aggregation and integration.

In Annex 2 an example (assessment of chloride content) for the whole RWSR-procedure is given for one of four types of RWSR-indicators.

<table>
<thead>
<tr>
<th>Score</th>
<th>Index value (^3)</th>
<th>Deviation of standard</th>
<th>Colour</th>
</tr>
</thead>
<tbody>
<tr>
<td>5</td>
<td>100</td>
<td>No deviation</td>
<td>Blue</td>
</tr>
<tr>
<td>4</td>
<td>90 – 99,99</td>
<td>Slight deviation</td>
<td>Green</td>
</tr>
<tr>
<td>3</td>
<td>80 - 90</td>
<td>Deviation</td>
<td>Yellow</td>
</tr>
<tr>
<td>2</td>
<td>60 – 80</td>
<td>Strong deviation</td>
<td>Orange</td>
</tr>
<tr>
<td>1</td>
<td>&lt; 60</td>
<td>Very strong deviation</td>
<td>Red</td>
</tr>
<tr>
<td>0</td>
<td>0,00</td>
<td>Not applicable</td>
<td>Blank</td>
</tr>
</tbody>
</table>

In the following text the RWSR-technique is treated more profoundly according to the stepwise sequence mentioned above.
Use of the system

A set of representative indicators has been developed for all water management functions used in The Netherlands and for some policy themes. Most of these indicators more or less describe the quality of the water system (surface and groundwater); fewer indicators describe sources of pollution or hydrological effects.

From 1997 to 2000 the RWSR method has been tested in practice (1) in many small pilot projects to study specific problems of the method as well as (2) in integral projects describing large areas in which general practical problems were determined. The following pilot projects can be mentioned: Midden-Texel (Rot & Bergfeld, 1997), Bergambacht (Hoogheemraadschap Krimpenerwaard a.o., 1997), Afwateringsgebied Cadzand (Koster, 1997), three water system studies in the province of Fryslan (Witteveen + Bos, 1998), two rivulet system studies in North-Brabant (Witteveen+Bos, 1999) and the River Grift system (Waterboard Veluwe, 1999). The province of Gelderland carried out a RWSR pilot project about groundwater and desiccation and the Regge en Dinkel waterboard studied the use of the RWSR method for the description of the morphology of brooks (Ecoquest, 1997).

Already six of the twelve Dutch provinces (Flevoland (1999), Limburg (2001), North-Brabant (2001), Gelderland (1999), Zeeland (1997, 2000) and North-Holland (2000)) already carried out integral water system reports on basis of the RWSR method describing their whole policy areas (2,000 to 4,000 square kilometres). These reports acted as important building stones for evaluations of existing provincial Water Policy Schemes or for drafting new Water Policy Schemes. Meanwhile many water boards tested the method as one of the tools used for drafting or evaluating Water Management Schemes with positive results.

The experiences of all projects mentioned above have resulted in adaptations of the RWSR method (Miedema & Van Dijk, 2000). The revised manual 2000 of the RWSR method with over one hundred water management indicators, became available to all Dutch waterboards and provinces. The software modules that are necessary for an operational use (central calculating module with possibility to construct new indicators, Oracle database, GIS-applications for Arcview and Smallworld, modules to fit on water assessment instrument BEVER) had been tested and were sent to all water managers. It is expected that using the available tools described, general use of the uniform RWSR system to deliver water management data (required test values) to provinces and governmental organisations will start in 2002. Based on the test values, uniform reports on water data, management and policy at different levels can be drafted.

DISCUSSION

The main remarks on the use of the RWSR method in practice were as follows:

The main positive points observed were:

- Communication within the projects has led to more mutual comprehension of management problems.
- Reports divided into various aspects of water management and various areas can be replaced by one internally coherent and comparable reporting enabling a more integral view on water management.
- Measurement programs in relation to water management which sometimes appeared to be outdated, have been regauged with respect to optimization of the information which is necessary for evaluation of water policy and management.
- Introduction of the RWSR method leads workers to express new policy goals into more objective words as well as to better insight in the degree to which existing policy aims are realistic.
- The implementation of the RWSR method is seen as a growth process in which practical experiences gradually are integrated in the method.

3The water index values used in the RWSR method are different from the original values of the Water Dialogue method. The water index values of the Water Dialogue are: Score 1 (very strong deviation): water index < 53; Score 2: w.i. ≥ 53 and < 80; score 3: w.i. ≥ 80 and < 93; score 4: w.i. ≥ 93 and < 100; score 5 (no deviation): w.i. = 100. The reason to depart from the original water index values was that the regional water managers wanted to use limits based on simple, intelligible factors for deviation of standards (<2, 2-3, 3-5, >5).
Bottlenecks observed in practice were:

- The number of RWSR-indicators (> 100) is large. Not all these indicators can be included in the set of obligatory indicators.
- The indicators should comply with the policy targets for water management.
- Especially the data on surface water quantity were not yet available in an objective form and the accuracy fell below the required standard. This resulted in "expert judgements" about average classes for whole water systems. The size per class can not be calculated per class which reduces the degree to which results can be reproduced and mutually compared.
- Groundwater data are usually less detailed than surface water data.
- The existing measurements of water quality and quantity often appear not to be attuned to the existing water systems; in these cases the measurement programs must be adjusted.
- The RWSR-indicators developed only describe the health and development of water systems (water quality and quantity parameters, ecological assessment). General indicators on a higher policy level have not been developed yet.
- After integration at the highest level of water function for social use and policy themes, the results of RWSR are often levelled out too much to present useful information.

Finally the RWSR method has been checked in view of the demands of the Framework Directive Water of the European Union (2000) which is in its implementation stage. Although the RWSR method has been developed since 1994, the demands of this directive appear to fit well with the possibilities and characteristics of the RWSR method in the following respects:

- the catchment area is the starting point for analysis, policy aims, management schemes and measurement programs;
- integral reporting about water quality and quantity, surface and groundwater;
- possibility of uniform aggregation and integration of indicators;
- consequent classification according to 5 class graduation.

It is expected that the RWSR method will be a generally employed basic tool for the analysis of regional catchment areas, a useful tool for drafting reports about water management and an important building block for evaluating the water policy in the Netherlands. After completing the software modules automating the RWSR-calculations (test value, water index, integration and aggregation) and the presentation of RWSR-results via GIS-maps and graphs (in 2001), the use of the RWSR method has been strongly facilitated which enables a widespread implementation in all Dutch organisations working in the field of regional water management and policy.

N.B. Current developments of the RWSR method and its software applications (named iWSR) can be followed (mainly in Dutch language) by way of the homepage www.waternet.net/iporwsr. From this site the complete guidebook of the RWSR method (in Dutch language) can be downloaded as well.

ACKNOWLEDGEMENT

The authors are indebted to Mr. W.G.G. Hurkmans for a critical review of the manuscript.

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ANNEX 1

Calculation method of the WATER DIALOGUE method (Source: Stutterheim, 1996)

The calculation steps of the WATER DIALOGUE method –roughly presented in table 1- are worked out more profoundly below.

Step 0 Calculating the test values

For each relevant aim variable the TEST VALUE is calculated for each year and for each water system. The test value is calculated based on the measurement data in one water system during one year (one or more representative sampling plots). The way in which the test values are calculated differs for the various indicators. Calculation can be on basis of e.g. 90-percentiles, the median value or the average of a measurement series.

Step 1 Distance to standard

The distance between test value to standard value (= test ratio) is determined. This distance is expressed in the "test ratio". The test ratio is obtained by:

a. dividing the below value of the standard by the test value:
   - if only a below value occurs;
   - if as well below as exceeding values occur AND the test value is smaller than the below value;

b. dividing the test value by the exceeding value of the standard:
   - if only an exceeding value occurs;
   - if as well below as exceeding values occur AND the test value exceeds the exceeding value;

c. test ratio = 1 if test value between below and exceeding standars.

---

4Aim variable: characteristic or component of water system which is representative for parts of that water system or for the use of that water system. Aim variables are distinguished into aim variables which describe the status of the water system and aim variables for the size of (a part of) the water system.
Average for a period of several years
The years of the period involved are aggregated by averaging the test ratios. In this way (biological or weather) fluctuations such as dry and wet years and nutrient rich and poor years are averaged out.

Step 2   Limiting the test ratio
The test ratio ranges from 1 to 11. A test ratio smaller than 1, is set to 1; a test ratio exceeding 11, is set to 11.

Step 3   Individual water-index
On basis of the test ratio (>=1, <=11) the individual water-index-value for each aim variable is calculated. The water index varies between 0 and 100.

The formula is: INDIVIDUAL water-index-value = 110 - (10 x test ratio).

Step 4   Aggregation to water-index
The procedure of clustering5 aim variables and aggregation of the water systems consists of two steps.
1. Clustering of aim variables:
   the aim variables are clustered by category (e.g. chemistry or economy). The totals of the aim variables "SYSTEM" and "USE" are determined separately.
   When using 2 to 5 aim variables the simple average value is calculated. In case of 6 or more aim variables the 50-percentile (median) value is determined.
   The TOTAL of "SYSTEM" and "USE" is determined by its average.
2. The aggregation of water systems:
   within one category, within total "SYSTEM", total "USE" and TOTAL the water systems are aggregated.
   In case of 2 to 5 water systems the simple average value of the individual water-index-values of the categories involved per total column is calculated. In case of 6 or more water systems the 50-percentile (median) value is calculated based on individual water-index-values of the categories involved, or based on the total column.

Aim variables in various categories
The whole of SYSTEM- or USE- aim variables is determined by performing the aggregation step for all separate individual index values. Thereby the results of e.g. chemistry are not added to the results of ecology.

For adding up "total SYSTEM aim variables" and "total USE aim variables" to "TOTAL" the average of both is used. The reason is that ecology and economics must have equivalent impact in the final assessment, not dependent on the number of ecological and economical aim variables.

In case of regional water systems the aggregation step is carried out on basis of individual index values of the regional waters of a catchment area to obtain the water-index per watertype.

Step 5   Presentation
The water-index-value, of a separate aim variable as well of a cluster, is presented as a colour. This colour indicates the deviation from the standard (red: large deviation, blue: no deviation; see table 2).

5Clustering: joining (integrating) of information of individual aim variables within one indicator.
ANNEX 2

RWSR case assessment of chloride concentration (RWSR indicator basic ecology AEF-9) (one of four RWSR indicator types)

Parameter:
- Chloride
  - Surface or length
  - ppm
  - m² or m

Transformation: Determine 90 percentile value of concentration values

Test value of indicator
- 90 percentile value of chloride content

Determine grade mark: compare test value with basic standard; calculation individual water index. (WATER DIALOGUE-method)

Clustering: not applicable

Water index for substance

Classification: determining score (1,2,3,4 or 5) based on waterindex for chloride

RWSR indicator score

Aggregation: Adding up of average scores of sampling points for each water system to one score

Integration: e.g. average of the indicator scores related to one aspect (surf. water quality) to one aspect score

Presentation: Class scores for individual substances, group of substances or water system
In modern practice, water quality assessments have been made employing comparisons of the parameters measured with the water quality standards. The information obtained is complicated to be perceived and used in making decisions on water management and waste water discharge.

The present work is an attempt of comprehensive approach to water quality assessment involving major regime-forming factors and the results of chemical and biological analysis. The programme measurements at a water quality network cover oxygen, including saturation percentage, pH, BOD, COD, nutrients, salt composition, oil products, heavy metals, pesticides, zoo- and phytoplankton. Each observation station is informed of the quality of waste water that enters the water body and of the water flow. This allows to determine a complex water quality index (water pollution index, WPI) at different natural water \(Q_{\text{nat}}\) / waste water \(Q_{\text{wst}}\) ratios. The comprehensive approach to water quality assessment that employs all the standardized parameters allows grading of quality of water bodies and specific sites (representative, reference, input, flux) to be made.

INTRODUCTION

In modern practice, water quality assessments have been made employing comparisons of the parameters measured with the water quality standards established under legislative documents. The information obtained is complicated to be perceived and used in making decision on water management and waste water discharge.

The present work is an attempt of comprehensive approach to water quality assessment based on major regime-forming factors and chemical and biological analysis, with reference to major rivers of Latvia exposed to different pollution load.

Such an approach to the integrated assessment of water quality has been employed in the national monitoring practices and a general review of the tendencies in water quality at each site under observation.

METHODOLOGY

The programme measurements at a water quality network (Figure 1) cover oxygen, including saturation percentage, pH, BOD, COD, nutrients, salt composition, oil products, heavy metals, pesticides, zoo- and phytoplankton. Each observation station is informed of the quality of waste water that enters the water body and of the water flow. With this information, a complex water quality index (water pollution index, WPI) can be derived for different natural water \(Q_{\text{nat}}\) / waste water \(Q_{\text{wst}}\) ratios. The WPI is the sum of the ratios of the parameters measured to the standards in force:

\[
WPI = \sum_{i=1}^{n} \frac{C_i}{SFQS} \times \frac{1}{n}, \text{where}
\]

- \(C_i\) – annual mean concentration
- \(n\) – is the number of elements
- \(SFQS\) – is the surface freshwater quality standards

With the aim to unify the calculations, each site is recommended to determine WPI involving standardized substances, \(O_2\), \(O_2\) saturation, pH, suspended matter, BOD\(_{5(7)}\), P\(_{\text{tot}}\), NO\(_2\), NH\(_4\), Zn, Cu, Cd, Pb, benthos saprobic index and phytoplankton biomass (Table 1). The selection of these parameters is dictated by the surface fresh water quality standards effective in Latvia.
Figure 1. Water quality network in Latvia

Table 1 Surface freshwater quality standards

<table>
<thead>
<tr>
<th>N</th>
<th>Parameter</th>
<th>Measurement unit</th>
<th>Quality standard</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>O2</td>
<td>mg/l O2</td>
<td>8</td>
</tr>
<tr>
<td>2</td>
<td>Saturation (Dissolved oxygen)</td>
<td>% sat. O2</td>
<td>90</td>
</tr>
<tr>
<td>3</td>
<td>pH</td>
<td>mg/l</td>
<td>7</td>
</tr>
<tr>
<td>4</td>
<td>Suspended matter</td>
<td>mg/l</td>
<td>25</td>
</tr>
<tr>
<td>5</td>
<td>BOD (biological oxygen demand)</td>
<td>mg/l O2</td>
<td>2.0</td>
</tr>
<tr>
<td>6</td>
<td>Total phosphorus</td>
<td>mg/l P</td>
<td>0.100</td>
</tr>
<tr>
<td>7</td>
<td>Nitrite</td>
<td>mg N/l</td>
<td>0.012</td>
</tr>
<tr>
<td>8</td>
<td>Ammonium</td>
<td>mg N/l</td>
<td>0.04</td>
</tr>
<tr>
<td>9</td>
<td>Zinc</td>
<td>mg/l Z</td>
<td>0.3</td>
</tr>
<tr>
<td>10</td>
<td>Copper</td>
<td>mg/l Cu</td>
<td>0.04</td>
</tr>
<tr>
<td>11</td>
<td>Lead</td>
<td>mg/l Pb</td>
<td>0.05</td>
</tr>
<tr>
<td>12</td>
<td>Cadmium</td>
<td>mg/l Cd</td>
<td>0.005</td>
</tr>
<tr>
<td>13</td>
<td>Saprobic index, by benthos</td>
<td>-</td>
<td>1.7</td>
</tr>
<tr>
<td>14</td>
<td>Phytoplankton (biomass)</td>
<td>mg/l</td>
<td>1.57</td>
</tr>
</tbody>
</table>

The WPIs are employed in grading quality of water bodies (sections) (Table 2)

Table 2 Water quality classification

<table>
<thead>
<tr>
<th>Class</th>
<th>Characteristics</th>
<th>WPI</th>
</tr>
</thead>
<tbody>
<tr>
<td>I</td>
<td>Very pure</td>
<td>≤ 0.3</td>
</tr>
<tr>
<td>II</td>
<td>Pure</td>
<td>0.3-1.0</td>
</tr>
<tr>
<td>III</td>
<td>Moderately polluted</td>
<td>1.0-2.0</td>
</tr>
<tr>
<td>IV</td>
<td>Polluted</td>
<td>2.0-4.0</td>
</tr>
<tr>
<td>V</td>
<td>Impure</td>
<td>4.0-6.0</td>
</tr>
<tr>
<td>VI</td>
<td>Heavily impure</td>
<td>&gt; 6</td>
</tr>
</tbody>
</table>
Results

1. According to the calculated WPI indices for 1998, the water bodies of Latvia were mostly classed as pure and moderately polluted (Figure 2). Judging from Figure 2, maximum WPI values were obtained for the Abuls (downstream from the town of Smiltene) in the Gauja basin and the Tebra (downstream from the town of Aizpute) in the Baltic Sea small rivers basin. This is caused by low waste water dilution by natural water.

