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AN OUTLINE OF A MODEL-BASED EXPERT SYSTEM TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING CONTAMINATED ACQUATIC ECOSYSTEMS: THE PROJECT "MOIRA"

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RIASSUNTO

Il presente rapporto descrive i principi del programma di ricerca "A MODEL-BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING RADIONUCLIDE CONTAMINATED AQUATIC ECOSYSTEMS AND DRAINAGE AREAS" (MOIRA) finanziato dalla CE (Contratto N° FI4P-CT96-0036). Gli interventi sull'ambiente volti a ridurre i livelli di esposizione della popolazione alle radiazioni dovuti alla contaminazione radioattiva di ecosistemi acquatici continentali possono avere un impatto non trascurabile sulle condizioni socio-economiche e sull'ambiente stesso. Gli amministratori incaricati di valutare l'opportunità di attuare contromisure per la riduzione dei livelli di radioattività devono tenere conto di tali impatti. Scopo del progetto MOIRA é di produrre un sistema esperto che, mediante modelli validati, consenta di valutare l'evoluzione della contaminazione ambientale, i costi socio-economici e l'impatto ecologico delle diverse contromisure applicabili. Il sistema esperto "MOIRA" rappresenterà un valido supporto per la scelta delle azioni ottimali per la riduzione dei livelli di esposizione nel rispetto delle condizioni socio-economiche e dell'ambiente. Esso fornirà a personale non esperto nell'ambito della modellistica ambientale uno strumento efficace e facile da utilizzare.

SUMMARY

The present report describes the fundamental principles of the research programme "A MODEL-BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING RADIONUCLIDE CONTAMINATED AQUATIC ECOSYSTEMS AND DRAINAGE AREAS" (MOIRA) financed by the EC (Contract N° FI4P-CT96-0036). The interventions to restore radionuclide contaminated aquatic systems may result in detrimental ecological, social and economical effects. Decision makers must carefully evaluate these impacts. The main aim of the MOIRA project is the development of an expert system based on validated models predicting the evolution of the radioactive contamination of fresh water systems following countermeasure applications and their relevant ecological, social and economical impacts. The expert system will help decision makers, that are not necessarily gifted with experience in environmental modelling, to identify optimal remedial strategies for restoring contaminated fresh water systems.

SAMMANFATTNING

Denna rapport beskriver de grundläggande principerna bakom forskningsprogrammet "A MODEL-BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING RADIONUCLIDE CONTAMINATED AQUATIC ECOSYSTEMS AND DRAINAGE AREAS" (MOIRA) finansierat av EU-anslag (Contract N° FI4P-CT96-0036). De olika åtgärderna för att åtgärda vattensystem kontaminerade med olika radionuklider kan föra med sig såväl positiva som negativa ekologiska, sociala och ekonomiska effekter. Beslutsfattare måste därför noga kunna utvärdera dessa effekter. Huvudsyftet med MOIRA-projektet är att utveckla ett expertsystem som inkluderar validerade modeller med vars hjälp man kan prediktera utvecklingen av en kontaminering med radioaktiva ämnen i sötvattenekosystem och olika åtgärders ekologiska, sociala och ekonomiska effekter. Expertsystemet skall kunna hjälpa beslutsfattare, som saknar erfarenhet eller specialkompetens inom ekosystemmodellering, att identifiera optimala åtgärdsstrategier för att restaurera kontaminerade vattensystem.

SAMMENDRAG

Denne rapporten beskriver de grunnleggende prinsippene bak forskningsprogrammet "A MODEL-BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING RADIONUCLIDE CONTAMINATED AQUATIC ECOSYSTEMS AND DRAINAGE AREAS" (MOIRA) finansiert av EU (Kontrakt nr. FI4P-CT96-0036). Restaurering av akvatiske systemer kontaminert med radionuklider kan medføre positive såvel som negative økologiske, sosiale og økonomiske effekter. Forvaltningsmyndigheter må derfor vurdere disse forholdene nøye. Hovedhensikten med MOIRA-prosjektet er å utarbeide et kunnskapsbasert ekspertsystem som inkluderer valideringsmodeller som kan predikere utviklingen av kontaminering med radioaktive elementer i ferskvannsøkosystemer og angi de økologiske, sosiale og økonomiske effekter ved ulike tiltak. Ekspertsystemet skal kunne være til hjelp for forvaltningen som eventuelt mangler erfaring eller spesialkompetanse på økosystemmodellering, slik at optimale strategier for restaurering av kontaminerte ferskvannsøkosystemer kan identifiseres.

RESUMEN

El presente informe describe los principios fundamentales del programa de investigación "A MODEL-BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING RADIONUCLIDE CONTAMINATED AQUATIC ECOSYSTEMS AND DRAINAGE AREAS" (MOIRA), financiado por la CE (Contrato N. FI4P-CT96-0036). Las intervenciones para la restauración de sistemas acuáticos contaminados con radionucleidos pueden causar impactos ecológicos, sociales y económicos. Las autoridades han de evaluar dichos impactos cuidadosamente antes de tomar decisiones sobre las intervenciones. El principal objetivo del proyecto MOIRA es el desarrollo de un sistema experto basado en modelos validados para la predicción de la evolución de la contaminación radiactiva en sistemas de agua dulce así como de los impactos ecológicos, sociales y económicos producidos tras la aplicación de diferentes contramedidas alternativas. El sistema experto MOIRA ha de servir de ayuda para quienes, no poseyendo necesariamente experiencia en el modelado del medio ambiente acuático,hayan de identificar las contramedidas óptimas para la restauración ambiental de dichos sistemas de agua dulce tras su contaminación accidental con sustancias radiactivas.

SAMENVATTING

Dit rapport beschrijft de fundamentele principes van het onderzoekprogramma "A MODEL BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING RADIONUCLIDE CONTAMINATED AQUATIC ECOSYSTEMS AND DRAINAGE AREAS (MOIRA)", gefinancierd door de EC (contract nr. F14P-CT96-0036). Het doorvoeren van maatregelen om met radionucliden besmette aquatische systemen te reinigen zou in negatieve ecologische, sociale, en economische effecten kunnen resulteren. Besluitvormers moeten deze consequenties dientengevolge zorgvuldig evalueren. Het hoofddoel van het MOIRA project is het ontwikkelen van een expertsysteem gebaseerd op gevalideerde modellen die het verloop van de radioactiviteit na genomen tegenmaatregelen kunnen voorspellen als ook de ecologische, sociale, en economische effecten ervan. Het expertsysteem zal de besluitvormers, die niet altijd noodzakelijkerwijs ervaren zijn in milieumodellering, als hulp kunnen dienen om de optimale stategie voor de sanering van besmette zoetwater systemen te bepalen.

PREFACE

MOIRA (A **MO**DEL-BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO **I**DENTIFY OPTIMAL REMEDIAL STRATEGIES FOR **R**ESTORING RADIONUCLIDE CONTAMINATED **A**QUATIC ECOSYSTEMS AND DRAINAGE AREAS) is a project financed by the European Commission (Contract N° FI4P-CT96-0036) in the frame of the **Nuclear Fission Safety** research and training programme (framework programme 1994-1998). The aim of the project is to construct a model-based computerised system using cost-benefit analysis for the identification of optimal remedial strategies to restore radionuclide contaminated fresh water environment. The following institutions are participating in the MOIRA project: ENEA (Co-ordinator, Italy), KEMA (The Netherlands), the University of Oslo (Norway), the UPM (Universidad Politécnica de Madrid, Spain), Studsvik Eco & Safety AB and the University of Uppsala (Sweden). The present report was prepared during a meeting held in Nyköping (Sweden), 29-31 May 1996.

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INTRODUCTION

The accidental introduction of radioactive substances into the environment has caused high and persistent levels of environmental contamination. Aquatic ecosystems are of utmost importance in considering dose to man. Drinking and irrigation water are the obvious examples, but also shellfish and fish products may be of even greater importance, especially in areas where few alternatives are available. Aquatic systems are also a major pathway for the transport and dispersal of radionuclides from terrestrial fallout, as seen in the aftermath of the Chernobyl accident. Purely local contamination can, through the medium of aquatic systems, lead to widespread consequences.

The necessity of reducing doses to man has lead to restrictions in the economical exploitation of the contaminated areas. A variety of restoration strategies have been proposed to reduce these restrictions and thus the economical costs:

- reduction of radionuclide remobilisation from sediments (e.g., liming);
- removal of radionuclides from aquatic systems (e.g., modification of water discharge to influence contaminant turnover and retention, dredging and disposal of contaminated sediments);
- reduction of uptake by aquatic organisms (e.g., potash treatment of aquatic ecosystems contaminated with radiocesium);
- reduction of radionuclide transfer from contaminated drainage areas to water bodies (e.g., dam building to avoid contact with contaminated floodplain areas, removal and disposal of contaminated soil from flood plains).

On the other hand, some particular restoration strategies, whereas not directly applied to fresh water systems, may have a major impact on the aquatic environment. Among them, the forced decontamination of urban environment is of paramount importance, causing the movement of deposited radionuclides to fresh waters (for instance through sewage systems).

As a result of the Chernobyl accident and of the contamination of other areas in the former Soviet Union, a whole new field is developing in the application of countermeasures. Large scale engineering works are being tried and tested. However, despite their obvious benefits, countermeasures may result in detrimental effects that must be carefully evaluated by cost-benefit analysis.

Lakes, rivers, and coastal areas are extremely complex webs of physical, chemical, and biological interactions (fig. 1). It is difficult to describe their characteristics, and to predict the turnover of toxic substances within these aquatic areas is even harder. To develop scientifically warranted programs of conservation, management and remediation are an even greater challenge. Every aquatic ecosystem is a unique natural feature, and yet it is impossible to study each system in the detail necessary for a case-by-case assessment of radioecological threats, and proposals for realistic remedial measures. In this situation, quantitative models are essential to predict the behaviour of contaminants in the aquatic ecosystem, to guide assessment, and to direct intervention. Such models must be validated, the uncertainty limits of the model predictions must be given and the domain of the model applicability stated, so that the model is not used for other purposes and predictions than intended.