![Figure 2. Quality of water bodies, Latvia, 1998](image)

2. WPI is dependent of the extent of waste water dilution by natural water (Figure 3).

![Figure 3. Waste water dilution – WPI relationship, Lielupe River (Jelgava)](image)

The extent of the relationship depends on the intraannual dynamics of natural water flow.

3. Surface waters of Latvia in 1999 were classified as Class II (conventionally clean), Class III (moderately polluted) and Class IV (polluted). The Abuls, Gauja basin, falls under the latter (Figure 4).

4. Compared to the previous year, the percentage of Class III and Class IV water bodies has increased (Figure 5).
CONCLUSION

1. The comprehensive approach to water quality assessment that employs all the standardized parameters allows grading of quality of water bodies and specific sites (representative, reference, input, flux) to be made.
2. The WPI provides for a good comparison of water quality in river tributaries and basins.
3. With higher extent of waste water dilution in water bodies, the magnitude of WPI reduces.

References

LAKE OHRID CONSERVATION PROJECT. MONITORING PROGRAMME

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The Lake Ohrid, the oldest one in Europe and one of the oldest in the world, represents a refuge for numerous freshwater organisms from the tertiary period whose close relatives can be found only as fossils. It is inhabited with numerous endemic and relict forms or organisms. Lake Ohrid (max. length 30.4 km; max. width 14.8 km), is situated in the SW part of FYR of Macedonia in the Ohrid valley, on the border with Republic of Albania at 639.2 m above sea level. Around 2/3 of the lake surface area belongs to FYR of Macedonia and 1/3 belongs to Albania. It has a surface area 358.2 km², maximum depth of 288.7 m, mean depth 163.7 m, shoreline length 87.53 km (56.02 km Macedonian and 31.51 km Albanian). It has a drainage area of 1042.25 km² and 40 tributaries most of which are temporary flowing creeks and rivers—especially during heavy rains and melting of snow from the surrounding high mountains.

INTRODUCTION

A Lake Ohrid Conservation Project supported by the World Bank, has been established with joint efforts from both countries sharing the lake FYR Macedonia and Albania. It consists of four components:


This project is aimed at improvement in restoration and future conservation of the lake.

IMPLEMENTATION OF THE LAKE OHRID MONITORING PROGRAM

The design of the monitoring program is based on present knowledge of Lake Ohrid’s characteristics and of the main pollution hazards to its ecosystem. Each year, the Monitoring Task Force (MTF) will review the effectiveness of the programme, taking into account increasing knowledge of the system and possibly new threats of contamination. It will make necessary changes in the program and work plan, and report its findings and actions to the public and the Lake Ohrid Management Board.

ADOPTION OF JOINT PROTOCOL BY THE MTF ON SAMPLING AND ANALYSIS METHODS

The MTF will develop and adopt technical methodologies and procedures to be used by Albanian and Macedonian institutions and specialists participating in the Lake Ohrid Programme (LOP). The technical protocol will encompass the following key areas:

(i) sampling and monitoring stations,
(ii) sampling methodology and sample handling,
(iii) laboratory procedures and interpretation,
(iv) data format and exchange,
(v) lake and watershed modelling
(vi) quality assurance.

To ensure a joint evaluation and interpretation of monitoring data and to avoid sources of errors, the same sampling and analysis methods and procedures will be used by the institutions involved. Parallel testing for each parameter at the moment of its incorporation into the monitoring program, should ensure comparable analytical procedures.

EVALUATION AND INTERPRETATION OF EXISTING DATA

Evaluation and interpretation of existing data is part of the monitoring program. This study would provide a jointly agreed reference point for the LOP. By comparing the results of new investigations with earlier results, such as species lists, threatened and endangered (endemic)
species can be recognized, and anthropogenic activities may be changed accordingly to avoid further negative impacts on the environment. A joint team of scientists, appointed by the MTF, will compile and screen all available data from widespread sources. On the basis of the accepted data, the joint scientific group will prepare a scientific report documenting the development and present status of the specific character of Lake Ohrid's ecosystem. Of primary interest are the phyto-and zooplankton, benthic fauna, fish, reed-belts and Chara-meadows in the littoral and sublittoral zone, respectively. Data in the pelagic zone (e.g. physical and chemical parameters such as temperature, phosphorus etc.) and existing load data of main tributaries should also be taken into account.

DEFINITION OF SAMPLING STATIONS

The monitoring program consists of permanent as well as of sporadic measurements and investigations. Five main types of sampling stations are planned (Figure 1):

• 10 permanent sampling stations in the pelagic zone (stations 1 to 8 on Lake Ohrid, stations 9 and 10 on Lake Prespa),
• 13 stations to identify and control local pollution and contamination (locations a-m),
• stations to monitor the main tributaries (Sateska, Koselska, Velgoska, Cerava, Cerni Drin and Golema (Lake Prespa), the effluents of wastewater treatment plants (Struga and Resen) and non-point or diffuse sources,
• transects of sediment cores in Lake Ohrid, sediment cores in Lake Prespa,
• investigation of littoral zone (macrophytes, fish spawning grounds etc.),
• frequency of monitoring and relevant parameters.

PARAMETERS AND SAMPLING FREQUENCY

The sampling frequency for the parameters to be monitored will vary from monthly to much longer annual intervals depending on the variability of each parameter and the cost to monitor it.

The LOP will initially include chemical parameters (such as phosphorus, nitrogen, organic carbon, oxygen, chlorophyll, heavy metals, pesticides), physical parameters (such as temperature, conductivity, and Secchi depth), and biological parameters (such as phytoplankton, primary production, zooplankton, fish status and statistics, macrophyte species, benthic community species, and bacterial contamination).

MONITORING RESPONSIBILITIES

Tasks within the monitoring program would be divided between the Albanian and Macedonian monitoring institutions as follows:

(i) On the Albanian side of Lake Ohrid, Albanian institutions monitor:
• locally contaminated sites,
• tributaries and diffuse sources,
• the littoral zone and
• the pelagic sampling stations 6 to 8,9 (Lake Prespa)

(ii) On Macedonian side of Lake Ohrid, Macedonian institutions observe:
• locally contaminated sites,
• tributaries and diffuse sources,
• the littoral zone and
• the pelagic sampling stations 1 to 5, and in addition, the pelagic sampling stations 10 on Lake Prespa.

For the monitoring of the sediment transects in Lake Ohrid-as a typical transboundary issue-the HBIO (Hydrobiological Institute of Ohrid) plays a leading role and shall be provided with appropriate equipment. Albanian experts may however participate in core sampling activities along the Lake Ohrid transects.
LAKE AND WATERSHED MODELLING

There is very little experience in either Albania or Macedonia with mathematical and simulation models of lake and watershed hydrologic and ecological systems. This capability will however be an essential tool for the long-term management of the Lake Ohrid system. Hence, a long term programme to develop this capability and to test existing approaches and models presently used in Europe and North America will be carried out by the MTF and its participating institutions. The Project Managers and the core teams will jointly prepare a preliminary terms of reference for this program which can be presented to selected institutions for expressions of interest and preliminary proposals, and to donors for support. Particular attention will be given to institutional arrangements, including staffing, costs and sustainability necessary to implement and sustain this activity.

MANAGEMENT OF MONITORING DATA

Each activity in the Albanian and Macedonian work plan will be carried out by single or teams of scientists in one more of the cooperating and participating institutions. The data resulting from these activities, as well as existing available data, will be compiled and organized by the respective lead institution in an information system accessible to all participating in the work. The MTF will jointly evaluate and adapt new computer facilities for such data management (hardware and software), and decide on the location and detailed organization of the data management system (including data recording and analysis procedures, data structure, compatibility, and data access and exchange mechanisms). The MTF may procure specialized technical assistance for this task.

EVALUATION AND PUBLICATION OF RESULTS

The MTF will report relevant monitoring results as well as recommendations for complementary measures to the Lake Ohrid Management Board on a quarterly basis. The relevant results of the LOP will be published once or twice per year in a joint Lake Ohrid Monitoring Bulletin. The MTF will consider establishing a mechanism for publishing occasional scientific and technical notes and papers, for which it may wish to establish an Editorial Board, possibly with support and participation of international scientists. Monitoring results of broad public interest will be periodically published in newsletters to the local population and to other interested parties.
INTEGRATION OF GIS AND RELATIONAL DATABASE FOR THE TRANS-NATIONAL MONITORING NETWORK IN THE DANUBE RIVER BASIN

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The Environmental Programme for the Danube River Basin was established to reduce the pollution load in the region. Water quality is managed by the Monitoring, Laboratory and Information Management expert group. A Data Exchange File Format (DEFF) relational database was used for the storage, processing and exchange of the Trans-National Monitoring Network (TNMN) data. The Croatian Reference Laboratory became the Central Point for the collection and merging of the TNMN Data for the entire Danube River Basin. Using Autodesk MapGuide products, we integrated the digital vector-based map of the Danube River basin with the DEFF relational database. A dynamic web application was created using the Autodesk MapGuide™ Author and Active Server Pages (ASP) tools. The application is created in such a way that the amount of information visible is controlled by the scale of the digital map in the viewer. Different reports as well as dynamic, thematic maps may be created automatically on demand by the user for the selected year and determinand (officially accepted term for parameter determined). The aim of this paper is to describe the state of this research and the possibility of using Autodesk products for the management of water related environmental data.

INTRODUCTION

Environmental protection is a multidisciplinary area of human activity that has been attracting an increasing interest throughout the world and, lately, also in developing countries (such as Croatia). To enable sustainable development it is necessary to avoid lasting environmental damage and preserve the long-term overall quality of life while maintaining continued access to natural resources. In the European Union (previous EC/DG XIII: Telecommunications, Information Market and Exploitation Research), significant support has been given to research and technical developments in the domain of Environmental telematics (infrastructure consisting of communication networks, electronic equipment, databases and services built on these components) applications launched in the context of the 4th and the ongoing 5th Research, Technological Development and Demonstration Framework Programmes. The Telematics Application Programme (TAP, 1994-1998) and Information Society Technologies Programme (ISTP, 1998-2002) include environment activities to provide access to, and share, better quality environmental information, thereby enabling informed decisions to be taken that improve the quality of the environment, as well to achieve both the environmental protection and the economic growth (EC/DG XIII, ENTAG, 1997). The development of integrated systems, Geographic Information Systems (GIS), distributed and meta databases, decision support systems and use of modern technologies such as the World Wide Web (WWW or Web)/Internet, Common Object Request Broker Architecture (CORBA) and Java were recommended. The need to understand the temporal and spatial distributions of information is also an important consideration.

Environmental Programme for the Danube River Basin

The Environmental Programme for the Danube River Basin (EPDRB) represents the synthesis of the main goals in TAP and ISTP as applied to the Danube Basin (UNDP/GEF, 1998; UNDP/GEF and ISEP, 1999). One of the main aims of the EPDRB was to perform water quality and quantity assessments on the established Trans-National Monitoring Network (TNMN) which includes monitoring stations from 12 Danubian countries. Upon the decision made by the Danubian countries in co-operation with the EU, international organizations and donor countries, the EPDRB was established in 1991 during the Ministerial Conference in Sofia. According to the accepted Program Work Plan (PWP), three Expert Groups (EG) were established:

• Accident Emergency and Warning alert System (AEWS),
• Data Management (DM) and
• Monitoring, Laboratory and Information Management (MLIM) performed at border stations located on the Danube River and its main tributaries, at important water intakes as well as at significant pollution sources.
In 1994, the Danube River Protection Convention (DRPC) was signed in Sofia, which was similar to the conventions for the Rhine, Elbe and Odra rivers. During that year, the suggestions of WTV consortium experts (including WRc from the United Kingdom, TNO from the Netherlands and VKI/DHI from Denmark) to improve the current situation in water related environmental management for Danubian countries were accepted, and the plan to carry out Phase I of the EPDRB was examined in detail. During Phase I, the most important role was given to the activities of the MLIM EG (Lack, 1997). Within the MLIM EG, Expert Sub Groups (ESG, earlier called Working Groups) were organized to perform the Monitoring (MESG), Laboratory Management (LMESG) and Information Management (IMESG). The WTV consortium emphasized the need to harmonize the information collected within the MLIM EG and to this purpose, the first version of the Data Exchange File Format (DEFF) database and application were developed using the PARADOX Relational DataBase Management System (RDBMS) and its development tools (Kuipers, 1996). Various data types were collected from TNMN stations and their exchange for Phase I had been initiated in 1996 (see Table I). Data were collected mainly from monitoring stations of type b and data collection from monitoring stations of type c has started recently.