The MOIRA project seeks to develop management models for assessing the effects of remedial measures on lakes, rivers and coastal ecosystems. The basic aim of the project is to develop a software system based on validated, predictive user-friendly models driven by readily available parameters. Such a system may be used by decision-makers to choose optimal strategies of restoration for different radioactive contamination scenarios of aquatic systems and drainage areas. It should enable quick, accurate predictions shortly after a new accident (given a certain

fallout of radionuclide), should help to provide feasible proposals for remedial measures and should give realistic cost/benefit analyses. Alternative remedial strategies will of course be assessed both in terms of their success in reducing dose to man and in terms of their socio-economic benefit, in addition to their impact on the environment.
One of the goals of the project MOIRA is to define general criteria to identify "performance indices" that will offer the opportunity of ranking remedial strategies according to their effectiveness and costs when applied to specific types of environmental contamination, reducing the element of "art and chance" in environmental decision-making.

THE IMPACT OF COUNTERMEASURES ON KEY SPECIES AND TARGET ECOSYSTEMS

• Remedial strategies

Different remedial measures have been tested to try to decrease the concentrations of radiocesium in lake fish (see Håkanson and Andersson 1992 for further details), namely:
- To reduce the transport of cesium from land to water; wetland liming and full-scale liming have been tested to change soil chemical processes in order to reduce cesium mobility.
- To increase the proportion of cesium sedimented on the lake bottom, thereby preventing or delaying its biouptake, lake liming was carried out. The hypothesis is that the flocculation tendency of cesium-carrying particles is increased by increasing the pH, alkalinity and hardness of the lake water.
- By blocking and/or substitution to reduce the proportion of cesium taken up by fish. For this reason, potash treatment has been carried out to increase the potassium concentration of the lake waters. Potassium and cesium are taken up in fish in a similar manner (Black 1957; Fleishman 1963; Carlsson 1978). Other ions may also participate in different blocking processes (e.g. Ca, Mg and Na), which implies that the different liming measures, which give a general increase in the ionic strength of the water may also give a positive effect in this way.
- To reduce the bioavailable amount of cesium in the system. The intention is both to remove cesium already taken up in the nutrient web by reducing the fish stock (intensive fishing) and also to influence the predation pressure so that the nutrient web is modified to contain relatively more plankton, which could result in lower concentrations of ^{137}Cs in biota.
- By means of "biological dilution", decreasing the concentration of cesium in each individual fish. For this purpose, the water systems can be treated with different types of fertiliser (discharges from aquaculture, commercial fertiliser and P-enriched lime). The intention is to increase the biomass and thereby disperse the given amount of cesium. The method is based on theories involving biological buffering (Jansson et al., 1981).
The results presented by Håkanson and Andersson (1992) concerning the effectiveness of the remedial measures in Swedish lakes may be summarized by the following points.

* Potash treatments, on average, gave a two-fold increase in lake K-concentrations compared to the natural concentrations (from about 10 to about 20 μeg/l). In lakes with very short water turnover times, it was, as expected, most difficult to obtain increases in long-term mean K-values. Where the long-term mean K-concentrations increased by more than 5 μeq/l, concentration up to 1989 in perch fry had a larger decrease than in other lakes. A reduction of 5-10 % per year of ^{137}Cs in perch fry, which is the maximum reduction of a well-conducted potash treatment, may be important in a long-term perspective for perch but also for other fish with similar feeding behaviour.
* There was a weak trend for the Cs-concentrations in pike, to be slightly lower, corresponding to a decreased uptake of about 5 % per year, in lakes where the conductivity of the lake water was increased > 1.5 mS/m by lake liming and wet land liming. There were no significant differences between different varieties of lime (like primary rock lime, sedimentary lime or "mixed" lime).
* The differences between the lakes concerning the continued magnitude of the change in Cs-concentration in perch fry could be linked to foremost factors controlling the internal loading and the secondary input of ^{137}Cs from the catchment. Shallow lakes generally have a slower reduction in the Cs-concentrations in fish, probably due to a greater resuspension and a higher internal load.
* The time interval between the remedies adopted and the latest fish analyses (on average about 2 years) was not sufficient to statistically establish the small effects of the remediation. Longer time data series are required.

• Ecosystem characterization and categorization

European aquatic systems differ widely in their characteristics, ranging from the Arctic to the Mediterranean. However, it should be possible to categorize such systems, for example on the basis of productivity. A simple framework, based on lake productivity and characteristic fish species/communities can be developed for lakes, rivers and coastal areas, using knowledge of the VAMP lakes as a framework (IAEA, 1996).

Using a template of different ecosystems based largely on productivity and characteristic communities, the concept of ecosystem sensitivity can be employed. Certain types or categories of ecosystem are sensitive both to fallout and remedial measures, while other are more robust.

Lake systems can be classified according to trophic level; oligotrophic, mesotrophic, eutrophic, etc, all of which have characteristic chemical and biological properties, including fish communities (Table 1). These are of course not absolute categories, but nevertheless provide a very useful framework for predictive modelling where it is necessary to encompass a wide range of ecosystem types, as is the intention in MOIRA.

Coastal areas can be classified in a similar way (Table 2, Håkanson, 1991a) and appropriate fish communities identified (Mann, 1982). Although coastal areas can be defined by many interrelated physical, chemical and biological factors, certain parameters cluster, such as salinity and water exchange. Also certain effect parameters such as Secchi disc transparency can be related both to nitrogen dose and to ecosystem sensitivity (Fig. 2, Wallin and Håkanson ,1991). This makes it feasible to develop relatively simple empirical models. Water turnover time, as in freshwater lakes, is also a critical parameter in coastal ecosytems. It can also be predicted by simple empirical submodels (Fig. 3, Persson et al., 1994). In addition, land use in adjacent terrestrial systems draining to coastal areas will be important, determining nutrient and freshwater inputs.

All coastal areas do not have the same morphometry, although there are typically shallow and gradually shelving. However, certain coastal areas, such as the fjord areas of Norway, Sweden and Scotland represent a rather different ecosystem and must be approached in a different way. It may be possible to adapt a similar approach to that of lake stratification and exchange between surface waters and deeper layers.

In **rivers**, flow/discharge is the major factor which drives most other variables (Allan, 1995). In the same way as for lakes (e.g. VAMP seasonality submodel, Håkanson et al., 1996) it should be possible to develop submodels for discharge in relation to slope, precipitation, catchment size, etc. Here again drainage area charcteristics are important. Table 3 gives some of the important environmental variables used in the analysis and classification of running water ecosystems.

The conceptual ecological framework for running waters has seen much advance during the last 20 years after the publication of the River Continuum Concept (RCC) (Vannote et al., 1980). Although not entirely accepted worldwide in its original form, it has contributed significantly to our understanding of ecosystem functioning. It has also been modified for use in regulated rivers (Ward and Stanford 1983) and considered in relation to Nordic rivers (Petersen et al. 1995). It is based on stream size, energy inputs and functional groups, which differ along the river continuum from the headwaters to the sea (Fig. 4). The effects of dams and regulation (summarized in Brittain and L' Abée-Lund 1995) is particularly important both in terms of the consequences of fallout and in applying remedial actions. The RCC superceeded a number of earlier classifications, largely developed in central Europe. These include the concepts of rhithron and potamon (Illies and Botosaneanu 1963), each with their characteristic trophic relationships (Table 4, Figs 5, 6). This has been further divided accorded to typical fish species/communities by various authors and related among other things to slope (Table 5, see also Hynes 1970). More recently it has been shown that species richness in fish communities is mainly a function of catchment area and to a lesser extent productivity, while historical factors such as glaciations are of minor importance, at least on a global scale (Oberdorff et al. 1995). Within the scope of MOIRA and remediation it should be possibile to combine these concepts into a single ecological framework for European rivers.

There are many links and indeed similarities between rivers, lakes and coastal areas, and although it is necessary to treat then separately in the development of models for management, the possibility of interfacing and applying submodels developed in one area in another should be continuously assessed.

• Criteria for the evaluation of the ecological impact

Remedial measures, by their nature represent a greater or lesser impact on ecosystem function. Although such impacts are potentially more important in natural and semi-natural ecosystems than in urban and other man-made ecosystems, there is the need to develop the concept of ecological cost in all remedial actions. In some cases the remedial action may give rise to more serious consequences than the original fallout. This has been seen for example in remedial actions for acidification, where in some cases naturally acid systems have been limed causing profound unnatural structural changes in the ecosystem. Boundary layers generated between limed and non-limed waters are another example of undesirable side effects of remediation.

Ecological cost can be defined as the impact of the countermeasure on ecosystem functioning. In addition to the obvious benefit with regard to the dose to man one can also in certain cases envisage ecosystem benefit, for example fertilization in a reservoir rendered unproductive by regulation and liming in an area affected by acid precipitation. Possibly apart from such parameters as fish yields, quantification of ecological cost is difficult and would probably involve complicated ecosystem modelling and the application of life cycle analysis. Nevertheless, the main object of ecosystem impact/ecological cost is to make management aware of the potential impacts and the dangers of a particular remedial action. This could be incorporated into a simple matrix between ecosystem type and remedial action.

Lack of remedial actions may also have profound ecosystem effects. For instance, the cessation of fishing for an economically important fish species may well affect the dynamics of the whole ecosystem. This effect was seen in many Scandinavian lakes as a result of the Chernobyl accident, whereby mean individual fish size decreased markedly as a result of population growth.

Ecosystem functioning can be defined in several ways. A detailed approach would be to consider the growth and reproduction of all individual species and their interactions with other species and with the physical and chemical environment (Fig. 7, 8). This is however rather unrealistic, both in modelling terms and in terms of our present ecosystem knowledge. There are two simpler alternatives, firstly to assess changes in biodiversity, that is the number of species and their abundance, and secondly to select key species and/or communities. The concept of indicator species has been widely employed in the assessment of human impact on ecosystems (Fig. 9). In the aquatic sciences their use has been widespread in detecting and monitoring organic pollution and acid precipitation (Fig. 10, Hellawell, 1986, Rosenberg and Resh, 1993). The assessment of the ecological impact in terms of key species and/or communities has a number of advantages. It has the advantage of being simpler, less expensive and more easily enables linkage with the target parameters for remedial actions such as lake waters and consumable fish. This approach has been used successfully in the VAMP model for lakes (IAEA, 1996) and could be extended to rivers and coastal ecosystems.

Thus for lakes, rivers and coastal areas it is necessary to select target ecosystems. That is ecosystems that are relevant and will be influenced by a particular remedial measure. For instance, there would cleaerly be no need to fertilize a eutrophic lake. Also within each ecosystem selected it is necessary to identify target parameters or indicators. For example in the VAMP model 1kg pike were selected as the target organism in a certain type of lake.