Table I.: The basic structure of the TNMN data

<table>
<thead>
<tr>
<th>STATIONS TYPE a</th>
<th>Meteorological Data</th>
<th>STATIONS TYPE b</th>
<th>Water Quality and Quantity Data</th>
<th>STATIONS TYPE c</th>
<th>Sediment Quality Data</th>
<th>STATIONS TYPE d</th>
<th>Groundwater Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>General Physical &amp; Chemical Indicators</td>
<td>Water medium determinands</td>
<td>All sediment determinands</td>
<td>Planned for future</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>- frequency 6 days (2nd Phase hourly)</td>
<td>Station types M, P and W</td>
<td>- frequency 2/year Biological and microbiological indicators - Station type M and W</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Radioactivity</td>
<td>Biological indicators</td>
<td>- frequency 26/year Automated stations</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>- frequency daily (2nd Phase hourly)</td>
<td>1-12/year Station types B and L</td>
<td>- frequency hourly, mean values 12 or 26/year</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

B – border stations, C – stations on important tributaries, P - stations near important pollution sources, W - stations at important water intakes, L – stations for the pollution load determination and M – stations for the mass balance determination

Participation of Republic of Croatia in the EPDRB

The Republic of Croatia has participated in the activities aimed at the establishment of the EPDRB from the very outset. Responsibilities for the activities within AEWS EG were transferred to the State Water Directorate while the responsibilities for the DM EG were transferred to the State Directorate for the Environmental Protection. In the activities within MLIM EG, the State Water Directorate participated in co-operation with the institution Croatian Waters and the National Reference Laboratory, Center for Marine & Environmental Research of the Rudjer Bošković Institute in Zagreb. Since 1996, the Republic of Croatia has played the role of the Central Point (CP) for collecting and merging TNMN data from the whole Danube Basin into the DEFF database. All these activities, as well as DEFF database management are performed by the National Reference Laboratory (Pećar-Ilić and Ružić, 1998; Ružić et al., 1998). At the end of 1996, the first static Web Pages on the Internet dealing with TNMN Data in Croatia were created by the Reference Laboratory in co-operation with the Croatian Waters (Ružić et al., 1998). At the end of 1997, these static Web Pages were presented at a joined meeting of all three ESGs within MLIM EG, held in Tihany at Lake Balaton (Hungary). Since no direct Internet connections were available, a cellular telephone was used for remote Internet access to enable the presentation. At that time, the first investigations were started to make possible temporal and spatial presentations of the relevant TNMN Data for Croatia using the Autodesk MapGuide™ integration products from Autodesk (Ružić and Pećar-Ilić, 1997). In 1999, the actual development of the Integral Information System, the so-called DANube Basin Information System (DANUBIS) was initiated in order to manage the main activities of the International Commission for the Protection of the Danube River (ICPDR), which entered into force in 1998.
(UNDP/GEF and ISEP, 1999). This project was sponsored by the United Nations Development Programme/Global Environment Facilities (UNDP/GEF) and coordinated by the International Society for Environmental Protection (ISEP) of Vienna. We participated in the creation of the conceptual model for DANUBIS (UNDP/GEF, ISEP, 1999) and later in the implementation phase of a subsystem based on the DEFF system, concerning the monitoring of water data in the Danube Basin (Kuipers, 1996; Ružič and Pećar-Ilič, 1996). DANUBIS incorporated the organizational aspects of water quality and quantity monitoring (emission) and pollution sources monitoring (emission) in the Danube River Basin. DANUBIS was implemented in Oracle RDBMS, and made available to authorized participants from the Internet as a trial version (UNDP/GEF and ISEP, 1999). TNMN Data management will be performed using the Oracle version of DEFF system (Kuipers, 1996) so that the quality evaluation could be performed on the data stored within DANUBIS.

From these experiences we recognized the fact that spatial distribution of monitoring stations can play a significant role in the process of efficient data management. The organization of such activities needs special solutions on local, national and international levels. For example, databases could be used to manage all the necessary information about the water quality and quantity monitoring. GIS could be used for an efficient processing of spatial data. Nevertheless, for efficient water quality and quantity monitoring in the Danube River Basin, as well as for online decision making in various management levels, both the temporal and spatial data presentations should be provided.

The main goal of this paper is to describe the methodology developed for a specialized application system which would provide the integration of the digital map of the Danube River Basin with the existing database in order to improve the water quality and quantity monitoring within ICPDR activities. Development of such an application system was performed according to the EU’s TAP and ISTP recommendations.

**METHODS**

The modern approaches and information technologies are necessary to enable GIS-DBMS and Web-DBMS integration. The Web as a platform for database systems can deliver innovative solutions for complex management issues. The three-tier architecture maps to the Web environment with a Web browser acting as the ‘thin’ client (responsible only for the application’s user interface and performing some simple logic such as input validation) and a Web server acting as the application server (the middle tier business logic and data processing runs on a server). A DBMS stores data required by middle tier and may be run on a separate server called the database server. Database design includes conceptual database design (independent of implementation details), logical database design (targeted at specified data model) and finally physical database design (physically realized logical data model). The Entity-Relationship (ER), a high-level conceptual data model facilitates database design, conceals its technical aspects and supports user’s perception of the data. This model is independent of the particular DBMS and hardware platform that is used to implement the database. GIS database technologies include three principal types of database architecture: Computer Aided Design (CAD)-based, georelational, and object-oriented. The georelational data model stores the geometry of map elements together with their attributes. The attributes are stored as tables in a RDBMS and associated with the spatial data. There are several approaches to integrating databases into the Web environment (Connoly T. et al., 1999): a simple Common Gateway Interface (CGI), extensions to the Web server, scripting languages, Java, Active Server Pages (ASP), Microsoft’s Active Platform and Oracle’s Network Computing Architecture (NCE).

In using such approaches the following actions were performed: analysis of available information sources (so that the main goals and user requirements could be defined), application of various program products, development tools and languages (to enable the system design to be made), and system verification with the data collected within the EPDRB Programme. In the development process, the following products, tools and languages were used for different purposes:

- GIS - AutoCAD MAp® 2000 (Autodesk Inc., 1999) for the creation and management of digital backgrounds,
- RDBMS - Paradox® for Windows® NT for database and additional knowledge implementation and management (Simpson and Robinson, 1996),
- REPORTING SYSTEM (database – selected objects on the digital map) – ASP (Walther et al., 1999) for Microsoft® Internet Information Server (IIS),
- INTEGRATION SYSTEM – Autodesk MapGuide™ Server (Autodesk Inc., 1998) - dynamic access to vector-based and raster-based map data as well as to ODBC data sources, over an intranet, or the Internet, Autodesk MapGuide™ Author (for creating special digital maps, the objects of which are linked with reports) and Autodesk MapGuide™ Viewer (for presentation of such digital maps, with three possible versions Microsoft ActiveX Control, Netscape Plug-In and Java client),
- WEB SERVER – Microsoft® IIS 4.0 and
- WEB BROWSER – Microsoft® Internet Explorer 5

This application system is based on the integration of three different program systems which enables its main characteristic, automatic generation of reports a thematic maps. The first system is the Paradox RDBMS (which may later be replaced by Oracle), used to manage the existing relational database DEFF with relevant data about the water quality in the Danube River Basin. The second system is GIS AutoCAD Map, used for the creation and management of spatial data such as digital maps of the Danube River Basin. The third component is the Autodesk MapGuide integration system using which performs the automatic generation of reports and thematic maps. The DEFF system (Kuipers, 1996) may be applicable in the development process, however, only after the reconstruction of its conceptual model. To this purpose, the existing documentation (Kuipers, 1996), DEFF database structure files, together with application forms were analyzed by the reverse engineering Information System (IS) development method. This method was applied to the development of IS based on a unique database; either a completely new IS was developed over the existing IS or the existing IS had to be expanded. By applying this method at the mentioned sources it is possible to reconstruct the conceptual model of the DEFF system by representing its data structure using the main concepts of the E-R diagram. Based on the described entity dependencies, it is possible to define the mechanisms for additionally building-in the knowledge necessary for automatic generation of reports and thematic maps. To these purposes, the capabilities of Paradox RDBMS, as well as the capabilities of Structured Query Language (SQL), have been used (Simpson and Robinson, 1996). For creating digital Danube River Basin maps, the Autodesk Map GIS was used while for the integration with relational database, the capabilities of Autodesk MapGuide products were used. Business rules and application logic are implemented in a Web server using various tools, services and technologies in the Microsoft Active Platform such as Hypertext Markup Language (HTML), ASP, VBScript, ActiveX and Active Data Objects (ADO). Different reports in HTML format have been created using ASP. The preprocessing of specified user queries is necessary in order to enable automatic generation of thematic maps.

RESULTS AND DISCUSSION

The main concerns of this paper are the presentation and discussion of the most significant results as well as those which are crucial for achieving the main goals. In the development process of this application system, the reconstruction of the conceptual data model and its extension with additional knowledge have very a important role. Special attention is paid to the role of individual development tools in the process of GIS and relational database integration to provide specified requirements.

DEFF Data Model and Built-in Additional Knowledge

DEFF system (Kuipers, 1996), realized in Paradox RDBMS, includes database tables (database files, .DB) and application forms (main menu and frames, .FDL) for data insertion, for data reviews and reports and for additional connection with the AARDVARK statistical package. The DEFF database represents a set of mutually dependent tables (database files), which could be classified into static and dynamic types. Static database tables do not change very often in time and contain common information needed for all countries. Dynamic database tables contain various information about sampling as well as about analyses of determinands, which are different for particular countries.

Static database tables (files) are:
- COUNTRY.DB – table of countries participating in the programme,
- MONPOINT.DB – table of monitoring points located in individual countries within the Danube River Basin, from which samples have been taken,
DETERMIN.DB – table of determinands agreed by the programme,
SAMPMEHT.DB – table of sampling methods,
ANAMETH.DB – table of analytical methods for different determinands and
REMARKS.DB – table of standard remarks concerning relevant analyses.

Dynamic database tables (files) are:
SAMPLE.DB – table containing relevant information about different samples,
ANALYSIS.DB – table containing relevant information about the analyses of determinands performed with those samples and
LOCDETER.DB – table containing determinands registered for particular monitoring point.

Figure 1. Normalized E-R diagram for DEFF System
The reconstructed conceptual model of the DEFF system with E-R diagram is presented in Figure 1 (Ruzić and Pećar-Ilić, 1996; Pećar-Ilić and Ružić, 1998). E-R diagram contains entities of various types (such as elementary, weak and composite types) between which various dependencies exist (such as existentional, identificational or their combination). Elementary, or the so-called strong entity types are uniquely identified with their own identifiers and they do not depend on other entity types. In contrast, the weak entity types depend on other entities, most often on strong entities. Composite entity types are introduced as replacements for multiple binary connections enabling optimal data structure after the performed normalization of E-R diagram, as well as easier later implementation of complex relationships into the database. For example, ANALYSIS is a composite entity type, which has Existentional and Identiﬁcational Dependencies (ED&ID) with SAMPLE and ANAMETH which are weak entity types. This combination of dependencies is expressed in the fact that an entity cannot exist alone and cannot be uniquely identiﬁed by itself in a data model. In the given example, the mentioned dependencies mean that the information about the analysis could be given if the information about the corresponding sample exists while the analysis is uniquely identiﬁed with its own identifier as well as with the identifier for the corresponding sample (composite identifier).

Based on the described dependencies between individual entities (tables), it is possible to perform extensions of the data model with additional knowledge and afterwards deﬁne the necessary mechanisms for their building into the database using the capabilities of SQL. For example, the individual determinand membership in the appropriate parameter group, as well as the average determinand value membership in the proper category could be introduced, while the number of categories depending on the particular determinand have been recognized and then built into the database.

Development of the Integrated Application System

The Integrated Application System was developed during 1999 in the Croatian National Reference Laboratory (Ruzić et al., 1999; Ružić and Pećar-Ilić, 1999). The following important requirements have been achieved with this application system:

- Temporal and spatial data presentation using the combination of digital maps of the Danube River Basin, containing relevant information and relevant data reports in HTML format from the database,
- The possibility of specifying user queries, which increases the application dynamics,
- The possibility of generating thematic maps with the corresponding automatic generation of legends, which represents a great improvement in the ﬁeld of thematic mapping, since there are no limits to the number of determinands, dates of analyses and the number of monitoring points,
- Application could be accessed through the intranet by authorized participants on the programme, which provides eﬃcient information management,
- Basic resources (database, digital maps, etc.) could be used remotely, enabling the creation of a distributed management systems and
- Different RDBMS and GIS systems could be integrated within the same application.

The application is started when the appropriate Uniform Resource Locator (URL) is selected and the authorization has been successfully performed. In this case, the digital map of the Danube River Basin containing the monitoring stations included in TNMN, together with the relevant legend presenting active layers, is shown by Autodesk MapGuide Viewer. This special digital map is created using Autodesk MapGuide Author. The map contains other important layers such as the Danube Basin outline, borders of the countries participating in the programme and other European countries, rivers inside the Danube River Basin, major lakes and marine coastline. These layers are mainly based on vector ﬁles created using AutoCAD Map. The layer showing monitoring stations is based on geographic coordinates passed through the deﬁned ODBC data source for DEFF database. The viewer enables a set of standard operations that can be used for operations such as copying the map, zooming in/out, zoom go to, panning, return to the initial map scale, calling reports, and stop action. The amount of available information is controlled by the scale of the map and its layers activation. This means that performing a zoom in an operation under the speciﬁed scale activates other previously invisible layers (e.g. main roads, cities, as well as smaller rivers, cities and lakes). It is also possible to get information about various objects from the digital map, since its integration with database has been performed through relevant reports created using the capabilities of the ASP scripting. The ASP scripting enables the digital map integration with database by using the ODBC data source and
Active Data Object (ADO) concept (Walther et al., 1999). This means that in such scripts the transferred parameters (for example the codes of selected objects on the map) are used by the corresponding SQL queries, so that the results of their execution could be different reports in HTML format, including tables based on DEFF database and also graphs based on appropriate Java Applets. In such a way, scripts for different report types have been created. These reports are automatically generated by the corresponding ASP scripts after the selection of objects on the map, and/or after the additional user queries have been performed through the corresponding data input forms. For example, the monitoring stations from particular locations, countries, rivers or any of these combinations could be selected by the user and then the common information as well as water quantity and quality data made available. For the selected monitoring point, the basic statistical data processing, as well as the creation of time series diagram, could be automatically performed for the selected time interval and determinand that exist in the database for this monitoring point. (Ružič et al., 1999).

The most complex requirement built into this application are dynamic, thematic maps, which can be created automatically on demand by the user for the selected year and determinand. For example, the distribution of the BOD5 determinand for the 1996 is presented for the entire Danube River Basin (see Figure 2).

This may be performed in two different steps (Ružič and Pećar-Ilić, 1999). In the first step, the preprocessing, which includes selection of monitoring points (one, more or all of them from the map) and selection of the determinand and year through data the input form, should be performed. When these selections by the user, represented as specified requests to the database are submitted, the categorization of determinand average values may be performed based on the built-in knowledge. After this, the second step may be initiated, based on the information resulting from the first step. It includes automatic generation of the thematic map, where monitoring points are presented with coloured marks depending on the categories of average determinand values.

The colours used are linked with the corresponding list of average determinand categories. These mutual dependencies are presented in the legend in the following way: blue colour for...
category I (the best), green for category II, yellow for category III, red for the category IV, black for category V (the worst). The white colour is reserved for the cases when data are not available (not measured at the selected monitoring point and for the selected year). The corresponding range of values for individual categories may be automatically generated when the Legend for Categories report is started. For the selected monitoring points on the map (one, more or all of them) it is also possible to generate automatically another report, which includes complete information about these monitoring points together with the corresponding average BOD5 values for the selected year.

Each change in the DEFF database is automatically reflected on the presentations in this dynamic Web application without any other additional intervention, as can be seen from the previous description and examples.

CONCLUSIONS

In the process of TNMN data management, efficient presentation of relevant information, as well as efficient presentation of the water quality index distributed within the Danube River Basin are provided by integrating GIS and the DEFF database. This Application System enables dynamic temporal and spatial data presentations which are directed by specified user conditions and automatically create thematic maps based on the additional knowledge built into the database.

The possibility of automatic generation of thematic maps with the corresponding dynamic legend for the selected monitoring points, specified time period and determinand represents an improvement of the common standard procedure for thematic maps. This standard procedure includes creation of thematic maps by year for only a limited small number of most significant determinands for the entire Danube River Basin.

The application described here could be available through the intranet to all authorized participants within ICPDR so that the efficient management of relevant information and the online decision making for different management levels are provided.

The great advantage of this Integrated Application System is manifested in the possibility of using the Autodesk MapGuide Server with different types of GIS and RDBMS, which could be individually located on various sites connected with the Internet/intranet network. However, in that case, efficient and fast communication between these locations has to be provided.

For GIS-DBMS and Web-DBMS integration a number of very powerful software products exist in the market (such as ArcInfo and ArcIMS from ESRI, GeoMedia from Intergraph and MapGuide from AutoDesk). They differ from the point of view of their robustness, ease of use, technology, flexibility and price. Each of them could be used to produce similar applications provided that all of their components are used to enable innovative and efficient solutions.

REFERENCES


DRAINAGE WATER QUALITY INFORMATION MANAGEMENT IN EGYPT

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Drainage water quality monitoring is an effort to obtain information about the physical, chemical and biochemical characteristics of drainage water via statistical sampling. The usefulness of the information is highly dependent upon a monitoring program, or system, being properly designed and operated. The most common water quality monitoring systems are the continuous monitoring station and the routine monitoring. In the Nile Delta, around 140 locations were monitored on a monthly basis (Routine Monitoring System) through the water quality-monitoring program developed and operated by Drainage Research Institute (DRI) in co-operation with Delft Hydraulics and Taua Milliu firms. The routine monitoring is considered to be an integral part of regulatory water quality monitoring. It is the simplest and least expensive means of checking compliance with a standard and provides an overall picture of the drainage water quality in the Delta and Fayoum region. Routine monitoring can be made more efficient, reliable, and economic by improving network design, data collection, and data utilisation. The DRI Management Information System designed a drainage water quality database in which data can be stored, processed and retrieved over a network of minicomputers at DRI.

Water quality is variable by nature, however it appears that some data originally stored in the database show extreme values that cannot be explained easily. Therefore it is necessary to check the validity of these values. The aim of this research is to present a data handling system in DRI in order to provide accurate and reliable drainage water quality data. The data handling procedure is concluded from four stages. The quick scan stage is to control as much as possible the human errors such as sets calibration, handling samples, handling data forms, typing errors, etc. The logical check stage is to check and analyse more extensively to find or explain any outlying results. This stage mainly contains three items: checking individual variables; comparison of field and laboratory measurements and checking correlation between several variables. After all checks have been performed, it is possible that still data are entered in the database that are highly unlikely therefore, the data elimination stage is to eliminate the unlikely data according to data elimination criteria approved within DRI. The documentation stage is to keep track of changes made in the database after each sampling campaign.

INTRODUCTION

The Egyptian water budget is confined to the country’s share of Nile water, which is fixed at 55.5 * 10^9 m³/year, in addition to minor quantities of rainfall in the coastal regions. As the population in Egypt increases and as the demands of society become more complex and divers, the requirements placed upon available water supply continue to rise. Reuse of drainage water appears to be one of the most promising, practical and economical means of increasing the Egyptian water budget. The drainage system in the Delta is rather extensive and serves 4.7 million feddans of a total of 7.4 million feddans of agriculture land in Egypt. The drainage system in the Nile Delta is composed of 22 drain catchments where it either discharges to the northern Lakes or is pumped by 21 reuse pump stations into irrigation canals, mixed with fresh Nile water and reused for irrigation. The total length of the main drains in the Nile Delta is about 1600 km serving 4.7 million feddans. Assessment of water resources requires full understanding for the water quantity and quality processes.

Water quality monitoring is the effort to obtain quantitative information on the physical, chemical, and biochemical characteristics of water via statistical sampling. Besides collecting data, monitoring activities cover the subsequent procedures, such as laboratory analysis, data processing, storage, and data analysis to produce the expected information. Separate QA programs exist for both the field sampling procedures (collection, preservation, filtration and shipping components) and analytical procedures (laboratory component) (Chapman, 1992). Therefore, QA is essentially the management system that operates to ensure credible results. The Quality Control (QC) component of this system is a set of activities intended to control the quality of the data from collection through to analysis. Together, quality control and quality
assessments operate as a feedback system throughout the duration of the sampling program to provide early warnings of dubious data. Additionally, this feedback is the primary tool to determine if the current monitoring effort (i.e., site locations, sample frequency, and selected variables) meets the program objectives.

The prime objective of the field QA program is to improve accuracy by reducing introduced variability. Accuracy is the degree of agreement of a measured value with the true value of the quantity (variable) of concern (Kneese and Bower, 1988). Both random and systematic errors are factors that reduce accuracy and therefore, these errors must be minimised. Random errors refer to the precision (or random variation) of the data, while systematic errors refer to bias (or systematic deviation) in the data (Krenkel and Novotny, 1980).

DRI implemented its QA program, as a part of the National Drainage Water Quality-Monitoring Program to minimise the variability in the collected drainage water quality data.

The primary objective of this study is to present the data handling system in Drainage Research Institute as a part of the QA of the National Drainage Water Quality Monitoring Program.

DATA HANDLING PROCEDURES

Data and information collected from a comprehensive monitoring network are handled with complete interactive application relational database (Microsoft Access) designed for that purpose with some supporting programs written in VB (Visual Basic) and querying in Sequential Querying Language (SQL). The main objective of checking the data that are originally entered in the database is to obtain reliable data in the water quality database. Water quality is variable by nature, however it appears that some data originally stored in the database show extreme values that cannot be explained easily. Therefore it is necessary to check the validity of these values on a monthly basis. However, outliers are not necessarily wrong, there is just an explanation needed to be able to trust the results. Usually unusual high or low concentrations do not come alone.

If more variables show unusual concentration levels, the figures are probably correct, but the conditions in the field at the time of sampling may be different from normal causing excessive pollution or dilution;
- an extremely high value of Biochemical Oxygen Demand (BOD) is OK if at the same time the Chemical Oxygen demand (COD) reaches a peak and e.g. the Dissolved Oxygen (DO) is low;
- high BOD and COD concentrations can coincide with high ammonia levels;
- during the closing period of the irrigation period salt concentrations rise: this must be seen for each individual ion, otherwise other mistakes are more likely;
- high Coliform content might coincide with high BOD-levels.

Figure 1. Flow chart of water quality data handling in DRI from field to database
If only one variable has an unusual high or low concentration it is likely that there are more locations that show the same problem (combination in one field trip, combination in one round of analyses) or e.g. for this variable at this location an outlier is produced every time.

If only one variable has an unusual high or low concentration and it occurs only at one location only at only one occasion a typing error is the most likely mistake.

A schematic representation of the procedure of collecting water quality data from field trip until building the database is shown in Figure 1.

The following sections present the procedures for checking the data in DRI through the National Drainage Water Quality Monitoring Program.

**QUICK SCAN**

**Field data**

*Check field forms*

As soon as the field team finish their field data entry (the day after the return from the trip), the data analysis team receives the forms from the field engineers and starts checking the field forms and field database for that trip. Possible remarks may be made to field forms, completeness of the field form information (dates, names, instrument-identification, observations, etc.). Reviewing the calibration status and results for pH, EC and DO should also be performed. DRI approved acceptable ranges for these variables as following: 9.5 - 10.5 for pH, 1.36 - 1.46 dS/m for Electrical Conductivity (EC) and 95 - 105% for DO.

A quick scan of the actual data is needed to see whether there are any strikingly strange results. The number of locations, sampling duration, sampling intervals, field remarks and the database codes should also be reviewed. Special attention is paid to the DO mg/l; DO% of saturation and DO % is suspicious. Usually only when algal blooms occur at weirs and other structures and with sudden temperature changes.

All remarks are checked with the field engineer as soon as possible, to use the fresh memory of the engineer in explaining all remarkable features. Especially calibration results that are out of range give reason for action since reliable data depend on a correct calibration of instruments.

*Field data entry*

The field engineers enter the data of the field forms into the database. The first check of the database concerns the occurrence of any typing errors. This is best performed using a print of the relevant table from the database.

At this stage it is also useful to make graphs of the relation between several variables, measured with the same instrument. For example, the relation between EC, Total Dissolved Solids (TDS) and Salinity or DO (mg/l) and DO(%) should be checked. These graphs help to detect obvious typing errors and other strange features, like differences between different instruments used in different field trips.

*Corrections in database*

When all field data have been checked, the first step is to make the appropriate changes in the database if any mistakes have been found. After receiving the results of laboratory analyses more extensive checks can be performed.

**Laboratory data entry check**

*Arrival of laboratory data*

As soon as the laboratory is finished with their data, the data analysis team receives the data from the laboratory engineers and start with the check of the laboratory database for that month.
Check of laboratory data

As soon as the results of laboratory analysis come in a quick scan is performed. The data analysis team meets and reviews printouts and discusses observations, on the same day if possible. The team should look at missing values (completeness of the data set), obvious strange values and correct use of sampling date etc. The observations are written down and given to the laboratory on the same day so that the laboratory has time to re-analyse.

Corrections in database

Again the appropriate changes in the database have to be made after receiving the reaction from the laboratory if any mistakes have been found or incomplete data can be completed. This has to be done before continuing with the next step, the more extensive logical check. Any comments on the reaction from the laboratory should be written down.

Logical check

After obtaining the complete data set for one month, a more extensive logical check can be performed, existing mainly of three items checking individual variables, comparison of field and laboratory measurements and checking correlation between several variables. To find or explain any outlying results it may sometimes be necessary to look into the data of individual locations.

Looking at the data can be performed using subsets of variables Oxygen budget, Salts, Nutrients, Trace metals and visible pollution Bacteria.

Checking individual variables

Graphical analysis

For each region graphical output of concentrations categorised per month over a period of 13 months is made. This should be done for each variable. In this way both extreme outliers can be detected as well as changes in the range of values in the course of time. Through the graphs, unusual high or low results should be checked. Special attention is needed for order of magnitude differences between sampling campaigns, these differences might reflect calculation errors or dilution errors in the laboratory.

In general data, which is outside a certain range, could be considered as outliers and have to be checked for example:
- Whether they occur every time on that specific location;
- Whether they can be explained by other variables;
- If not: whether they are correctly analysed or entered in the database.

Any mistakes found in the values, by checking with the engineers in the laboratory, should be reported and corrected in the database.

Figure 2. Cadmium concentration in Eastern Delta (July 99 to August 2000)
Figure 2 shows the cadmium concentration in Eastern Delta for the period from July 1999 to August 2000. It was clear from the graph that three values were relatively higher than the other concentrations. These three values were reported to the lab to be checked. The lab commented that the value 0.14 mg/l was a typing error and should be replaced to 0.01 mg/l. The other two values were correct, even though they are strange values.

Figure 3 shows the change of temperature in Eastern Delta for the period from July 1999 to August 2000. It was clear from the graph that again three values were relatively lower than the average temperature in this period. These three values were reported to the field engineers to be checked. The result was that there were typing errors and should be replaced to 22, 25 and 26 instead of 12, 15 and 16 respectively.

**Statistical analysis**

Since it is not possible to look for all strange values from graphical presentations a statistical analysis is introduced, using validated data of the previous period. In this analysis new data are compared with the old data of the same location.

A confidence interval is calculated according to the following formulas:

- Lower limit of confidence = average - 2*σ
- Upper limit of confidence = average + 2*σ

It should be reminded that the application of this method requires a number of conditions, e.g. normal distribution of the data and the absence of seasonal fluctuations in the data. However this method is only used for checking the data. Any of the new data that lie outside the confidence interval of the location need to be checked, either with the field engineers or the laboratory engineers.

**Comparison of field and laboratory data**

Some variables have been measured in the laboratory as well as in the field. In principal the measurements in the field are more accurate, since they are performed on the 'fresh' sample. However the comparison between field and laboratory measurements may indicate changes in the sample:

**EC:** There should not be a large difference between both measurements (+/- 10%). Larger differences should raise suspicion, for something might have happened with the sample, or the instruments may not have been calibrated correctly. A check has to be performed using the analyses of individual ions to find which measurement is most probably correct; the not correct value than should be deleted from the database.

**pH:** Large differences between lab and field for pH might indicate change of the sample during the field trip or at the laboratory, caused by biochemical processes possibly resulting in false results for e.g. nutrients and HCO3-. Changes in pH of more than ±0.5 pH-unit are acceptable; larger changes and either the result of incorrect calibration or
changes in the sample. Incorrect calibration can be observed from the field forms; changes in the sample should raise suspicion about the laboratory results.

**Turbidity:** The measurement of Turbidity show very large variations; this is caused by the handling of the sample both in the field and in the laboratory, it is not easy to draw conclusions from the differences between these measurements, comparison with Total Suspended Solids (TSS) and Visibility measurements might help.

**TDS:** Although both values are calculated (field: calculated from EC field & EC lab: calculated from individual elements) it is a useful comparison since it may indicate typing errors or errors in analysis of individual elements or data entry. When differences larger than 10% occur, a further comparison with EC and ions is useful.

Graphical presentations give a quick overview of the relations and an indication whether any errors can be expected. However a table, made in Microsoft Excel with differences between field and laboratory measurements could be used as an easy way to check the original data sheets with either field or laboratory engineers.

**Relation between different variables**

For some variables it is possible to locate mistakes by relating them to other variables. The possible relations are listed below, graphical presentation or the calculation of ratios is the easiest way to find 'strange' values. Relations usually show an almost straight line. Outliers can easily be detected.

**Oxygen budget**

- Relation between BOD and COD \((BOD < COD!); \text{ usual range: } 1.5*\text{BOD} < \text{COD} < 10*\text{BOD})
- High BOD and/or COD can coincide with high ammonia levels
- High BOD and/or COD usually do not coincide with high DO levels

**Salts**

- Correlation between EC, TDS and salinity:
  - EC-field & TDS-field (should show a straight line)
  - EC-field & salinity (should show a straight line)
  - EC-lab & TDS-lab
- High EC/TDS/salinity should be explained by high levels of individual ions:
  - Total ions & 10*EC
- Relations between individual ions:
  - Sodium (Na) & Chloride (Cl)
  - Na & Cl+ SO4 (Sulphate)

Figure 4 shows relation between the EC (dS/m) and TDS (mg/l) in Eastern Delta for the period from July 1999 to August 2000. It was clear from the graph that there is a clear correlation between the two variables and there are no strange values during this period.