• Lake sensitivity and target variables

We mean to address the following fundamental question: Which is the most sensitive type of aquatic ecosystem for a given radionuclide fallout? It is evident that the question cannot be scientifically answered unless it is further specified. What is meant by "sensitive", how can it be defined? The aim of this section is to provide a framework for these questions, as interpreted in the MOIRA project.

A given fallout of radiocesium (e.g. after an accident) will be distributed and taken up by biota very differently in various types of lakes (see Håkanson, 1991b; IAEA, 1996, which discuss the results related to the Chernobyl accident). Thus, lakes have different "sensitivities" to radiocesium. Important environmental factors regulating the biouptake of ^{137}Cs are the lake water retention time and the K-concentration of the water. Several practically useful and ecologically relevant methods exist to remediate lakes contaminated by radiocesium, e.g. liming, potash treatment and fertilization of low-productive lakes.

The most important sensitivity factor are ions similar to Cs, like K and Na (IAEA, 1996). The more of these ions, the higher the conductivity and the lower the uptake of ^{137}Cs. This is a case of "chemical dilution". Note also that for a single pulse, like after the Chernobyl accident, the biouptake of ^{137}Cs, and the Cs-concentration in fish, is lower in lakes with fast water turnover - a case of normal "water dilution". The opposite is valid for mercury, which is supplied to the lake "continuously". In that case, the biouptake increase with increased runoff of Hg and water from the catchment, i.e. when the Hg-load from the runoff is renewed many times per unit of time (Meili 1991). An increase in total-P, i.e. in lake production, causes a "biological dilution" of ^{137}Cs in lake biota (Håkanson and Peters 1995).

From this, certain important questions may be asked:

• What can be done in practice (in a cost-efficient and realistic manner) to reduce the Cs-concentrations in water and predatory fish (or speed up the recovery) in lakes, rives and coastal areas?
• Is it possible to reduce the secondary load, i.e., the transport of radiocesium from land to water? Or the internal loading from the sediments? Or the bioavailable portion of the load to the aquatic system?
• Is it possible to test other remedies linked to the many factors regulating the differences among water systems? For example, liming to change pH, potash treatment to change the K-concentration, or fertilization (adding of phosphorus) to change bioproduction?

Primary load is considered to be the average fallout (in Bq/m^2) of radiocesium, whereas secondary load concerns the load of radioactive cesium from catchment runoff during a given space of time after the fallout event, while the internal load is the flux of radiocesium from sediments back to water via many different advective and diffusive processes.
For ^{137}Cs in aquatic ecosystems there are no generally accepted ecological/biological effect parameters. The threat does not appear to be directed against life in the aquatic ecosystems but, besides drinking water and irrigation, is mainly directed towards humans eating fish and then primarily towards the foetuses of pregnant women and small children. This means that for ^{137}Cs one generally focus the interest on fish consumed by man. The fish are caught from several places in each system. This gives a typical area value and not a site-typical value.
In the present context, target sensitivity parameters are those that both influence the dispersal and uptake of cesium in fish and those that can be modified by different practical remedial measures.

ECONOMICAL AND SOCIAL IMPACT OF COUNTERMEASURES.

In the process of identification of the optimal remedial strategies, social and economical factors can be of a major importance.
Countermeasures are an obvious source of economic costs and, in many cases, can cause more or less serious social and individual disturbance. The evaluation of such impacts is always an important part of the decision-making process.

According to the international recommendations for intervention after a radiological accident (IAEA, 1994; ICRP, 1993), the protective actions to avoid delayed health effects should be initiated when they produce more good than harm in the affected population, and should be introduced and withdrawn at levels that produce a maximum net benefit to the population. In applying these principles, the terms "good", "harm" and "benefit" should include, obviously, health and safety and the tangible costs of protective actions, as well as other non-quantifiable factors such as reassurance, stress, and other societal values that should be taken into account by the decision-maker. Normally, these non-quantifiable factors can only be considered in a qualitative manner. However, these can be used if techniques like the Multiattribute Value Analysis are used for the optimisation. Depending on the nature of the countermeasure itself, different quantitative measures of the social impact of a countermeasure on the aquatic environment can be envisaged, the economic cost being one of them. For instance, if a ban is imposed on drinking water, a good measure of the social impact would be the number of persons affected and the duration of that ban. If restrictions are implemented on the recreational use of an aquatic site, the average number of visitors can give a good measure of the social impact rather than the strict economic cost. For countermeasures affecting the normal economic activity of the area, the economic cost itself provides a good measure of the social impact. The basic data for these evaluations (population, water and fish consumption, etc) should be included in the MOIRA data base.

With regard to the economical cost, three perspectives can be addressed:

Firstly, the direct cost of a countermeasure can be a limiting factor for its applicability on a large scale. Secondly, those countermeasures which are more cost-effective will be generally preferred. Thirdly, the impact of countermeasures on the economy of the affected (and maybe also of the non-affected) areas is an important issue in which every decision-maker will be interested, in particular for countermeasures to be applied on a large scale.

There exist a number of specific factors that could help to characterise a given countermeasure. They will normally be measured per unit of the element to be treated (e.g. volume or surface of a lake, surface of a catchment area). For each particular situation, the response of each alternative countermeasure should be described in the system with regard to different performance indices. A first evaluation (filter) could be based on generic information, taken from existing literature, past experience, or from scenarios previously anlysed. Among these factors, some are amenable to be evaluated in terms of economic cost, like the following:

Manpower.
Depreciation of equipment.
Consumables (e.g. lime, potash).
Duration (and frequency needed if applicable), implying impossibility of a normal use of the element.
Cost of management and disposal of wastes.
Changes in the value of the element to be treated.
Changes in the amount and quality of production of the element.

The list above could be enlarged or reduced for specific situations. It is just a first attempt to structure the different factors in order to identify those which can have a clear economical impact. The corresponding data should be collected (or, if necessary, may be evaluated by suitable models) for every countermeasure included as an alternative in the system.

A usual method for maximising the benefits derived from an intervention is cost-benefit analysis, which is based on a conversion of all the factors to monetary terms, and then on the minimisation of the overall cost. Some of the factors are more or less straightforward to convert into monetary units, but this is not true for all. This method is usual in normal practice optimisation of radiation protection, and, for instance, reference values can be found in the literature for the unit of collective dose, for different countries and situations (ICRP, 1993). In this case, an assessment of the direct and indirect costs of every alternative countermeasure is a primary part of all the analysis, and relatively well established methods are available for countermeasures dealing with population movements or food restrictions.

For countermeasures dealing with the aquatic ecosystems, direct costs incurred during their application should be not difficult to evaluate with sufficient accuracy, based on past experience collected by members of the group or on future experimental work. The assessment of the indirect costs is perhaps more difficult. This is related to the change in the value of the affected elements or in the potential use of them, which not always has a direct monetary translation, and which can be related more to recreational uses.

In order to evaluate the effectiveness of a given countermeasure, a good measure frequently used is the cost per unit collective dose averted. It implies a careful evaluation of all the direct and indirect costs caused by a countermeasure, combined with the evaluation, using the appropriate models, of the doses which can be averted (basically, a comparison between the dose predicted without any countermeasure and that resulting from the implementation of countermeasures).

This allows the establishment of a first ranking of the alternative countermeasures, without the need to assign monetary values to the dose. The same methods for costing countermeasures should be used in this case as in the cost-benefit analysis. Normally, a microeconomic approach is the best to use in this context, since it will be always possible to discretise all the cost element affecting the application of a given countermeasure in a given site. Many of these elements should be incorporated in the data bases of the MOIRA system, both those related to the restoration techniques as well as with the site specific characteristics (basically all the affected uses of the aquatic ecosystem).

For very large accidents, the scale of countermesures can seriously affect the economy of a country and the society's welfare, maybe inducing indirect consequences to the population as a whole, which should be not neglected by the decision-maker. In this third perspective, a regional impact could be assessed based on Input-Output modelling of the economical structure of the area, with some productive branches affected by the countermeasures. Such a model has already been developed for the case of population relocation (Gallego, 1995), and it could be expanded for other kind of countermeasures.

PRINCIPLES FOR MODELLING ENVIRONMENTAL SYSTEMS

Environmental systems, such as water bodies, generally show high levels of complexity that reflect on the structure of the predictive models aimed to assess the consequences of the human interventions on the environment. A great deal of the so called "conceptual models" have been developed during the past decades to predict the behaviour of the environment and of its components. Such models, based on the detailed analysis of the involved processes and on the relationships among them, represents extremely important tools for understanding the dynamic of the systems considered.

At a first sight, it seems unquestionable that the development of reliable environmental models requires to account for the *"totality"* of the processes taking place. Unfortunately, such kinds of omniscient models need knowledge of a great deal of environmental parameter values which are, frequently, difficult to measure and evaluate. As consequence, they are often not usable in practical circumstances.

To overcome this huge difficulty, some models take into consideration a scanty number of system components and of the relevant mutual interactions, neglecting those that may be judged not significant ("model simplification"). The "model simplification" process generally is deemed one of the main causes of uncertainty in environmental models. To all appearances, a simple model is more manageable than a complex one, but on the other hand is affected by uncertainty levels that may be rather large. For instance, the complex process of the dry deposition depends on a variety of factors and environmental circumstances, such as the nature of the depositing particles, the properties of the surface on which deposition occurs, the micro meteorological conditions. The dry deposition process, in spite of its phenomenological complexity, is generally modelled by means of a single "aggregated" parameter defined as "dry deposition velocity" (proportionality factor between the rate of dry deposition onto a surface and the air concentration of a substance). This parameter, as it summarises a great deal of phenomena, shows a large uncertainty. Similar considerations may be made for the process of wet deposition that is usually modelled by the "washout rate" (Brenk and Vogt, 1981). However, as demonstrated by a variety of studies, complexity does not necessary mean reliability. Indeed, complex models, despite their seeming accuracy, may show high levels of uncertainty due to the use of a large amount of parameters affected by non-negligible uncertainties (Håkanson, 1995).

Parameter "aggregation" has two different effects on model reliability. As previously stated, "aggregation" implies neglecting the detailed structure of the analysed system and, consequently, may result in a loss of information that induces an increase in model output uncertainty. On the other hand the introduction of "aggregate" parameters may take advantage of the possibility of treating statistical ensembles having mean properties showing strong and simple correlation.

In some circumstances the various phenomena summarised by such kind of parameters and models, may show competitive effects that make the parameter values less variable than the ones relevant to the single involved phenomena. This characteristic is the essence of the so called "collective models". Such models are based on a small number of "collective parameters" that are characterised by low variability from site to site. "Collective parameters" is a peculiar sub-class of the set of the "aggregate" parameters. "Collective aggregation" takes place when competing phenomena or statistical ensemble averages for classes of similar processes may be grouped into a single quantity.