**Nutrients**

Usually Nitrate and Ammonia are not at a high level at the same time \((\text{NO}_3^-\text{N} & \text{NH}_4^+\text{-N})\).

**Trace metals**

According the previous experience in DRI, It was agreed that no very high temporal fluctuations can be expected in trace elements. Also no occurrence of order of magnitude differences between sampling campaigns. If any one of those happen that might indicate errors in laboratory analyses (calculation or dilution). Other consideration could be taken into account:

- To check outliers in Cadmium (Cd) and Zinc (Zn) it can be checked whether these metals show the same pattern: this is most common, since they show the same chemical behaviour.
- When related to point sources of pollution, concentrations of different metals are expected to be higher at the same location.
Physical variables

Correlation between TSS & Turbidity and visibility could be checked. These variables might show a similar pattern. Total Volatile solids (TVS) is approximately 10% of TSS (normal range 5-15%, but it depends on the composition of the suspended material. Other correlation should be taken into account (TSS & Visibility, TSS & Turbidity and Visibility & Turbidity).

Bacteria

High Coliform content might coincide with high BOD-levels.

Figure 5 shows an example for checking the related water quality variables (BOD, COD and NH₄ mg/l) in Eastern Delta for the period from July 1999 to August 2000. It is clear from the graph that the three variables almost have the same pattern.

Submit check to laboratory

All comments and questions are written down and submitted to the laboratory for further checking using the original notes.

Corrections in database

When the laboratory comments come in, the data are updated with corrections from the laboratory.

Any comments on the reaction from the laboratory should be written down.

Figure 5. Checking related water quality variables (BOD, COD and NH₄) in Eastern Delta (July 99 to August 2000)
Data elimination

After all checks have been performed, it is still possible that highly unlikely data are entered in the database. According to the previously approved rejection criteria, this data will then be eliminated from the database (see Table 1).

Table 1. Rejection Criteria Approved in DRI

<table>
<thead>
<tr>
<th>Variable(s)</th>
<th>Criteria</th>
<th>Remarks</th>
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<tbody>
<tr>
<td>BOD and COD</td>
<td>$\text{BOD} \geq 1.05 \times \text{C-OD}$</td>
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<tr>
<td>DO (mg/l) and DO (%)</td>
<td>$\text{DO} &gt; 110%$</td>
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<tr>
<td>TDS and data of the individual ions</td>
<td>$\text{TDS} &lt; 300 \text{mg/l}$</td>
<td>TDS of Nile water close to Delta Barrage = 300 mg/l</td>
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<td>$\text{EC}<em>{\text{field}}$ or $\text{EC}</em>{\text{lab}}$</td>
<td>$\frac{\text{EC}<em>{\text{field}}}{\text{EC}</em>{\text{lab}}} &gt; 1.1$ or $\frac{\text{EC}<em>{\text{field}}}{\text{EC}</em>{\text{lab}}} &lt; 0.9$</td>
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<tr>
<td>$\text{NO}_3^-\text{N}$</td>
<td>$\text{NO}_3^-\text{N} &gt; 50 \text{mg/l}$ and no high levels of BOD and/or Coliforms occur</td>
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<tr>
<td>$\text{NH}_4^+\text{-N}$</td>
<td>$\text{NH}_4^+\text{-N} &gt; 50 \text{mg/l}$ and no high levels of BOD and/or Coliforms occur</td>
<td></td>
</tr>
<tr>
<td>TSS and TVS</td>
<td>$\text{TSS} &lt; \text{TVS}$</td>
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<tr>
<td>Trace metals (Cu, Fe, Mn, Zn, Pb): extremes</td>
<td>individual values (not several in a row) exceed 5x or are less than 0.2x the average per location</td>
<td>No very high temporal fluctuations can be expected</td>
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Documentation

To keep track of changes made in the database after each sampling campaign the graphs and comments and replies from the lab (data tracking sheet) should be stored in a folder.

A data-tracking sheet is available to allow a proper assessment of the progress and status of the data from each monthly survey. These sheets should be used and be available in a folder for easy access for evaluation.

CONCLUSIONS

Although the management information system designed and developed by the Drainage Research Institute is newly applied in the drainage water quality monitoring in Egypt, it provides a good tool to keep track the water quality data in order to minimise as much as possible the human errors. It could be introduced as a part of the quality control activities in the water quality-monitoring networks, to provide timeless and reliable data. The situation enables to support and provide formulated strategies and polices issued by water resources decision-makers with information regarding the water quality.

The data handing protocol developed by the data analysis team of DRI can be adopted to be applied for other waterways and other involved institutes in the field of water quality monitoring.

REFERENCES

IMPROVEMENT OF STAGE-DISCHARGE RELATIONS AND ITS EFFECTS ON DISCHARGE-RELATED QUANTITATIVE COMPUTATIONS

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Since the beginning of the 20th century discharges in the Rhine and Meuse have been measured periodically using flow-velocity measurement apparatus such as the Ott-mill. On the basis of these data stage-discharge (Q-H) relations are drawn up from time to time, with which a continuous discharge record is computed from daily or hourly measured water levels. The results are stored in the main data recovery system DONAR of Rijkswaterstaat. A re-evaluation of historic Q-H relations has revealed that the DONAR-data show various inaccuracies as a consequence of neglecting the influence of hysteresis in the Q-H relation, and of the fact that bed level changes in time were not taken into account correctly by the procedures followed. A more correct procedure for drawing up Q-H relations for the main measurement stations along the Rhine and Meuse has not only yielded more accurate daily discharge data for the period 1956-2000, but has also resulted in a new concept for the Q-H relation. In this so-called Qf-relation, hysteresis and bed level changes are incorporated explicitly. Besides a description of methods and results, the paper gives some examples of systematic and variable errors in the DONAR-data and their propagation into computed annual suspended sediment and chloride loads for the Rhine at Lobith.

INTRODUCTION

Discharges in the main Dutch rivers Rhine and Meuse are already for decades determined in the field using the Ott-mill (see Figure 1). One discharge determination at a specific location requires about a quarter of a day, which makes these measurements quite expensive. Therefore this is only done periodically. Figure 2 shows the discharge measurements carried out in the year 1993 in which an extreme high flood occurred. As may be noticed, normally the measurement frequency is intensified at high floods.

Figure 1 With Ott-mills flow velocities are measured at different fixed depths in a series of prescribed verticals, after which the total discharge through the river cross-section is computed by integration with the velocity-area-method.
CONVENTIONAL DETERMINATION OF STAGE-DISCHARGE RELATIONS

Starting with the measured discharges over a certain period of time, the normal procedure has always been to establish a relation between the discharge $Q$ and the water level $H$ at the time of discharge-measurement: the so-called $QH$- or stage($H$)-discharge($Q$)-relation. Figure 3 gives an example of the $QH$-relation which could be drawn up by means of polynomial regression analysis for the year 1993.

THE HYSTERESIS EFFECT

At first sight (upper graph in Figure 3) such a $QH$-polynomial looks quite good, but when one looks at the residue in more detail, a relatively poor accuracy results. The deviations from the measured values range up to 4%. At lower discharges this is due to normal instrumental errors (i.e. imprecision of the method of discharge determination), at higher discharges this is mainly due to neglecting the influence of hysteresis. Hysteresis is the phenomenon by which, at a given water level, in the rising stage of a flood wave a higher discharge occurs than would be expected at permanent flow conditions, and a lower discharge in the falling stage. The influence of hysteresis during the December flood of 1993 is illustrated by Figure 4. Especially the lower graph shows the strong underestimation of the discharge in the rising stage of the flood wave (up to 7%), and a similar overestimation in the falling stage. The magnitude of the over/underestimative effect depends on the steepness of the flood wave, hence on the rising velocity ($\partial H/\partial t$) in time.

THE BED LEVEL EFFECT

The river bed level at the measurement site may change due to morphological processes, by which the discharge will increase at a given water level in case of erosion, and decrease in case of sedimentation. Hence, the period over which a $QH$-relation is established should be of such length that one may account sufficiently for the morphological changes in time. For the Upper Rhine a more or less continuous drop in bed level has occurred since 1926 due to an incision process, hence, the $QH$-relation has shifted gradually in time.
Figure 3  Stage-discharge-relation for the Upper Rhine at Lobith for the year 1993 and residual values.

OTHER INFLUENCES

In general, the $QH$-relation may also be influenced by:
- the presence of downstream weirs influencing the backwater-curve, and hence the water level at given discharge;
- changes in the geometry of the cross-section at the measurement site and changes in hydraulic roughness at, as well as downstream the measurement site;
wind forces acting upon the water surface, especially in the direction across the river, creating a cross-slope and therefore the water level at given discharge. In the case of the Lobith station the influence of downstream weirs has been well established (see also further in this text), but no statistical sound proof could be established concerning the influence of wind.

Fig 4 Reproduction of the measured discharges during the December 1993 flood at Lobith and the influence of hysteresis. The strong overestimation of the peak discharge (4%) is probably due to (gross) measurement error.

RE-EVALUATION OF QH-RELATIONS FOR LOBITHE

Although discharges have been measured at Lobith since 1900, and a continuous series of daily discharges has been derived over the period 1901-present, the original measurement data are only available from 1956 on. This data set has been analysed by means of least-squares multiple regression analysis, which yielded a set of so-called Qf-relations for Lobith which reproduce the measured discharges in a more accurate way than the QH-relations (based on water levels only) did. The indication "Qf" for this type of stage-discharge relation has been introduced so as to express the idea that the discharge is not only dependent on the water level H, but also on:

- the change in water level in time (∂H/∂t) around the measurement time as a predictor of the hysteresis effect. Theoretically this should be the actual slope in the water level (∂H/∂s), but since water level slopes in rivers are extremely difficult to measure with sufficient precision, the change in time is used as being sufficiently representative of this. For the Upper Rhine at Lobith, the best results are obtained when a period of 6 hours around the measurement time is considered;
• a time-parameter (day-number) in order to follow the bed level changes;
• various parameters concerning the influence of the downstream weir at Driel (see Fig 5),
such as the fall in water level over the weir.

In order to follow bed level changes optimally, the period 1956-2000 had to be subdivided into
7 sub-periods, each having its own $Q_f$-function.

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**Figure 5 Rhine and branches near Lobith**

**COMPARISON OF OLD AND NEW DISCHARGES**

The daily discharges for the period 1901-present mentioned before are computed with $QH$-relations determined in the conventional way, hence with the water level as the only variable. These data are stored in the main Rijkswaterstaat data archive for the water sector called DONAR. Figure 6 gives an impression of the reproduction of the measured discharges from the period 1956-2000 (totalling ca 1550 measurements) by the DONAR-values.

At the level of the average annual discharge for the Upper Rhine at Lobith (ca 2200 m$^3$/s) the prediction error is 4.3%. The point measurement error at that level - representing the lower boundary for the prediction error to be achieved - is about 1.3%. (All errors being defined at the confidence level of one standard deviation). The right hand graph of Fig 6 gives the result for the $Q_f$-discharges. The improvement in terms of the prediction error amounts to one full %, which has to be seen in connection with the fact that the distance to the point measurement error (3% for the DONAR-discharges) has been diminished by 30%.

The prediction error treated until now is the overall error. The gain in precision at high discharges can be much greater, as is shown by Figure 7. As for the improvement attained until now, it has to be noted that for arriving at the present state the hysteresis variable ($\frac{\partial H}{\partial t}$) has been computed on a 24hr-basis, hence that further improvement may well be possible when a shorter period (i.e. 6 hr) is applied. However, this is only possible for the years after 1-1-1989, since before that time only daily (8:00 MET) water levels were recorded.
SYSTEMATIC DEVIATIONS IN THE DONAR-DISCHARGES

Onto this point only variable errors have been discussed. The variable errors may be attributed
to neglecting hysteresis, imperfections in the description of the influence of impounding, etc.
The analysis of long term differences between the DONAR-discharges and the Qf-discharges
has revealed that the DONAR-discharges also show significant systematic deviations over
periods of years.

\[
s.e. = 120.8 \text{ m}^3/\text{s} \ (4.3\%)
\]

\[
y = 1.004x + 26.52 \\
R^2 = 0.9965
\]

\[
s.e. = 94.0 \text{ m}^3/\text{s} \ (3.3\%)
\]

\[
y = 0.9975x + 4.02 \\
R^2 = 0.9979
\]

Figure 6 Variable prediction error in the reproduction of measured discharges in the period 1956-2000 by DONAR- and Qf-values. The (instantaneous) measured values have been converted to daily values by means of a water level correction. s.e. is the standard error of prediction, the % error is related to the average measured discharge.

Figure 7 Residual values of measured discharge versus DONAR-discharge and of Qf-discharge versus DONAR-discharge for the flood period of December 1993 at Lobith. The top graph shows the deviations in the DONAR-discharge as a result of neglecting hysteresis, the lower graph shows in how far the Qf-discharge eliminates these deviations. Note also the systematic underestimation of the discharge by DONAR due to bed level changes (see also further in text).
Figure 8 gives the annual deviations between the two types of discharge over the period 1956-2000. The varying systematic underestimation in various periods is a consequence of the way in which the DONAR QH-relations determined on basis of measurements carried out in a specific period are applied to the proper period. If this is done properly, like in the period 1960-1975, the QH-relation is applied to the period in which the measurements were carried out. In more recent years, however, it was decided to apply the newly determined QH-relation to the period after the measurement period - in order to prevent recalculation of already published data - from which a serious underestimation results. As a result of this, the annual discharge over the period 1987-1997 deviates with an amount of ca 80 m³/s or 3.5% of the Qf-discharge. This deviation happens to coincide with a difference that had already been established between the DONAR-discharges and those measured by the German water authorities near Lobith. It is not yet clear why in the period 1956-1960 an even larger deviation occurs.

EFFECTS ON (ANNUAL) LOAD COMPUTATIONS

The DONAR-discharges are widely used to compute annual loads of a range of solute and suspended substances monitored at Lobith. It is obvious that the deviations treated before will have their effect on the results of such computations. Two types of substances may be distinguished as for their specific behaviour in this sense:
1. Solute substances due to more or less constant emissions which show a dilutive effect at rising discharges, like chloride;
2. Suspended matter (solid particles, humus) which is taken up by the river itself and therefore increases at rising discharge.