Examples of such parameters are the "effective depth" of lakes (Monte, 1995a), the migration velocity of a radionuclide from the water to the sediments and the parameters relevant to the radionuclide transfer from catchment to water bodies (Monte, 1995b; Monte 1996).
Effective depth was introduced to evaluate the initial concentration (averaged over a period of a few weeks) of radionuclide in water bodies following a deposition event. This parameter, that accounts for the interaction of radionuclide with bottom sediment, allows one to evaluate the initial concentration of the radionuclide in water simply as the ratio "deposition/effective depth". Effective depth is equal to the average depth of the water body added to the so called "incremental depth", Δ_h, which depends mainly on the k_d (the radionuclide partition coefficient sediment-water). This approach was applied successfully to a variety of lacustrine systems contaminated after the Chernobyl accident using, for ^{137}Cs, $\Delta_h \approx 6$ m.
In many lakes, the time behaviour of radionuclide in water may be evaluated using the radionuclide "velocity of deposition" that summarises the radionuclide migration phenomena from water to bottom sediment (diffusion, sedimentation, resuspension). In the volcanic lakes in central Italy, the average value for the ^{137}Cs deposition velocity is $5.x10^{-3}$ m d^{-1} with a range from $2.x10^{-3}$ to $9x10^{-3}$ m d^{-1}. Both incremental depth and migration rates are related to the characteristics of the bottom sediments and of the contaminant. The radionuclide partition coefficient sediment-water (k_d) influences the above collective parameters. A high value of k_d implies, on the one hand, a large concentration of the radionuclide in the upper part of the bottom sediment, but, on the other hand, as the effective diffusion of the radionuclide through the sediment is inversely related to k_d, high values of this parameter imply that the radionuclide remains bound in a thin upper layer of sediment. In contrast, low k_d values induce low concentrations of radionuclide in the upper sediment layer and a deep diffusion through the sediment. These two competing effects imply a low variability of the migration velocity and of the incremental depth despite the fact that the variability of the k_d may be, in principle, very large. The analysis of the migration of radioactive substances in catchment (L. Monte,1995b) leads to the identification of certain collective parameters to predict the fluxes of radionuclides from drainage areas to water bodies. Among these the most important is the effective decay time of dissolved radionuclide concentration in water running through the catchment. Evaluations of these parameters were carried out for some rivers in Europe. The results showed that, despite the complexity of the involved phenomena and the variability of the analysed catchments, the values of the effective decay constant were within a range of a factor 5 (average values $\approx 1.5x10^{-8}$ s^{-1} for ^{137}Cs and $\approx 4.9x10^{-9}$ s^{-1} for ^{90}Sr). The success of the approach previously described, suggested the extensive use of collective models to predict, for practical applications, the radionuclide migration through catchments (Monte et al. 1996). These models are, generally, very easy to use and require a limited amount of data and driving variables that can easily be evaluated for each specific case.

THE USE OF THE OPTIMAL CONTROL THEORY FOR THE IDENTIFICATION OF EFFECTIVE COUNTERMEASURE STRATEGIES TO REDUCE THE RADIOACTIVE CONTAMINATION LEVELS OF WATER BODIES: COST BENEFIT ANALYSIS AND MULTI-ATTRIBUTE ANALYSIS

Let us suppose that a process is characterised by a set of n magnitudes $x_i(t)$ ($1 \le i \le n$) (the state variables) depending on the time t, and controlled by a set of p control parameters $u_i(t)$ ($1 \le i \le p$) that represent the external intervention on the system. To better understand the concept of "control parameter" for the present applications, it is useful to emphasise that the values of the functions $u_i(t)$ may be fixed, somehow or other, according to human wishes and needs.
Let us suppose, moreover, that the functions x_i are solutions of a set of differential equations

$$\frac{dx_i}{dt} = f_i(x_1,\ldots,x_n,u_1,\ldots,u_p) \qquad (1)$$

which satisfy some specific initial conditions $x_i(0)=x_{i0}$.
Let us defined a functional J, called the "performance index" of the process,

$$J = \int_0^{t_1} F(x_1,\ldots,x_n,u_1,\ldots,u_p)dt \qquad (2)$$

where F is a given function of the state variables and the control parameters. The functional J maps the space of functions $\{x_i,u_i\}$ into the set of the real numbers.
The fundamental problem of the theory of optimal process may be stated as follows: it is necessary to find the control functions $u_i(t)$ and the state variables $x_i(t)$ that minimise the value of J and satisfy the equations (1) and the relevant initial conditions. The functions u_i and x_i are the unknowns of the stated problem that is, by no means, easy to solve. The optimal process control theory is based on a set of complex mathematical procedures and theorems (see for instance Pontryagin et all., 1964). The detailed analysis of the mathematical aspects of the optimal control theory is beyond the scope of this report. On the other hand, in the present application for identifying the most efficient countermeasure in an aquatic system contaminated by radioactive substances, the existence and the uniqueness of an optimal solution of equations (1) and (2) may be intuitively demonstrated (see Figure 11).
The performance index J is the net total "cost" of a countermeasure. Such "cost" is the sum of the following terms: the production cost of the countermeasure (that is the direct cost borne to put the countermeasure into practice), the cost of the human health detriment due to the contamination of the environment and of the food chain, the cost of the environmental impact of the countermeasure and the cost of the social impact of the countermeasure.
The state variables $x_i(t)$ may be subdivided in two categories:
a) the concentrations of radionuclides in the components of the aquatic environment (water, sediment, etc.) at time t;
b) some specific environmental magnitudes that, due to the global perturbation of the environment following the countermeasure accomplishment and the radionuclide contamination, influence the behaviour of the environmental system and, in turn, are influenced by the changes induced in the system itself (for instance, the alterations of the chemical characteristic of the water, of the concentration of nutrients, of the amount of prey fish and of predatory fish).

The set of equations (1) is the environmental model accounting for the ecological processes, the migration of radionuclides through the environment and the impact of the countermeasure on the ecosystem.
The control variables $u_i(t)$ are quantities that define the entity of the countermeasure (for instance, the thickness of the removed layer of bottom sediment, the amount of the chemical substance introduced to modify the water pH).
The "cost" definition and evaluation, a problem creating obvious difficulties, are of paramount importance. At present stage we will consider the "costs" as real numbers whose values increase as the health detriment, the environmental impact and the social impact of the countermeasure increases. We will investigate subsequently the possibility of defining costs in an objective way relating it to a "monetary equivalent".
The production cost, per unit time, of a countermeasure is a function of the "countermeasure entity " (u) and of the countermeasure duration time (Δt)

$$C(u, \Delta t) \tag{3}$$

The cost, per unit time, of the health detriment is a function of the concentration of the radionuclide in the environmental components:

$$H(x_1, \ldots, x_n) \tag{4}$$

The cost, per unit time, of the environmental impact is, obviously, a function of the countermeasure entity and of the levels of contamination of the environment:

$$E(u, x_1, \ldots, x_n) \tag{5}$$

Similarly the cost, per unit time, of the social impact (social disruption) is

$$S(u, x_1, \ldots, x_n) \tag{6}$$

The total cost J is the integral over time for the sum of the above terms:

$$J = \int_0^\infty \left[C(u, \Delta t) + H(x_1, \ldots, x_n) + S(u, x_1, \ldots, x_n) + E(u, x_1, \ldots, x_n) \right] dt \tag{7}$$

In principle, the above formula must also account for the physical risks and the anxiety due to the countermeasure and, as a negative term, for the reassurance that a countermeasure provides.
The functions $x_i(t)$ are solutions of the system of differential equation (1), which is the mathematical model predicting the migration of the radionuclide through the environment and the food chain. Figure 11 shows schematically the principle of countermeasure optimisation. The health cost is a decreasing function of the countermeasure "intensity" u. Of course, the social and environmental impact and the production cost of the countermeasure are increasing functions of u. As consequence, the total cost may show a minimum value corresponding to the most cost-effective countermeasure.

As emphasised above, the main problem of this approach is represented by the meaning of the term "cost". The production cost of a countermeasure is, obviously, its monetary cost. Unfortunately, the other terms in formula (7) are not directly and easly related to "monetary costs". The general approach is based on the assumption that the cost of the health detriment is an increasing function of the dose term. The approach suggested by ICRP (ICRP, 1982, ICRP, 1989) is based on a linear relationship between the population collective dose and the relevant cost:

$$H = \alpha D \tag{8}$$

α is a constant expressing the cost of the unit dose (ECU man-Sv^{-1}) and D is the collective dose (man-Sv). α is the sum of the following terms:

1) α_0 = the detriment to the society due to the death, the shortening of life or the loss of productivity of an individual (death or contraction of a disease) as a producer of goods and /or services;

2) α_1 = the patient treatment;

3) α_2 = the subjective price of the risk;

4) π = the psychological impact of the risk.

Some of the above components (α_0 and α_1) may be objectively evaluated. Unfortunately the characteristics of terms 3) and 4) are such that a similar straightforward evaluation is not possible. The subjective price of the risk has been introduced in the theory of cost-benefit analysis as a first approximation to account for the fact that the class of individuals incurring the detriment and the class of individuals receiving the benefit are not necessarily identical. It is implicit that if risk is incurred by a specific group of individuals it must be more carefully managed. The subjective price of risk reflects the reluctance of individuals to be affected by risks without deriving benefits from them (Babaev et al., 1986). Therefore, the choice of the value of α may be, to a large extent, arbitrary. A first suggested approximation is $\alpha \approx 10^4$ ECU man-Sv^{-1}. Equation (8) is the simplest functional relationship between the collective dose and the detrimental cost. Other models may be used. For instance, formula (8) may be modified to account for some important concepts of the risk analysis: risk aversion and "de minimis doses". The detriment cost, when the perception of the risk severity by the population is accounted, may be regarded as a non-linear function of the collective dose. This implies that H may be decomposed into the sum of different terms, each related to a specific detrimental consequence, that are proportional to a power of D. The exponent of D is called the "aversion factor". Risk aversion accounts for the occurrence that high consequence risks are considered more harmful than risks of low consequences even in case these two kinds of risk imply equivalent detriments to the society as a whole. On the other hand, it is possible to envisage that, although the collective dose may be very large due to the number of affected people, below a specific level of individual dose the detriment is negligible. This is a typical threshold effect that may be accounted for the evaluation of H. During the execution of the MOIRA project, this sort of problem will be analysed in detail for each specific case to identify the most suitable strategy of cost evaluation. Of course the final software product, will be characterised by the necessary flexibility to help the decision makers in choosing the countermeasure according to specific needs and policies.