Figure 9 shows the development of the suspended sediment load during a high flood in 1993. It may be noticed that the sediment concentration reaches its highest peak just before the flood peak (upper and middle graph), exactly when the hysteresis effect is strongest. From this, a maximum deviation in the computed load results (lower graph). For the chloride load this typical sensitivity to the hysteresis effect does not occur, as is illustrated by Figure 10. One would expect the hysteresis effect in case of the suspended load to propagate quite strongly into the annual load too, but this is not so much the case for the Upper Rhine. As Figure 11 shows, the net annual deviations for the suspended sediment and chloride load are comparable. This is caused by the fact that only a relatively small portion of the annual suspended sediment load of the Upper Rhine is delivered by high floods. For a typical rainfall-type river as the Meuse, in which ca 80% of the annual load is delivered by floods, this will be quite different. In both cases, however, the loads are systematically underestimated due to the underestimative effect explained before. As can be seen from Figure 11, an average underestimation of ca 3% results in both cases due to the bed level effect, with an extreme up to 6% in 1989.
Figure 9 Suspended sediment concentrations and loads during a high flood in the Upper Rhine at Lobith in 1993.

Figure 10 Chloride concentrations and loads during a high flood in the Upper Rhine at Lobith in 1993.

Figure 11 Deviations in annual loads of suspended sediment and chloride over the period 1979-2000 as a result of using DONAR-discharges.

CONCLUSIONS AND DISCUSSION

As can be shown from the results of the re-evaluation of Q-H-relations carried out for a number of measurement stations along the river Rhine and Meuse, the traditional method of drawing up such relations in the univariate way - hence, with the water level as the only variable - appears to be quite inaccurate in several ways. For the Upper Rhine at Lobith, neglecting hysteresis
results in deviations up to 7% during the rising and falling stages of flood waves, while not incorporating bed level changes properly easily results in systematic deviations up to 4% over periods of years. The new concept represented by the $Qf$-relation, in which hysteresis, bed level changes and other influences concerning the layout and properties of the river channel are explicitly accounted for, eliminates these errors already for a substantial part. The higher precision of newly computed discharges for the Rhine and its branches has already contributed considerably to the calibration and verification of the hydraulic model WAQUA with which design flood levels are predicted at Rijkswaterstaat.

For the Rhine and its branches, as well as for the Meuse, $Qf$-relations could only be drawn up from the year 1956 on since the original data on discharge measurements for the period before 1956 are no longer available. As a result, the daily discharge series - ranging from 1901 to present for the Rhine and 1910 to present for the Meuse - is inhomogeneous. It is considered to be feasible to derive a correction formula for the hysteresis effect based on the day-to-day rise and fall of the water level ($\partial H/\partial t$), but it will be more difficult to correct for bed level changes. Data on bed level changes are available since 1926, but the reliability of them is still questionable.

In the present practice, a straightforward multiple regression technique is used for the determination of $Qf$-relations in order to establish the physical background of the method in a clear way. For operational purposes, however, a method based on the neural network technique is recommended. Within the present study, a first limited survey with neural networks for Lobith has already yielded promising results, as is shown by Haskoning (1998). The advantage of such an approach is that data for quite different periods are interrelated automatically into one main model, by which also the implementation of new discharge measurement data will go more smoothly than with conventional regression techniques. This is of great importance for the maintenance of the $Qf$-relations at operational level.

REFERENCES


PRACTICES AND INTEGRATED APPROACH FOR MONITORING OF LAKE LADOGA, RUSSIA

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2 Institute of Limnology RAS, Sevastyanov Street 9, 196105, St. Petersburg, Russia

Environmental monitoring programmes have been conducted in Lake Ladoga since the late 1950s. These have included studies on hydrology, water chemistry and biology. Regular integrated studies of the lake are presently conducted by seven institutes. Long-term investigations, spanning some 40 years, serve as a good basis for understanding of the processes as well as for the assessment of the current environmental conditions in the lake. At present the administrative responsibilities related to monitoring are divided between a large number of authorities whose duties arise from a large number of different pieces of legislation. This makes co-ordination a difficult task and it is also difficult to change the overall monitoring.

For a long time the main monitoring aim for Lake Ladoga was to determine the natural fluctuation of chemical and biological variables and to evaluate the process of anthropogenic eutrophication. During the last two years a new integrated monitoring programme has been worked out as part of an international project. The proposed integrated monitoring programme is composed of the following studies: climatological, hydrological, loading, water chemistry, plankton, benthos, macrophytes, periphyton, palaeolimnology, sediment chemistry, ecotoxicology and fish. Some long term monitoring results and new approaches for monitoring and options to be considered for monitoring of Lake Ladoga will be discussed. One of the conclusions is that the proposed integrated monitoring programme should verified and regularly reviewed.

INTRODUCTION

Lake Ladoga (17 891 km²) is the largest lake in Europe; its mean depth is 47 m, maximum depth 230 m, and its 258 000 km² drainage area extends to much of northwestern European Russia and eastern Finland. The lake is an important link in the Volga-Baltic-White Sea waterway system. The main flow of transport passes along the Neva and the Svir rivers.

The ecological state of Lake Ladoga seems to have deteriorated, especially since the 1970s. The ecological condition of Lake Ladoga is of particular concern to the 6 million inhabitants of St. Petersburg and Leningrad area for whom Lake Ladoga is the only source of domestic and industrial water. Furthermore, the condition of Lake Ladoga affects water quality in the Neva River, the Gulf of Finland and the whole Baltic Sea.

Lake Ladoga has been monitored since the late 1950s (Table 1). Monitoring programmes have included studies on hydrology, water chemistry and biology (phytoplankton, zooplankton, benthos, macrophytes and fish). Regular integrated studies of the lake are conducted by two institutes of the Russian Academy of Sciences and five other organisations (Viljanen & Drabkova 2000a, 2000b). Long-term investigations, which cover 40 years, serve as good basis to understand the processes as well as for the assessment of the current environmental conditions in Lake Ladoga. At present the administrative responsibilities related to monitoring are divided between a large number of authorities whose duties arise from a large number of different pieces of legislation. This makes co-ordination a difficult task and it is also difficult to change the overall monitoring. State monitoring of aquatic objects is conducted on local, territory and federal levels.

For a long time the main monitoring aim for Lake Ladoga was to determine the natural fluctuation of chemical and biological variables and to evaluate the process of anthropogenic eutrophication (cf. Petrova & Terzhevik 1992, Rumyantsev et al. 1999, Viljanen & Drabkova 2000a, 2000b). Recent research on Lake Ladoga has been summarized by Viljanen et al. (1994), and in the proceedings-volumes of three International Lake Ladoga Symposia, arranged in 1993, 1996 and 1999 (Simola et al. 1995, 1996, 1997, Peltonen et al. 2000). The monitoring, environmental data systems and numerical modelling of Lake Ladoga and some other large lakes have been described in the publications of three workshops (Peltonen et al.)
Recent monitoring methods of Lake Ladoga have been further described by Holopainen et al. (1999, 2000), and in the summary and conclusions of a Tacis-project (1998-1999) (Viljanen et al. 2000).

The aim of this research is to describe the practices for monitoring of Lake Ladoga, evaluate the long-term changes and state of the lake as revealed by some chemical and biological variables, and present new practices and integrated approach for monitoring of the lake. This paper is based on the results, articles and proceedings mentioned above.

Current practices for monitoring of Lake Ladoga

Regular integrated studies of the lake were started by the Institute of Limnology (St. Petersburg) in 1956, and other organizations in the 1960s, the 1980s and the 1990s (Table 1) (Viljanen & Drabkova 2000a, 2000b). The number of stations varied between the institutes and years, rising up to some 80 in the 1990s. Surveys were repeated usually 2-5 times during the summer season. The assessment of state of the lake and the classification system included parameters that are generally considered to be important indicators of water quality in a wide sense, and for which methods and a reasonable amount of data were available (cf. Holopainen et al. 2000): hydrology, chemistry (water and sediment) biology (phytoplankton, zooplankton, benthos, macrophytes, periphyton; species composition, numbers and biomass) and palaeolimnology (sedimentary diatoms); in the 1960s respiration studies and the 1990s hygienic and bacteriological parameters and ecotoxicological observations have been included in the monitoring. Fish monitoring and stock assessment have been conducted since 1940s, in the southern and central part of the lake.

Regular observations of the material inputs into Lake Ladoga and output from the lake are conducted in the downstream parts of the main inflowing rivers (11 rivers) and the outlet of River Neva since 1959. Initial sampling frequency was 3-6 times per year, rising up to 6-11 times in the 1990s, with observations covering all the seasons. Sampling took place in the beginning of monitoring 3-6 times per year up to 6-11 times in the 1990s. The observations covered all the seasons. The number of parameters has increased during the monitoring (Holopainen et al. 2000, Viljanen & Drabkova 2000a, 2000b).

Table 1. Monitoring of Lake Ladoga by different organizations (1945-1999) (Viljanen et al. 2000).

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Long term changes and present state of Lake Ladoga

Until the early 1960s, Lake Ladoga was oligotrophic and characterized by good water quality, but within the past 20-30 years, as a result of human impact, the ecological state seems to have deteriorated (Rumyantsev et al. 1999). Especially since the 1970s, its trophic state has changed to mesotrophic, with elevated nutrient concentrations and decreased transparency. Conditions at some of the worst polluted sites have actually improved in recent years, due to closing down of some sources of industrial pollution, but there are alarming signs of general eutrophication of the main body of water.

The obtained monitoring data has revealed that the degree of human impact, e.g. eutrophication, varies in different time periods and areas of the lake, and the indication of eutrophication by different parameters and analyzes does not always agree. Transparency recordings exist since the beginning of the 20th century. In spite of sparse measurements, some indication of deteriorating water quality can been seen as decreased Secchi depth values in the deep northern part of Lake Ladoga. In August 1900 the maximum Secchi depth was 7 m (Witting 1929) but 90 years later it was less than 3.0 m (Figure 1). In the shallow southern part of the lake, where large rivers increase the turbidity, the Secchi depths have been more stable during the 20th century. Increased loading has also affected conductivity values. Since the late 1960s the conductivity of the water has increased gradually from 75 µS cm\(^{-1}\) to 87 µS cm\(^{-1}\) (Figure 1). The trend is evidently due to human impact (Petrova 1982, Krychkov et al. 1992).

Figure 1. Secchi depth (m) in August, average conductivity (µS cm\(^{-1}\)) in the pealgial and average total phosphorus (µg l\(^{-1}\)) during growing season in whole lake and in nearshore zone of Lake Ladoga.
The nutrients, especially phosphorus have been studied regularly since 1976 (Figure 1) (Lozovik & Raspletina 1999). In the whole lake the phosphorus concentration appeared to have decreased between 1976 and 1998, whereas in near shore stations the variation is large and no clear trend can be detected. According to recent observations the phosphorus content of Lake Ladoga has decreased by 30% from the 1970s value, associated with economic regression. In Lake Vättern, Sweden after water pollution control the reduction has been as much as 50% during the period of 1967-88 (Willén 1992).

In large lakes the spatial variation of water quality can be considerable; in Lake Ladoga there are great horizontal differences in pelagial areas and between single isolated bays and bays loaded by large rivers. In August 1995 Secchi depth varied from 1.4 to 3.3 m and total phosphorus concentration from 13 to 73 µg l\(^{-1}\) and chlorophyll a concentration from 3 to 23 µg l\(^{-1}\) in the surface water (Figure 2) Thus, representativeness of the sampling network is of crucial importance in planning of the monitoring programme.

The plankton communities of Lake Ladoga have been studied intensively during the last century. Regular monitoring work of phytoplankton started in early 1960s (Figure 3). Since the beginning of the 1960s, notable increase of algal densities, especially those of blue-greens, has been observed in the heavily loaded near-shore areas (Petrova 1987, Petrova et al. 1992, Holopainen et al. 1996). Due to wind induced currents, occasional blooms of blue-green algae have also been observed in the pelagial areas (Holopainen & Letanskaya 2000). However, in the central pelagial the changes in the phytoplankton biomass chlorophyll a have been small during all the period (Letanskaya 2000). The decreased nutrient load in 1990s can not be seen as decreased biomass or recovery in species composition in the loaded bays. Zooplankton data has been collected since the 1940s in Lake Ladoga. Two pelagial stations have been chosen to represent the changes in zooplankton community. In the 1990s the total biomass has increased and the vertical distribution has changed as compared with earlier observations (Figure 4), which is due to sparse material and varying methods in zooplankton sampling and calculating the biomass. According to Andronikova and Avinski (1997), mass development of *Asplanchna priodonta* is evident in the surface layers (0-5 m), high biomass of *Bosmina longispina* in the 10-20 m layer, and in deep water layers large copepods constitute the major part of the biomass.
Figure 3. Average phytoplankton biomass and chlorophyll a content in Lake Ladoga in 1960-98.

Figure 4. Vertical distribution of zooplankton in the pelagic zone of Lake Ladoga in stations 55 and 82 at depth of 70 m (modified from Andronikova 1996).
According to Slepukhina et al. (2000) long-term observations have revealed that during the last 40 years the macrozoobenthos fauna of the deep central part of Lake Ladoga has not changed significantly. However, in shallow areas the biomass of macrozoobenthos has increased, but the variation between years is quite large (Figure 5). Due to changes of water quality the most pollution sensitive species have declined or disappeared during the 1970s and 1980s (Slepukhina et al. 2000). Following decline of human impact in the beginning of the 1990s, the ecological conditions of the lifeless bottom areas near the most heavily polluted places of Lake Ladoga have improved and the benthic communities have recovered (Rumyantsev et al. 1999, Slepukhina et al. 2000).

Figure 5. Changes of biomass of macrobenthos on station 55 in 1960-1998.

Fish catches have been studied in 1945-1998 in Lake Ladoga. The total catches have been biggest during the 1980s, when also notable changes in the fish community took place: the pike-perch catches reached their maximum and the whitefish catches decreased (Figure 6) (Kudersky et al. 1996).

Although the pelagial biomass of plankton and benthos have varied only slightly during 1960-1998, considerable changes are seen in the community structures (Figure 3, 4 and 5). The biological results emphasize the importance of identification work, standardised and comparable methods, representative sampling programme, and integration with hydrological and chemical measurements and fish monitoring.

Figure 6. Annual total fish, whitefish and pike perch catches in Lake Ladoga in 1945-1998.

Integrated approach for monitoring of lake ladoga - future prospects

The proposed ecological monitoring programme for Lake Ladoga prepared during the Tacis project includes surveillance, operational and investigative monitoring elements in accordance with the principles of EU Water Framework Directive (Holopainen et al. 2000, Viljanen et al. 2000). The proposal will be based on the long term results obtained and practices conducted by different organizations monitoring Lake Ladoga. A comparison of international monitoring practices of large lakes was also made.
The goals of the proposed monitoring programme of Lake Ladoga are: 1) describe environmental status and trends, and determine anthropogenic effects, 2) provide a basis for identifying and assessing environmental threats from local and river basin/catchment levels, 3) provide data to assist economic activities to develop in an environmentally sustainable way, 4) provide data for the evaluation of water management practices, and 5) provide data which will allow the assessment of specific effects of the discharges of different pollutants.