The identification of the criteria to assign a monetary cost to the ecological and social impact of a countermeasure has obvious difficulties and is, to a great extent, arbitrary. As general rules

are not available, the problem of countermeasure optimisation must be addressed and solved according to strategies applicable to each specific circumstance.
These difficulties may be overcome when the costs are expressed as functions of the economical loss. For instance the reduction of productivity of a water body (ecological impact) due to the practice of a countermeasure may involve a loss of profit for the fishery (monetary cost of the ecological impact). Similarly, the restrictions of the recreational usage (swimming, game fishery etc.) of a water body and its shoreline (social impact of a countermeasure) may induce a loss of profits for the categories of employers and workers providing services related to diversion activities.
The above scheme supplies a general criterion necessary to carry out a cost-benefit analysis of countermeasure practices on the basis of the total cost. Nevertheless, it is reasonably to hypotheses that some financial constraints or the need to limit ecological and health detriments, due to specific policies of intervention, may be considered. For instance, in some cases, it may be required that the dose must not exceed a pre-determined value or that the production cost of the countermeasure must be lower than a limit imposed by budget constraints or, moreover, that some peculiar characteristics of the environment must be preserved. In such circumstances the most effective countermeasure must be identified in a region of the plan (u,cost) corresponding to the given constraints.
If limits to the terms in equation (7) must be included, some of the following constraints must be added to the equations for the identification of the most effective countermeasure:

$$\begin{aligned} &C(u,\Delta t) \le C_0 \\ &H(x_1,\dots,x_n) \le H_0 \\ &S(u,x_1,\dots,x_n) \le S_0 \\ &E(u,x_1,\dots,x_n) \le E_0 \end{aligned} \qquad (9)$$

As an example of application of the previous methods we will examine the following scenario:

A water body is contaminated by a radioactive substance characterised by a committed effective dc se per unit intake e(L); what is the optimal intervention level for the ban of fish consumption?

We put
V = amount of fish caught per unit of time (kg s^{-1});
b = cost of the fish (ECU kg^{-1});
Q(t) = radionuclide concentration in fish at time t (Bq kg^{-1});

The cost per unit of time due to the dose to population is

$$\dot{D} = \alpha Q(t) V e(L) \qquad (10)$$

The production cost of the countermeasure is equal to loss of money due to the consumption ban:

$$C(t^*) = bVH(t^* - t) \qquad (11)$$

where H(t*-t) is the heavy side step function whose value is 1 when t*-t≥0 and 0 when t*-t<0 and t* is the instant at which ban stops.
The total cost J may be calculated as follows:

$$J = \int_0^{\infty}[\alpha Q(\tau)Ve(L)H(\tau - t^*) + bVH(t^* - \tau)]d\tau \qquad (12)$$

The concentration of a radionuclide in a fish species living in a water body may be evaluated approximately by means of the following simple formula

$$Q(t) = Q_0 e^{-\lambda_{eff} t} \qquad (13)$$

where λ_{eff} is the ecological effective decay of the radionuclide concentration in the fish species (Brittain et al., 1995) and Q_0 is the maximum concentration of radionuclide in fish. Using formula (13) we get:

$$J = \alpha Ve(L)Q_0 \int_{t^*}^{\infty} e^{-\lambda_{eff}\tau} d\tau + \int_0^{t^*} bV d\tau \qquad (14)$$

The total cost J reaches a minimum value when

$$\frac{dJ}{dt^*} = 0 \qquad (15)$$

Solving equation (15) we get:

$$t^* = \frac{\ln \frac{\alpha e(L)Q_0}{b}}{\lambda_{eff}} \qquad (16)$$

The above approach is applicable if the contaminated food may be readily substituted by uncontaminated aliment and if no other impacts on the environment, on the social and economical systems and on the lifestyle are regarded of importance.
As previously stated, it is very difficult to assign a monetary cost to some factors related to the ecological and social impact of a countermeasure. Many factors affecting a decision will be better perceived by the decision-maker if they remain as they originally are, without any artificial costing in monetary units.
It can be particularly interesting, for instance, considering the number of persons affected by the ecosystem being restored, or the size of a pristine area affected, which not always will be well measured through a monetary value, because normally pristine areas do not clearly contribute to the economy of a country.
This kind of factor can be introduced into the decision-making process if multiattribute value (utility) analysis is used (ICRP, 1989). It basically consists in assigning preferences to every

factor (attribute) which is considered. To do this, the simplest technique is to assign a value to each attribute which can vary betwen 0 and 100 for the worst and the best score obtained for that attribute for each alternative available: for instance, a 0 value for dose reduction will be given to that alternative reducing the dose least, and a 100 value in the dose reduction scale will be obtained by the alternative offering the highest dose reduction; a 0 value in the economic cost scale will be given to the alternative with a highest cost, and a 100 to that with the lowest cost, etc.

After scaling the attributes (and thus adimensionalising them), a hierarchy can be established by assigning a different weighting factor to each attribute, with higher weighting factors for those in which the decision-maker has a greater interest to optimise. Additive or multiplicative combinations of the scaled attributes, with their corresponding weighting factors, allow one to define the so-called utility function, which can be maximised in order to obtain the best alternative.

Multiattribute value analysis is being incorporated in several decision-making support systems, like RODOS (French, 1996), in an interactive way which will facilitate the decision- maker or the user of the system to understand and observe the impact that preferences will have in the ranking of the different alternative countermeasures. Such an interactive module is of primary importance for a system like MOIRA, and it will be developed along the project based on existing software and on similar developments for EU projects.

In any case, in a system like MOIRA, it can be useful to offer open possibilities for decision-making, based either on cost-effectiveness, cost-benefit or multiattribute analysis. Cost-effectiveness can always be evaluated by the system; cost-benefit will need some conversion factors to monetary units input by the user of the system; and multiattribute value analysis will need the decision-maker (user) to express, with the help of guidance screens, his preferences for each attribute.

This could enhance the flexibility of the system in order to be adopted by the different countries in Europe. For all the methods, the basic inputs will be obtained from the ecological and countermeasures modules.

PRINCIPLES FOR THE DEVELOPMENT OF THE MOIRA EXPERT SYSTEM

Even if suitable models can be derived and validated, another important question needs to be addressed: "How can such models be disseminated usefully to those persons who are targeted as potential users?" The MOIRA project argues that expert systems are the response to that question. In the future, expert systems will create something of a revolution in the environmental sciences and management, just as computers did 10 to 20 years ago.

Expert systems are computer programmes that organise and structure the knowledge of experts and allow non-experts to use many different types of models, within the framework of the "traffic rules" appropriate for each model. The expert system is the next step in the technical evolution of this area.

In this section, the focus is on the great potential of a new expert system, the MOIRA system, for aquatic radioecology. Expert systems of this type will be of great interest in research, education, practical environmental management and consulting. We plan to develop this system from prototype and "research" to "management".
An expert system may be regarded as a "navigational tool" to reach certain defined goals along identified routes. In this expert system, the goals are (1) models (i.e. quantitative, validated, predictive models that link defined environmental target variables, fallout and ecosystem sensitivities for entire defined ecosystems), (2) remedial methods, (3) dose models and (4) a model for ecosystem index and environmental cost/benefit analyses (CB-analyses and multi-attribute analysis). The focus in this section is not on techniques of constructing expert systems, of programming, or of linkages to other programmes (like databases, statistical packages and modelling routines). Our aim here is to present components of the basic structure of the system.
Much of this information comes from Håkanson and Peters (1995). The expert system under discussion here is being developed for natural lakes, rivers and coastal areas in flexible and user-friendly programming languages, making it possible to run the program on both PCs (i.e. IBM and compatible machines) and Macintosh machines.

The expert system is intended to be:

- synthetic, such that knowledge may be collected, processed and included in the system in a structured manner,

- accessible, so that the information in the system will be found and used simply and rapidly,

- flexible and extensible so that new knowledge, literature references, and models can be included as they become available.

- practical, in the sense that many different types of predictive models could be very easily run by non-experts, but only in such a way that all "traffic rules" are followed and all directions given by the modeller are obeyed. The models themselves are derived outside the framework of the expert system, since that is work for the experts, not the users of this expert system.

It must be admitted that a lot of work must be done to develop this expert system and that even a highly developed form cannot deal with all problems. For example, consider an official at an environmental authority who has the task of examining a fallout from an accident at a nuclear facility somewhere in Europe. This expert system does not address probabilities of such accidents, the atmospheric transport under different meteorological conditions, etc. It starts with a defined fallout, for example, of radiocesium in Bq/m^2 in a given month. The basic questions are: "What concentrations of radiocesium in water (used by man for irrigation, drinking, etc.) and predatory fish (used by man as food) can this fallout cause?" What peak values can be expected for the target variables? How is the situation likely to develop in the coming years? What can be done by means of environmental remedial measures, in an rational way from both an economical and an ecological perspective, to minimize the risks to man? The expert system is designed with these questions in mind. The intention is to assist the user to move through the system and the knowledge it represents to definitive useful answers as soon as possible after the accident, and as a tool for further monitoring, research and action. The system should be designed so that it is very easy for the user to "jump" from one stack of information to another, to run a model, to make simulations, to select confidence bands for the predictions, to collect data on necessary driving variables from GIS (Geographical Information Systems), to make calculations in spread-sheet programs, etc. There is now a great need for methods that summarize and compile the vast amounts of existing data of relevance in radioecology. This may be achieved by using different types of Geographical Information Systems, which provide large quantities of data for defined geographical sites. Many such GIS-systems are already commercially available today (e.g. MapInfo). GIS-systems may be used within the framework of expert systems, but the latter are basically very different from the former.

Once established, the expert system would be of greatest interest to water managers, because it would organise our knowledge in a useful and practical way. Such a system should also interest scientists, because it would create order and structure within a very complicated sector and because the process of building, applying, and improving models captures much of the scientific enterprise. The completion, extension and updating of such an expert system is so great a scientific undertaking that it will provide fertile ground for relevant, scientific advance for many years to come.

The models to be included in the expert system are intentionally not very complex models, but comparatively small, general, predictive models to be driven by readily accessible environmental parameters from European databases and GIS-systems. These readily available environmental parameters can be related to different ecologically relevant and practically useful remedial strategies. By changing different environmental parameters related to alternative remedial strategies, like pH in relation to lake liming, K-concentration for potash treatment, lake total-P for lake fertilization (see Håkanson and Andersson 1992 and IAEA 1996 for results and discussions of many different remedial measures tested to minimize concentrations of ^{137}Cs in lake water and fish), it is possible to simulate realistic, expected effects for the target variables, ^{137}Cs in lake water and in predatory fish. Since it is also possible to cost these remedial measures, one can relate the costs to the environmental benefit, as expressed by changes in the target variables. This illustrates the great advantage of validated, predictive models in lake management. There are certain principle that would also apply for ecosystem management in general.

Our focus now shifts to providing some general ideas concerning the user software. The purpose is to form accepted and well-defined guidelines for the system software. The software system will guide the user from input handling, via execution of radioecological models, selection of remedial actions and their ecological, economical and social consequences to result presentation and comparisons.

There are some basic requirements that the software system must fulfil. Firstly, the system must be easy to handle. This means that the user will work in a well known environment, that he or she can get help and be informed when needed and that the user can be navigated by the system to find a suitable track from start to end. Secondly, there must be a very high level of control of data used in the system. This regards input given automatically from included databases or by the user, data calculated or derived in the system and the output data. Thirdly, the invisible interface to the models must be well defined showing what goes in and what comes out. The model makers must be very clear in setting up the contents and the structure of input data and in describing the output. Forthly, the error handling must be very rigorous to reduce the risk of execution interruptions and loss of data. There must be functionalities in the system that can handle errors in a way that allows the user to continue the execution with data used. Finally, everything done in system development must be documented.

To start the system execution some initial data concerning the fallout of radioactive material is needed. This input is given by the user and consists mainly of contamination levels and the time of year for the fallout. The contamination site must also be known and this could be given in a GIS (Geographical Information System) which includes maps for Europe. Since GIS is not just a system for map handling, it is also very good in handling attribute data, GIS could be used to supply the system with suitable input for a selected site which is marked on a map. For instance, basic descriptions like longitude, latitude. Based upon these, as well as season of the year, turnover times for water could be estimated by use of moderators. The data bases included in the GIS system could, for example, contain lake volumes, depths and areas, pH values, biomass, in- and outflow of water and precipitation data. Complete sets of this data for the whole water system in Europe are probably not available, so we must also allow the user, in some cases, to use their own data and to modify existing data. Thus, the start section of the system must contain functionalities for viewing existing data from a selected site or a neighbouring or similar site, helping the user to find the best data. Of course, this must be done within the framework of a careful control performed by the system.

Data must be prepared to be used as input for the radioecological models. We call this prepared data model variables. Some of them could be used as they are, other model variables have to be calculated from the given input. An important task is that the modellers must define and structure the data needed for the radioecological models. This work includes, among other things, to define which data is needed, which physical units that will be used, allowed limits for each variable and how the input is structured. The software system will check all model variables and create the input files needed for the different models according to the modellers specification. The user will have the possiblity to check the model variables before the models are executed. Accepting the input will start the model execution.

Concerning the radioecological models, which should be seen as external modules called when needed, it is important that they are checked and tested to prevent calculation errors and errors resulting in execution interrupts. As for the input it is important to define and structure the output, because we must know exactly what is coming out of the models and how the results can be used. One use is of course to view the results from the dose calculations as curves in a diagram or in a table. In addition, it will also be possible to view results from several runs in the same diagram for comparing analysis. The output from the radioecological modelling will also be used as input to the countermeasure modelling. All input and output will be saved in files using some type of identification system that uniquely identifies a run. We can call this identification system a history in which all input and output are given as well as the dates, user, execution pathway, etc, making it possible to reconstruct a run, perhaps with some modifications, and to view data used and to present the results.

Many of the general thoughts and requirements above are also valid for the countermeasure modelling where ecological and social impacts are treated together with the cost/benefit analysis, for example the definition and structure of input/output and the error handling. The possibilities for result presentation will also be the same and the execution pathway and input/output will be added to the history.

So far, it is not clear which data is needed for the countermeasure modelling, but the input could be given by the user, taken from databases or from the GIS system. Besides, the results from the ecological calculations will act as input to the countermeasure models. The user must, perhaps after suggestions from the system, decide which or which combination of remedial actions that will be used, if there are some limitations in the remedial work and perhaps the upper limit for costs. The size of the contaminated area or volume is something that could be retrieved from the GIS system.

To study different remedial actions the user must have the possibility to rerun the countermeasure models an arbitrary number of times to compare the different actions. An

alternative is that this is done automatically by the system, running the models for all possible, or a set of, remedial actions.
The results from the countermeasure modelling could vary considerably and perhaps all results could not be presented numerically. It should be a great advantage not only to show the costs and the benefits of different remedial actions, but also if there are some unfavourable consequences.
Finally, the purpose of the expert system must be to give the user possibilites to compare different remedial actions in cost and benefit, given an initial contamination of radioactive material. So, it should be possible to make a comparative analysis of results from the countermeasure calculations. This shows that it is important to construct an easily understandable and flexible method for result presentation where the effects of different remedial actions can be studied and compared.
There will be many different physical parameters used in the system and we must have full control over these data and their physical units. To make the system easy to handle it could be advantageous to use a flexible unit system where the user selects which units to use. Inside the system it is important to be consistent, therefore SI units will be used.
Perhaps the most difficult task for the developers of the expert system is to produce a software that could be general available, in terms of money, and accepted in a way that the system can be used by a wide range of people without expert knowledge of radioecological modelling or computer techniques. Good conditions for this can be created by using accepted, easily accessible and widely used hardware and software. Thus, we will work in an environment based on PC and Windows. Concerning GIS software we can note that it is not necessary to have a complete and expensive system with lots of functions and possibilities. Instead we will develop a tailored GIS-application containing the tools we need for our purposes. This could for example be done with the MapInfo/MapBasic system from MapInfo Corporation. Perhaps we also need a presentation tool, but before we make any proposal on this, we will investigate the possibilities to let the system developers program these parts with the Visual C++ tool. All the suggested software is also available for the Mac environment.
MapInfo/MapBasic is suggested as a GIS-tool. Irrespective of which tool to be used, we must supply the GIS-system with digitized maps and belonging attribute data, which will function as input to the external models. An investigation of which hydrological data is available in a scale of western (and perhaps also eastern) Europe and the cost for it must be performed rather early in the project.
In developing the system it is also important to use a consistent terminology and an attractive layout in the user interface and in printer output. This means among other things that we will use a predetermined font, distinct symbols and a predetermined set of colours. The layout of dialog squares, presentation windows, help text, diagrams and tables will follow certain rules. In this way the user will have a distinct screen where information is easily readable and understandable.

CONCLUSIONS

The general structure of the MOIRA project is given in figure 12.
The identification of the optimal remedial actions for restoring radionuclide contaminated aquatic systems requires two indispensable tools:
- A complete set of models for predicting the time behaviour of radionuclides in the fresh water environment and the ecological, the social and the economical impacts of the countermeasures;
- General techniques for ranking the different applicable countermeasures according to their effectiveness when the benefits due to the dose reductions and the ecological, the social and the economical detriments are accounted for.

Two techniques for ranking the effectiveness of the countermeasures are of major importance: Cost-Benefit Analysis (CBA) and Multi-Attribute Analysis (MAA). CBA offers the obvious advantage of assigning a real number (the cost) to each countermeasure action. As consequence, it allows one to rank unambiguously the countermeasures according to their cost effectiveness. When a general consensus exists concerning the evaluation of the costs relevant to an intervention, CBA directly guides the environmental managers to the choice of the "best" countermeasure action for a specific contamination scenario.
Unfortunately, an exhaustive cost-benefit analysis to evaluate the effectiveness of countermeasure actions is often not feasible due to the intrinsic difficulties in objectivly defining the social, the ecological and the health detriment costs connected to the countermeasures.
MAA was developed to support decision makers to account for a variety of factors affecting the decision process that cannot be evaluated in terms of monetary costs. MAA allows one to assign to each remedial action a vector whose components are the "performance indices" related to the effectiveness of the countermeasure as regards to its impact on the environment, on the society, on the economy, on the population health and on all other factors that, in each specific case, may be considered of importance.
Generally, the components of these vectors are heterogeneous. However, infrequently the performance indices may be suitably weighted to be combined in a single quantity (the "countermeasure value"); as at least one of the components of the MAA vector is a cost value (for instance, the production cost of the countermeasure), then in such case Multi-Attribute Analysis turns into CBA.
Both CBA and MAA require a detailed description of the evolution of the environmental, the social and the economical scenarios resulting from the countermeasure application (Scenario Description, SD). SD is the first information that the expert system must supply to the decision makers. In some circumstances there are insufficient clear and objective data to develop CBA and MAA. In such case SD is the only aid to assist decision makers. Of course, the degree of subjectiveness of the choice of the optimal countermeasure increases from the CBA to the SD (see figure 13).
Among the factors relevant to the evaluation of the countermeasure effectiveness, the ecological impact is one of the most difficult to quantify. The ecological cost may be defined as the impact of the countermeasure on ecosystem functioning.
One of the possible definitions of ecosystem functioning may be based on the detailed description of the growth and reproduction rates of the species and on the analysis of the interactions among them and with the environment. Simpler alternatives are available: a) to assess changes in biodiversity, that is the number of species and their abundance; b) to focus attention on a selected set of key species and communities. This second approach was

successfully applied during VAMP project for lakes and will be extended to other aquatic systems.
As repeatedly emphasised in the present report, the main goal of the MOIRA project is to construct an expert system to choose optimal countermeasures for restoration of radioactive contaminated aquatic systems. The expert system will be organised and structured to allow non-experts to use many kinds of models and to apply complex techniques of decision making. It will be an extremely useful tool for a variety of purposes: research, education, environmental management and consulting.
The MOIRA software will make use of a Geographical Information System (GIS) containing information, among others, on the main geographical, geological, hydrological and biological characteristics of the different European regions.
The models framed in the MOIRA software will be driven by simple environmental parameters that are strictly related to the main characteristics of the area where the countermeasure is applied. MOIRA software will be able to identify the most reliable values of the model parameters by means of appropriate sub-models using the environmental information obtainable from the GIS.
This strategy will make MOIRA software a very user-friendly tool for environmental managers not gifted with experience in environmental modelling.