Analyses. An integrated set of several comprehensive investigation programmes is needed in the monitoring of Lake Ladoga. The proposed integrated monitoring programme is composed of the following studies (Holopainen et al. 2000): hydrology including climatological observations, loading, water chemistry, bacterioplankton, phytoplankton, zooplankton, macro- and meio-benthos, macrophytes, periphyton, palaeolimnology, sediment chemistry and fish. Ecotoxicology is a new approach to study the contamination in ecosystem, and it needs further investigations before it can be routinely applied to the monitoring. The proposed monitoring programme places greater focus on the use of biological indicators. The main parameters for water quality criteria are: (1) nutrients/eutrophication: total phosphorus, total nitrogen, (2) oxygen and oxygen consuming substances: oxygen content, COD, (3) transparency: colour, Secchi depth, (4) acidity/acidification: alkalinity, pH, (5) metals: Fe, Al, Mn, Cu, Pb, Cd, Cr, Zn, Hg, (6) plankton: phyto- and zooplankton (density, biomass, species composition, community structure), chlorophyll, (7) macrophytes: density, biomass, species composition, community structure, (8) periphyton: density, biomass, species composition, chlorophyll, (9) macro- and meio-benthos: density, biomass, species composition, community structure, (10) palaeolimnology: sedimentary diatoms, sediment chemistry, (11) fish: stock assessment, catch, community and age structure, ichthyopathologic measures. The programme is planned to include also satellite monitoring for detecting spatial and temporal differences of water quality variables (e.g. chlorophyll a, suspended solids, turbidity, transparency). Riverine input of...
nutrients and pollutants, diffuse pollution, point source discharges, and atmospheric pollution are used for the assessment of external loading and watercourses including following variables: nutrients, suspended solids, colour, COD, BOD, conductivity, phenols, hydrocarbons and metals. The detailed analytical and sampling methods employed in the monitoring of Lake Ladoga are presented in Holopainen et al. (1999).

Sampling stations, depths and frequency. Samples are to be taken mainly from those stations, which have been used in the ecological monitoring of Lake Ladoga since the late 1950s. There are stations in the deep pelagial of the lake, in the northern archipelago and in areas loaded by waste water (Figure 7). All stations are not sampled every time, there are joint integrated sampling stations and special stations for different parameters. In some stations, located close to point sources, the main subjects are benthos, macrophytes, periphyton and palaeolimnology. Hydrological studies e.g. discharge, currents and temperature are carried out at the length and transverse axis transect stations, which are mostly the same as for water quality. The fish studies are proposed to be extended to the whole lake area. The pelagial sampling stations located along the longitudinal axis and transect of Lake Ladoga give support for modelling purposes. Integrated data (complex data) on the distribution of phytoplankton, zooplankton and other hydrobionts are to be collected at certain pelagic stations in the two cross sections (Holopainen et al. 2000, Huttula et al. 2000). The number of the stations will be greatly reduced since the 3D model simulations will be used for getting information about unmonitored areas.

Sampling depths and frequencies vary and are presented in detail by Holopainen et al. (2000). The monitoring frequency has been proposed to be at least five times during the growth season. The main sampling effort will be concentrated to the spring season (late May) and to the end of stratification time (August). If the assessment is based on late season samples, data from a few consecutive years should be collated because of great interannual variation.

Further development of the integrated monitoring programme

- The proposed programme should be verified and regularly reviewed in order to take into account the dynamic changes in the emissions from the catchment area and resulting changes in risk factors to ecological systems, human health and sustainable use of aquatic resources of Lake Ladoga;
- Time series, including spatial distribution and temporal variation of hydrological, chemical and biological data should be analyzed statistically in order to establish the sufficiency and representativeness of the new sampling network;
- The data should be stored in harmonized database for further use, for numerical evaluation of the data some basic statistics and indices should be used, and new parameters, characteristics and indices should be sought for, tested and compared with the existing parameters;
- If the monitoring programme will be changed in future, parallel sampling and analyses should continue long enough to test the differences between old and new methods and analyses;
- Intercomparison of chemical analyses and taxonomic work should be organised between all the participating institutes and also with some institutes in foreign countries, and analyses should be carried out in laboratories which are provided with state accreditation;
- 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;
- Algorithms should be developed by which reference values may be estimated in the absence of sitespecific measured values;
- It is obvious that toxic contamination, which plays a key role in the constitution of water quality, has to be included in the monitoring - until now such investigations have been conducted only sporadically;
- Water management and monitoring should become more river basin directed; consequently in the case of Lake Ladoga, international co-operation in water management and monitoring becomes ever more important;
- On the basis of monitoring data and mathematical modelling, it will be possible to develop specifications for maximal, allowable impacts on the recipients;
- 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;
- Co-operation and co-ordination among authorities in the Lake Ladoga basin is necessary for improving monitoring, management and use of monitoring results. Institutional
arrangements in the form of a council for different authorities may provide one way of improving co-operation and coordination.

CONCLUSIONS

At present the administrative responsibilities related to monitoring of Lake Ladoga are divided between a large number of authorities whose duties arise from a large number of different pieces of legislation. This makes co-ordination a difficult task and it is also difficult to change the overall monitoring. In order to avoid overlap in monitoring work, cooperation and information exchange between the participating organizations should be improved and careful planning of the work is necessary.

The process of monitoring and assessment of Lake Ladoga should be seen as a sequence of related activities that begins with the definition of information needs and ends with the use of the information product for management decisions. A better connection should be found between the data produced by the monitoring networks and the information necessary for management actions. To optimise the monitoring cycle there should be an equilibrium between the goal of the information and the costs to obtain it. The tools used for monitoring should reflect the monitoring objectives.

Increased international co-operation, including several present and drafted environmental directives of the European Community, and the tendency to build up multi-purpose national monitoring, calls for re-evaluation of the different approaches for measuring environmental changes and to combine these in integrated monitoring programmes. At the same time transboundary and river basin issues have been emphasised in monitoring design. Traditionally, the different subprogrammes of lake ecosystem characteristics (e.g. water chemistry, plankton, zoobenthos and fish monitoring) will function separately, often indicating the same ecological quality criteria, and are largely overlapping in their monitoring purpose (e.g. Holopainen et al. 2000). Rather than ask if one really needs to measure eutrophication development "in a thousand different ways", it is better to appreciate the complementary role of different measurements and to widen the monitoring approach. New challenges in monitoring, including biodiversity, possible climatic changes, ecosystem health and ecological integrity are of a highly international and global character (Heinonen et al. 2000). Therefore, monitoring methods need still further development, but also require intensive co-operation in order to harmonize their use.

ACKNOWLEDGEMENTS

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WATER QUALITY TRENDS ASSESSMENT IN THE MORAVA RIVER TRANSBOUNDARY SECTION USING OECD METHODOLOGY

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The Morava River represents one of the most important tributaries of the upper course of the Danube River. Draining not only water but also pollutants from three neighbouring countries in its river basin - Austria, Slovakia and the Czech Republic - the Morava River water ecosystems have been exposed to a considerable stress caused by pollutants as well as by some anthropogenic activities. In connection with the activities of the Czech Republic in different international activities, esp. in Commissions for water protection, and also with the presupposed entering EU, the pressure on meeting requirements connected with the pollution of the environmental components has been intensifying.

As a basis for national and international activities within this region, the results obtained from the complex national water protection study Project Morava solved in the period 1996 - 1999 provided more detailed knowledge about the water quality research and other activities enabling the application in the state administration before entering the Czech Republic EU and also supporting international activities including fulfilment of the Convention for Protection and Sustainable use of the Danube River. One of the basic methods used in the above mentioned water basin for water quality assessment and its performance was OECD methodology which enables to cover the long-term trends of environmental aggregated indicators splitting up the problem of water protection into three mutually connected parts: "pressure - state - response". In the Project Morava five problematic tiers were solved by using time series of indicators covering the years 1970 - 1998. The evaluation of all the problematic tiers was based on the national requirements and also on final intentions formulated for the Elbe River within the framework of the International Commission for the Protection of the Elbe River (ICPER).

The results gained in the transboundary Morava River control site showed that the organic pollution, bacteria and toxic metals reach the national requirements laid down by the relevant Government Order. However, nutrients and specific pollutants do not meet these limits. As for the claims of ICPER heavy metals have not reach them. The application of the above mentioned method enabled mutual comparison of the state of the protection of two main Czech transboundary rivers - the Morava and the Elbe River. It also enabled formulation of the pollution sources requirements as the presumption of fulfilling the international aims dealing with required temporal levels.
NEW APPROACH IN MONITORING OF SURFACE WATER QUALITY IN NATIONAL NETWORK

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We would like to inform you about first results of “Complex monitoring of water quality in water courses in the Czech republic” project. The goal of this project was a preparation of long term program of chemical and ecological state of surface water monitoring in the Czech Republic on selected profiles. There are 20 sampling sites chosen, where the CHMI has observed following since January 1999:

a) Water: Basic physical and chemical characteristics, heavy metals, specific organic compounds, basic biological and microbiological characteristics and radioactivity were analyzed. Samples were taken monthly, except for saprobity index, where the samples were taken twice a year.

b) Suspended solids: Heavy metals, TC, TP and specific organic compounds (AOX, PAH, PCBs and chlorinated pesticides) were analyzed. Samples were taken monthly.

c) Sediments: Heavy metals, TC, TP and specific organic compounds (AOX, PAH, PCBs and chlorinated pesticides) were analyzed on 0.02mm fraction. Samples were taken twice a year.

d) Bio indicators: Lead, cadmium, mercury, arsenic, PCBs and chlorinated pesticides were analyzed. Macrozoobenthos, Dreissena polymorpha and fish were sampled. These samples were collected on 8 from 20 profiles.

Finally, 24 profiles will be added to the complex monitoring network of water quality in year 2000.
The most used system of describing ground water quality are based on the identification of the main hydrochemical types and their components, or on the comparison between ground water quality and ground water uses for the drinking or domestic purposes.

The knowledge acquired in ground water monitoring activity shows that there were a few cases when the ground water quality parameters were in conformity with the established standards norms for drinking water; thus, one may be led to the wrong conclusions that the water source is polluted.

There has been conceived a multi-criteria system to get a complex description of ground water quality. This system starts from the main field of using these resources and correlates these with the identification of some risk potential thresholds.

By using an integrated analysis of minimum 50 general quality parameters and 30 optional parameters which characterise each ground water source, there were indentified 4 main classes of the ground waters.

This classification establishes the possible comparison between the water qualitative parameters with the possible utilization of the ground waters for drinking purposes, or animals water and industrial consumption, directly or with a preliminary treatment as well as the possibility of using the waters for certain corps irrigation. It also shows the risks potential which the direct consumer is exposed to.

The classification compares too, the qualitative parameters of the ground waters with the potential aggressive effects on metal, concrete and ferro-concrete parts of the underground constructions which are in contact with ground waters. The risk limits are established both by the number and the concentration of the determined pollutants in each type of ground water.

By using this system there can be established special survey and protection measures or priority interventions to identify polluting sources and to remove them, as well as rehabilitation measures to protect and ensure sustainable use of the hydrogeological resources.
ROLE OF INVENTORY FOR WATER MONITORING PROGRAMS IN THE NETHERLANDS

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In order to control the water quality and quantity of the Dutch surface waters, a national water monitoring program is run in the Netherlands. A determined number of physical, chemical and biological parameters is frequently measured in the national waters. In addition local water quality authorities control the regional surface waters with a more or less extended number of parameters. To avoid that monitoring programs suffer from data rich information poor syndrome inventories will be helpful. So, developments in for instance the agricultural sector, the pharmaceutical industries or climate changes, etc. may cause new, different water quality problems in the future. Consequently the involvement of new relevant parameters for monitoring programs has to be a continuous process. Signs for these developments may come from problems of the water quality authorities itself, from international (IRC, OSPAR) and national working groups, symposia, legislation, management plans.

In the Netherlands part of the monitoring capacity is used for Inventory. In these inventory projects the signalized developments are investigated and evaluated for their (possible) influence on the water quality and/or quantity. In such projects presence of the pollution in different water compartments, effects of compounds, development of analytical method(s), the sources and amounts of pollutants if relevant will be investigated. Examples of such projects are LOES (estrogen compounds), new pesticides, Crypto/Giardia).
The WRK water works supply pretreated surface water for further processing by two major drinking water utilities and for industry. There are two abstraction plants, one at the Rhine river near Nieuwegein, in the center of the country, and one at the IJssel lake near Andijk, in the North-West.

Especially at the Rhine river intake site the source water quality may fluctuate due to sudden quality deteriorations. This is why WRK operates an extensive Early Warning System (EWS), including chemical (e.g. GCMS, LCMS) as well as biological (e.g. daphnids, fish, mussels) systems. In total, well over 2 million people in the western part of The Netherlands depend on the WRK water as their drinking water source.

In order to be effective, the EWS has to be operated continuously or on a high frequency basis. Since unattended / automated operation is available only for a few, relatively simple quality variables and for certain bio-alarming systems, this implies working hours during weekends and holidays. Apart from a financial burden for the organization, due to overtime compensation, this implies an unwanted situation for staff.

The existing EWS was, therefore, optimized by taking into account a combination of hydrological considerations (actual river discharge / flow and the position of locks), a risk assessment (the origin and characteristics of potential threats), treatment considerations (the nature of the pollutants and the effectiveness of the treatment) and the available instrumentation. In general, most volatile and relatively non-polar pollutants tend to be of industrial origin and, since there are no major industrial inputs on the river trajectory between the Dutch-German border and the intake, the analysis of an upstream sample provides an impression of the water quality to be expected at the intake at a future time. In contrast, most pesticides are polar in nature and inputs may be expected during certain parts of the year on this river trajectory. Such inputs, however, are usually fairly gradual (non-point source), and concentrations below detection limits rarely exceed the threshold-level within a 2-day period.

The optimized system provides for an adequate intake protection while creating a two-day period during any given 7-day period, in which no on-site staff is needed, by the analysis of additional upstream samples at times of high water discharge. Low water discharge situations allow for an even larger interval since locks are closed and water flow is considerably reduced. Estimated savings amount to approximately $20,000 annually, or 10% of the corresponding salary budget (five trained and qualified EWS specialists). Additional costs are made, of course, by the additional sampling, transportation and analyses, reducing the overall net savings. Most importantly, however, an adequate intake protection can be guaranteed without the need of weekend labor unless, of course, the upstream sample indicates unwanted pollutants are to be expected at the intake. This considerably reduces the burden for the EWS specialists.
INTEGRATED MONITORING ON THE LAKE PEIPSI

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The Lake Peipsi/Chudskoe is one of the major lakes of the Baltic Sea water basin. Lake Peipsi/Chudskoe (3555 km²) is the fourth largest lake in Europe. It is situated on the Estonian-Russian border and is therefore the biggest transboundary lake in Europe. The lake consists of three unequal parts: the biggest northern L. Peipsi s.s. (2,613 km², maximum depth 12,9 m at water level 30.01 m above sea level, water capacity 21,79 km³); the southern L. Pihkva / Chudskoe (709 km², 5,3 m, 2,68 km³), and the narrow strait-like L. Larnmijärve / Teploe connecting them (236 km², 15,3 m, 0,60 km³). The watershed (including the lake itself) covers 47.800 km² of the territories of Russia, Estonia and Latvia.