Table 1 - Characteristic features in lakes of different trophic levels (from Håkanson and Peters, 1995)

Trophic level	Oligotrophic	Mesotrophic	Eutrophic	Hypertrophic
Primary prod. (g C/m^2 yr)	<30	25-60	40-200	130-600
Secchi depth (m)	>5	3-6	1-4	0-2
Chlorophyll-a (mg/m^3)	<2	2-78	6-35	30-400
Algal volume (g/m^3)	<0.8	0.5-1.9	1.2-2.5	2.1-20
Total-P (mg/m^3)	<5	5-20	20-100	>100
Total-N (mg/m3)	<300	300-500	350-600	>1000
Dominant fish	Trout	Whitefish	Perch	Roach
	Whitefish	Perch	Roach	Bream

Table 2 - Trophic characteristics for coastal areas smaller than 15 km^2 in the Baltic. Oligotrophic (low bioproductivity) to hypertrophic (very high productivity). From Wallin (1990)

Trophic level	Oligotrophic	Mesotrophic	Eutrophic	Hypertrophic
Secchi depth (m)	>6	3-6	1.5-3	<1.5
Chlorophyll-a (mg/m^3)	<1	1-3	3-5	<5
Total-N (mg/m^3)	<250	250-350	350-450	>450
Inorganic-N (mg/m^3)	<10	10-30	30-50	>50
Sedimentation (g/m^2 d)	<2	2-10	10-15	>15
O_2-B (mg/l)	>10	6-10	3-6	<3
O2-sat. (%)	>90	60-90	30-60	<30

Table 3 - Environmental variables of value in the analysis and classification of stream ecosystem

Geographic Variables	Chemical Variables
Latitude	pH
Longitude	Dissolved Oxigen
Physiographic region	Total nitrogen
Altitude	Chloride
	Orthophosphate
	Alkalinity
Physical Variables	**Biological Variables**
Temperature	Riparian vegetation
Discharge	Macrophyte cover
Velocity	
Depth	
Width	
Sampling date	
Slope	
Distance from source	
Substratum type and composition	

Table 4 - The division of rivers into rhithron and potamon zones and the four fish-zones of western Europe (from Hynes 1970). Capitals denote dominance, and italics sub-dominance

Rhithron	The stony stream to small river, which can often faunistically divided into *epi-*, *meta-*, and *hyporhithron.* In Europe for example, the meta-hyporhithron boundary is that between the trout and grayling zones.
Potamon	Which is also divisible into *epi-* and *metapotamon* both on faunistic grounds,e.g. barbel and bream zones in Europe, and on presence or absence of shallow shoals. The *hypopotamon* is the brackish water region affected by the sea and it is inhibited in Europe by such fishes as the flounder *Pleuronectes* and the sterlet.

Trout zone	Grayling zone	Barbel zone	Bream zone
Salmonids	Mixed fauna salmonids dominant	Mixed fauna cyprinids dominant	Cyprinids fauna with carnivores
TROUT	TROUT (GRAYLING) *fast-water cyprinids* accompanying cyprinids carnivores	trout (grayling) FAST-WATER CYPRINIDS *accompanying* *cyprinids* *carnivores* calm-water cyprinids	 fast-water Cyprinids ACCOMPANYING CYPRINIDS CARNIVORES CALM-WATER CYPRINIDS

The principal species are as follows:

Salmonids (trout and grayling) *Salmo trutta* and *Thymallus thymallus*
Fast-water cyprinids *Barbus barbus*, *Squalius cephalus*, and *Chondrostoma nasus.*
Accompanying cyprinids *Rutilus rutilus* and *Scardinius erythrophthalmus.*
Carnivores *Esox lucius*, *Perca fluviatilis*, and *Anguilla anguilla.*
Calm-water cyprinids *Cyprinus carpio*, *Tinca tinca*, and *Abramis brama.*

Table 5 - Huet's 'slope rule' for western European streams showing the fish zones which occur in watercourses of different widths and slopes (From Hynes 1970)

	Brooklet	Brook	Little river	River	Large river
Width m	0-1	1-5	5-25	25-100	100-300
	Slope in ‰ for breadth of				
	1 m	3 m	15 m	60 m	200 m
Trout zone	50.0-12.5	25.0-7.5	17.5-6.0	12.5-4.5	-
Grayling zone	-	7.5-3.0	6.0-2.0	4.5-1.25	>0.75
Barbel zone	-	3.0-1.0	2.0-0.5	1.25-0.33	0.75-0.25
Bream zone	12.5-0.0	1.0-0.0	0.5-0.0	0.33-0.0	0.25-0.00

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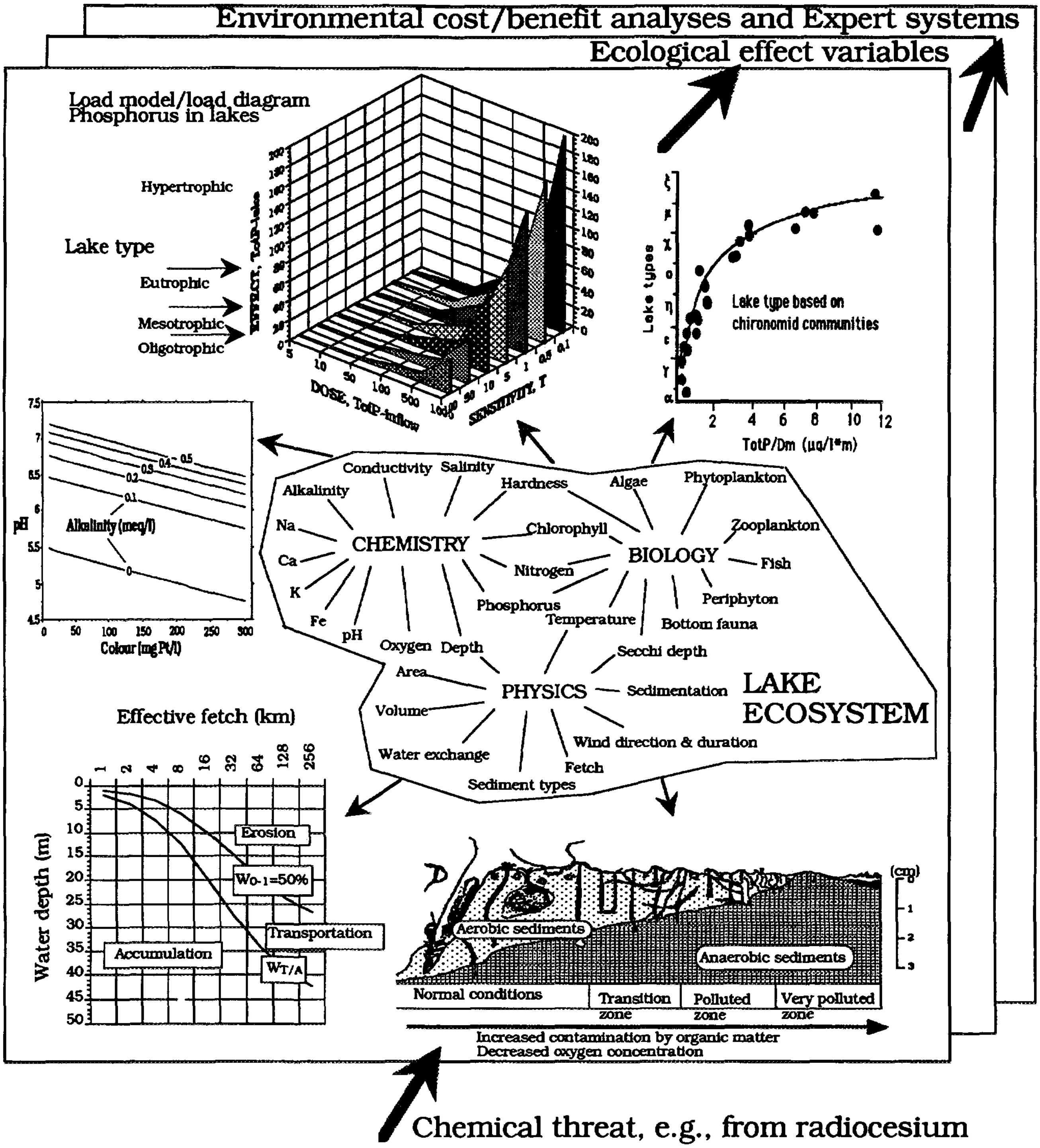

Figure 1 - Lakes, rivers and coastal areas are complex webs of physical, chemical and biological interactions. The prediction of the turnover of toxic substances within these ecosystems is based on the knowledge of a large amount of processes.

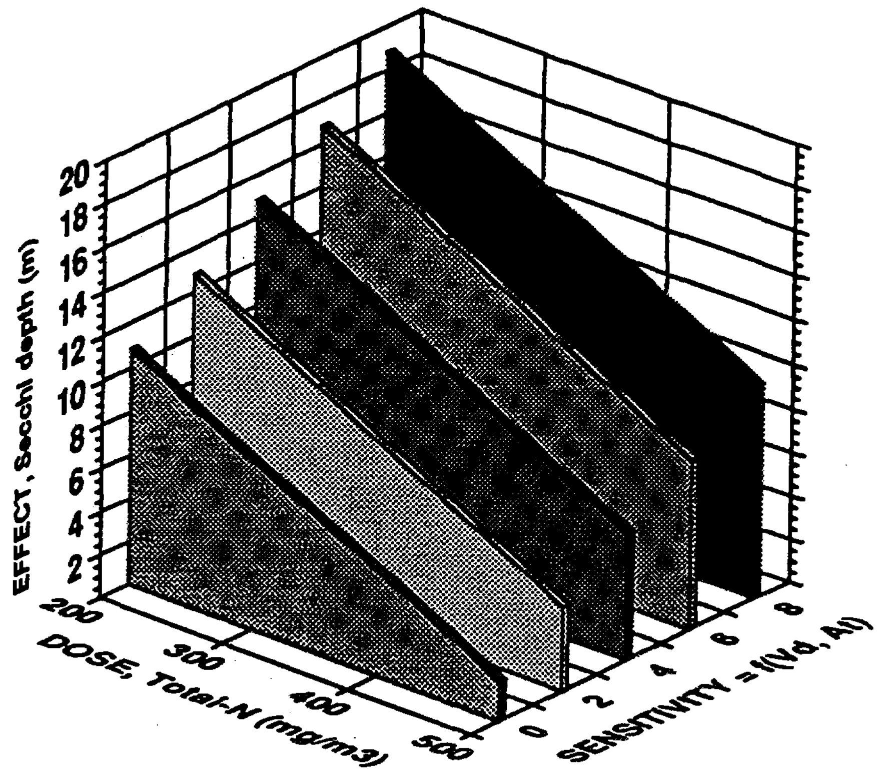

Figure 2 - A load diagram for Baltic coastal areas. The effect parameter is given by the Secchi disc trasparency, the "dose" by the concentration of total-nitrogen and the sensitivity by a function of two morphometric parameters (the form factor,Vd, and the section area, At). Redrawn from Wallin and Håkanson (1991).