The Lake Peipsi belongs to the watershed of Narva River, a 77 km long water- course, which connects the Lake Peipsi with the Gulf of Finland of the Baltic Sea. The Narva River annual water discharge into the Gulf of Finland is 12,6 km³ (approximately 50% of the average volume of the Lake Peipsi). About 240 rivers and streams flow into the Lake Peipsi. The major rivers are Velikaya (in Russian Federation) and Emajõgi (Estonia) with catchment areas 25,200 sq. kilometres and 9,745 sq. km respectively. The residential time of water is about two years in the whole lake.

Regular water chemistry monitoring on the Lake Peipsi started in 1950. Hydrobiological investigations had been carried out since 1962. The monitoring was a complex and integrated at the very beginning because it was a part of the surface monitoring program. So the assessment of the lake water analyses results has been done together with those from rivers in the catchment area of the lake.

In addition to water chemistry analyses the hydrobiological investigations in the lake and in river had been carried out. From the hydrobiological investigations the following should be mentioned.

- Phytoplankton (species composition and biomass);
- Chlorophyll a, b and c;
- Zooplankton (main species, primary production);
- Bacterioplankton (total count of bacteria, number of saprophytic bacteria, total Coliforms and Enterococcia);
- Macrozoobenthos;
- Macrophytes;
- Fishes and fisheries management (34 various species, productivity 25-34 kg ha⁻¹)

Pollution load from the catchment was extremely high in 1980-s, which causes eutrophication of the lake. The first complex estimation of the nutrient load to the Lake Peipsi was carried out in 1989-90s. The results of those investigations had published by Institute of Zoology and Botany of Estonian Academy of Sciences in 1990 and 1991(Loigu et. al. 1991) Those publications are in Estonian, with summaries in English. According to those investigations 74% of the total nitrogen load and 46% of the total phosphorus load was of agricultural origin. The very essential pollution sources were two big towns -Pskov in Russia and Tartu in Estonia.

The second complex estimation of the nutrient load to the Lake Peipsi was carried out in cooperation of Estonian-, Russian and Swedish scientists and specialists on water ecosystems monitoring. The results of those investigations and evaluations show that the lake received as the average 19 000 tonnes of nitrogen, 580 tonnes of phosphorus annually during the time period 1995-1998. Riverine transport is the most important pathway for input of nutrients to the Lake Peipsi. Examination of the spatial variation of nutrient loads showed that the Velikaya River alone accounted for approximately 55% of the total riverine load.

According to P, N and chlorophyll a, the trophic state of three parts of Lake Peipsi is different: Lake Peipsi s.s. is an eutrophic lake (Chl a mean values 14,7 mg m⁻³), Lake Pihkva / Pskovskoje ozero is considered to be hypertrophic (mean Chl a 47,8 mg m⁻³). The long-term average primary production is 0,8 g. C m⁻² d⁻¹. Diatoms and blue-green algae prevail in phytoplankton biomass. The blue-greens Gloeotrichia echinulata and Aphanizomenon flos-aquae dominate in
summer causing the water-blooms. Zooplankton is remarkably rich in species. The average biomass in the vegetative period is 2-3 g. m$^{-3}$ and production 22 g. C m$^{-2}$. The role of rotifers in production is 53% followed by that of cladocerans (30%), copepods (16%) and *Dreissena polymorpha* larvae (1%). The average abundance of macrozoobenthos (without big molluscs) is 2617 ind. m$^{-2}$, and their biomass 12,34 g. m$^{-2}$ are considered to be the highest among the large lakes of North Europe. Macroflora occupies a small percentage of the total lake area but is rich in species. Taxa forming communities are *Potamogeton perfoliatus*, *Phragmites australis*, *Schoenoplectus lacustris*, *Potamogeton lucens*, *Eleocharis palustris*, and *Polygonum amphibium* (Noges. et. al. 1996). The content of total nitrogen, nitrite (NO$_2^-$), ammonia (NH$_4^+$), total phosphorus, silicon (Si), orthophosphate (PO$_4^{3-}$), chlorophyll a, dichromate oxidizability (COD$_{CR}$) and permanganate oxidizability (COD$_{Mn}$) decrease from south to north, while water transparency, alkalinity (HCO$_3^-$), sulphate (SO$_4^{2-}$), chloride (Cl$^-$), calcium (Ca$^{2+}$), magnesium (Mg$^{2+}$), have the opposite trend. The first trend is caused by the impact of the pollution loads from big cities, Pskov (mainly to Lake Pihkva / Pskovskoje) and Tartu (to southern part of Lake Peipsi s.s. and to the Lämmijärvi / Tjoploje).

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INTEGRATION AND REDESIGN OF THREE EXISTING REGIONAL ROUTINE WATER QUALITY MONITORING NETWORKS (ABSTRACT)

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Recently a routine water quality monitoring network has been designed covering almost the entire province of Overijssel and the southern part of the province of Drenthe. Characteristics of this area include: a great variety of water, land use and soil types, various water functions, (geo)hydrological conditions and spatial scales, and - as a consequence - a wide range of ecosystems and ecological values.

The goal of the monitoring network design was to integrate three existing routine networks into one ‘optimal’ routine network, meaning full compliance with actual information needs of all parties concerned - and integrating physico-chemistry, ecology and surface water hydrology (the latter with regard to the calculation of loads) at the same time.

During the designing process the participants were faced with major constitutional and geographical changes which are, at present, widely seen in the Netherlands: (a) regional water boards shifting from either water quality or water quantity management towards so-called all-in water boards dealing with integrated water management, and (b) changing boundaries complying with existing catchment areas in order to facilitate (river)basin management. This means that in the aforementioned area now four water boards are managing water quality at the regional level. Besides these actors the provincial government also participated in the project. As a consequence monitoring objectives had to be tuned with the various policies, priorities and information needs of all parties concerned.

Derived from present (regional) water policies - which are of course related to functions, issues and pressures - the most relevant monitoring objectives are: (1) the assessment of the actual status of individual water bodies by qualitative descriptions on the one hand and by testing for compliance with standards on the other hand, (2) the detection of long term trends, and (3) the calculation of loads.

In a combined effort basic principles have been defined and sets of criteria for monitoring network design were agreed upon, with regard to sampling frequencies and selection of sampling points and variables/indicators.

Catchment areas (at the basin and sub-basin level) were defined, which were leading in the network design. Furthermore an analysis on geographical data was performed to identify most relevant combinations of soil and land use types.

By doing so, sampling points could be spatially distributed in compliance with ‘RWSR’ guidelines, producing information about a sub-basin as a whole or on a specific area which is being homogeneously affected by certain point and/or diffuse sources.

By means of additional statistical analysis optimal sampling frequencies were established with regard to trend detection and testing for compliance with standards.

Over all the project has been a process rather than a statistical exercise.
AN ANALOG MODEL OF THE NATURAL ENVIRONMENT AS A COMPONENT OF THE STATE BORDER ZONE MONITING SYSTEM

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As a rule, large failures on objects storing hazardous (toxic) waste are preceded by insignificant outflows, small failures etc. For timely warning about possible technogenic treat to water bodies in one country due to accidents on objects of a neighbouring country can be provided by the method of inverse problem solution, with mathematical analogue modelling of underground filtration and pollutants spreading in a surface water body.

INTRODUCTION

At the Russia territory, in the Nezhigol river valley, the Severski Donets left bank tributary (the inflow point is located 12 kms upwards from the border with Ukraine), is situated an industrial area, where large industries are concentrated, mainly chemical and machine-building profile. The area also contains some other smaller enterprises. The considered territory is situated within the right slope of the Donetsk - Dnipro depression. At that chalky aquifer alimentation region is situated in Russia, while the groundwater discharge is in Ukraine. The basic sources of pollutants contaminating underground waters are slurry ponds constructed without unfiltration screens. In such ponds only one chemical factory discharged over sodium sulfate over 400 000 t calcium salts lowmolecular acids, over 7 000 t of manganese, and other substances.

The results of in situ hydrochemical investigations elucidated that for almost forty years period in this industrial region an area of groundwater pollution has been formed, draining to the Nezhigol river. The site of pollution area is 8-10 km2, 50 m deep. For prevention of the Nezhigol river pollution with the groundwater flow a set of preventive measures aimed at localisation and quality restoration has been elaborated.

The measures include actions aimed at changes in production methods and at changes in separate elements (e.g. placing all communication lines above ground, insulating underground parts of tanks etc). The protective engineering measures include a ring system of drainage wells situated along contour of the polluted area, application of wetlands, lagoon screening to prevent pollutants infiltration to groundwater. Implementation of such measures would have drastically reduce pollutants inflow to the Nezhigol river (Russia), and later to the Severski Donets river and Pechenzhske reservoir (Ukraine).

But after collapse of the Soviet Union the source of the dangerous pollution of aquifer and interconnected surface water – including huge Pechenezhskoe drinking water reservoir – are at different sides of the state border. So, in spite of the border "transparency", going along the upper part of the reservoir, at present it is impossible to monitor the state of the pollution focus, as well as to oversee implementation of the elaborated set of the environment protection measures.

As reservoir is of primary importance for the Kharkiv city (the second by size city of Ukraine), water supply the Ukrainian environment inspections are regularly controlling the water quality across the reservoir area. Multiannual results of investigations when compared indicate that there is no reductions in pollutants inflow reservoir, though now they come across the border.

RESEARCH DESIGN AND METHODS

In the process of the monitoring network improvement, on the base of use Software FEFLOW (WASY, Germany), an analogue model of the region has been created. The model allows to prognosticate migration of the aquifer pollution focus (by main ingredients) towards the Nezhigol river. At that the, data on hydrogeological and technogenic conditions in the region were available. But if such data about adjacent area (situated in Russia) are absent, in will be possible to extrapolate and especially so if it is taken into account that any cartographic material, as a rule, includes data about adjacent territory.
On the basis of results on pollutant inflow to the Nezhigol River computations were performed on pollutants transfer with river water to the Pechenenzhske reservoir (with allowance for biochemical processes of selfpurification). Solution of an inverse problem demonstrated that the Nezhigol river receives groundwater with the following level of pollution: sulphates -1000-1200 mg/l, chlorides -600-1000 mg/l, oil products -10-15 mg/l, COD - 300-500 mg/l, surfactants 0.2-0.5 mg/l. This clearly demonstrated that the elaborated environmental measures were not implemented – this conclusion was confirmed just by telephone call. And prognostic computations demonstrated that pollution cores at the industrial area in the last 5-6 years have shifted for 250-300 m, i.e. reached the riverbank. After that, computations were performed on pollutants spreading with river water to the Pechenezhske reservoir. Results of these computations allowed augmenting monitoring data obtained only at the Ukrainian territory, and substantiating claims to the adjacent enterprises.

CONCLUSION

For determination of the groundwater and interconnected surface waters pollution intensity, inflowing from abroad but polluting important water body in Ukraine, a method of inverse problem solution can be applied with use of the mathematical analogue modelling of the subterranean filtration and of models of pollutants spreading in the surface water body. Obtained results would allow in due time to warn about emergency situations, and also damages to the bordering side in case of harmful environmental effects. Necessity to implement the offered approach has been demonstrated by large-scale accidents in Romania (in year 2000) at gold mines, which caused immense damage to water bodies in Hungary and Ukraine.
IMPLEMENTATION OF MONITORING GUIDELINES OF EEC/UN IN THE BASIN OF THE RIVER BUG AS A FORM OF PRACTICAL CO-OPERATION ON TRANSBOUNDARY WATERS BETWEEN POLAND, UKRAINE AND BELARUS – THE GAINED EXPERIENCE

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The transboundary river Bug (772 km) located in the catchment area of the Baltic Sea, whose sources are in Ukraine, forms the border of Poland and Ukraine – 185 km and further the border between Poland and Belarus – 178 km. The total area of the basin of the river Bug is 39,420 km², 50% of which is located in Poland, 20% in Ukraine and 30% in Belarus. The river Bug flows into the Zegrzynskie Lake which is the main source of drinking water for Warsaw, the capital of Poland.

The agreement signed in 1964 between Poland and the USSR on co-operation on transboundary waters was the basis for the co-operation between Poland and the USSR until its break-up. However, the results of this co-operation could not be used for any practical purposes. It applies mostly to water protection and quality assessment.

When the Guidelines were drawn up and the river Bug was selected for implementation of the Guidelines as a pilot programme, a chance appeared to initiate transboundary co-operation between Poland and the countries which came into being after the break-up of the USSR: Ukraine and Belarus. As there was no adequate agreement, in 1997 a trilateral Agreement between the Ministers of Environmental Protection of these countries was signed, which made possible the realisation of the project of pilot implementation of monitoring guidelines in the basin of the river Bug.

Drawing up of the project consists of four stages: Preliminary Stage, Preparatory Stage, Implementation Stage and Assessment Stage. The pace of works on the project is varied in particular countries and it is conditioned above all by different accessibility to financial aid which varies in time in Ukraine and Belarus and lack of financial aid for Poland.

The Preliminary Stage was ended in 1998 at the same time by the three countries. Ukraine, which was assigned financial means from TACIS, will finish the Implementation Stage this year. Poland finances the project from the means of the National Fund for Environmental Protection and Water Management and it is finishing the Preparatory Stage at the moment and Belarus is in the middle of tendering procedures for carrying out the project using the means from TACIS.

Uneven financing has a negative impact on the realisation of the project, complicates its organisation and increases the costs. It contradicts the assumption that the project should be carried out simultaneously by all countries in the river basin.

However, despite of these difficulties, the co-operation is taking a more concrete shape. It applies mainly to Poland and Ukraine. The Inventory Reports and Legislation Report have been drawn up and accepted, joint water sampling for quality analysis takes place and a study tour in the Polish and Ukrainian part of the basin of the river Bug has been organised. The human mentality which is still burdened after the period of socialistic regime and the influence of political fluctuations which is still too significant are the obstacles for organisation and realisation of the project.
TOXICITY-BASED ASSESSMENT OF WATER QUALITY

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Interspecies variations of acute toxicity observed in organic surface water concentrates with unknown constituents are used to assess potential effects in the naturally exposed aquatic community. The proposed method includes the following steps:
1) Application of an XAD-sorption-based concentration procedure to large volume samples of surface water
2) Laboratory toxicity testing of the concentrates with a number of micro-volume-tests on different species
3) Acute to chronic effects extrapolation
4) Calculation of the toxic potency (pT) of the original water sample, using the sensitivity distribution of effective concentration factors observed in laboratory testing.

The pT-value is expressed as the proportion of a generic species assembly that is potentially under toxic stress in the field (species exposed above their chronic combined NOEC). Using redundancy analysis, the set of applied toxicity tests is demonstrated to reveal different aspects of toxic pollution. With pT-values ranging from 0 to about 10 percent of species potentially affected, the results suggest that some of the aquatic ecosystems in The Netherlands are severely stressed by exposure to a variety of organic toxicants. Comparison to measured toxicant concentrations mainly attributes the observed effects to a variety of pesticides and PAHs.