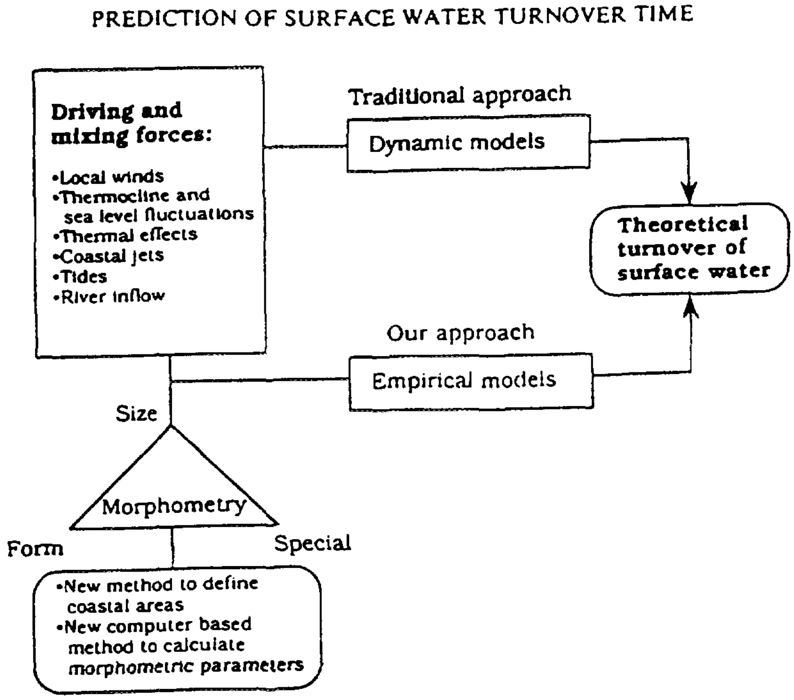

Figure 3 - Schematic illustration of two different approaches for estimation of water turnover in coastal areas. The traditional approach is dynamic modelling based on driving and mixing forces. Our approach is modelling based on empirical data of the surface water turnover time and morphometric parameters (From Persson et al. 1994).

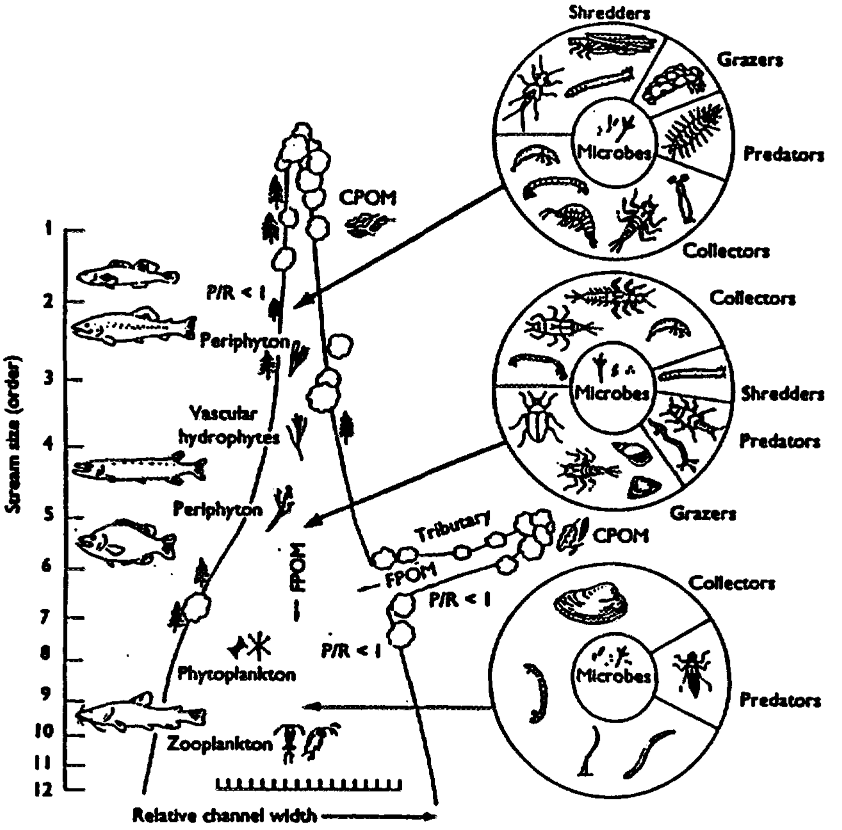

Figure 4 - Generalised depiction of the relationship between stream size (order), energy inputs and ecosystem function expected under the River Continuum Concept. (From Vannote et al., 1980)

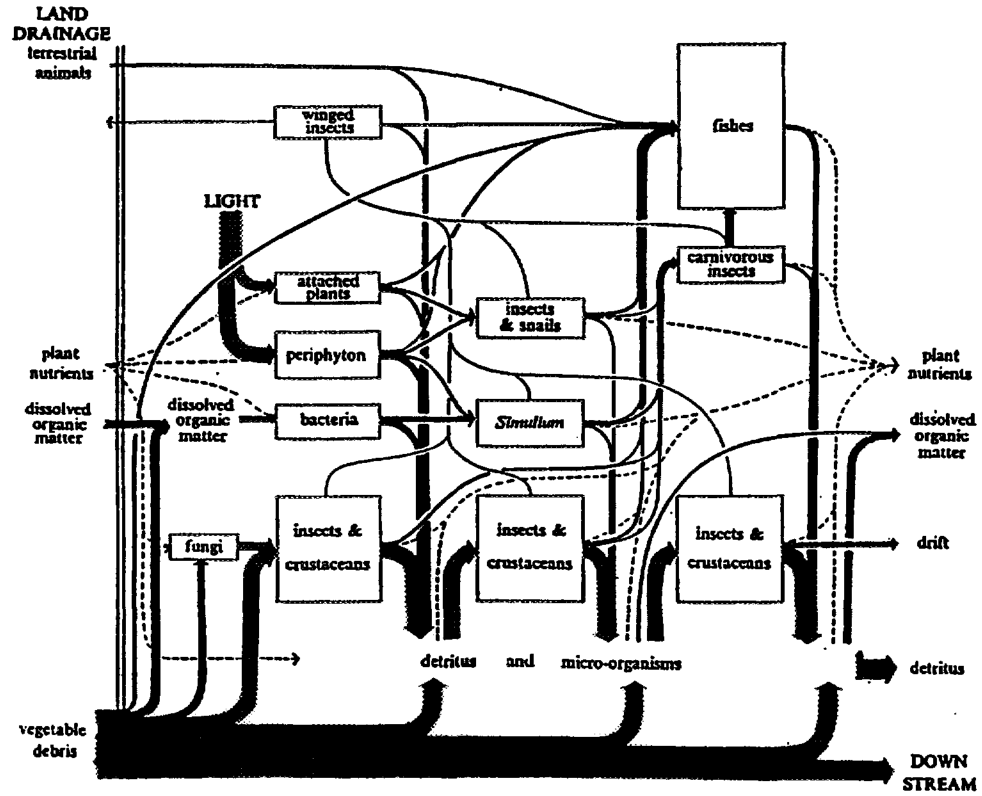

Figure 5 - A diagrammatic representation of the trophic relationships of the ecosystem in the rhithron. The relative sizes of the boxes are an indication of the biomass and the width of the arrows is proportional to the supposed relative importance of the energy pathways. Dotted lines represent salts in solution (From Hynes, 1970).

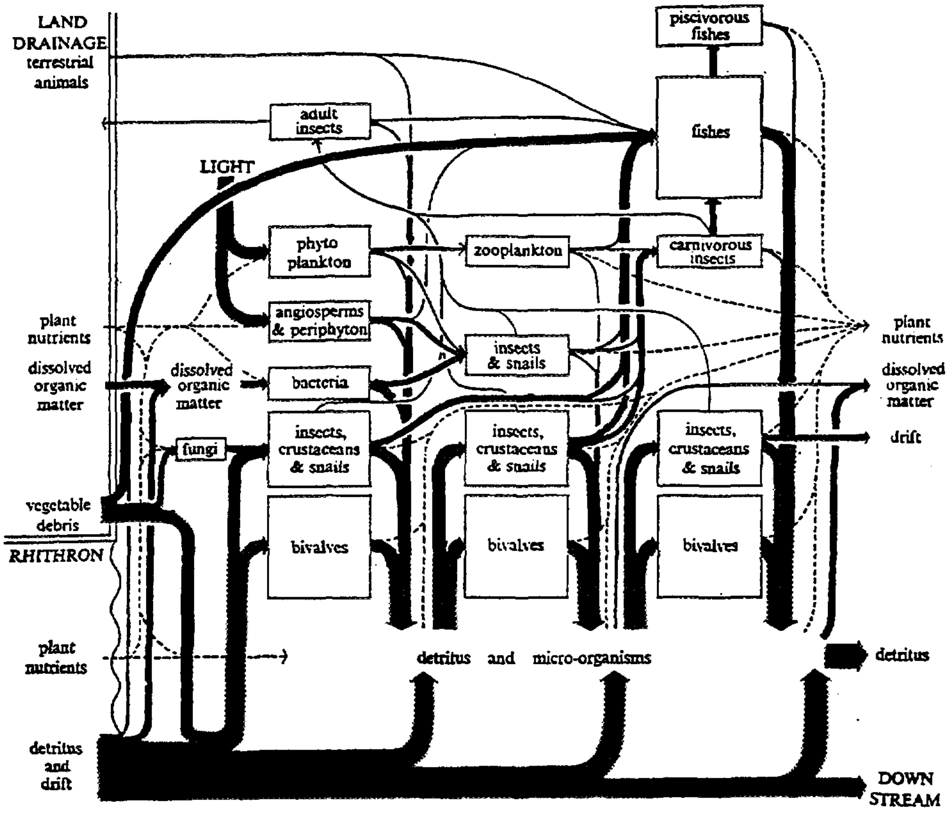

Figure 6 - A diagrammatic representation of the trophic relationships of the ecosystem in the potamon. Details as in Fig. 5 (From Hynes, 1970)

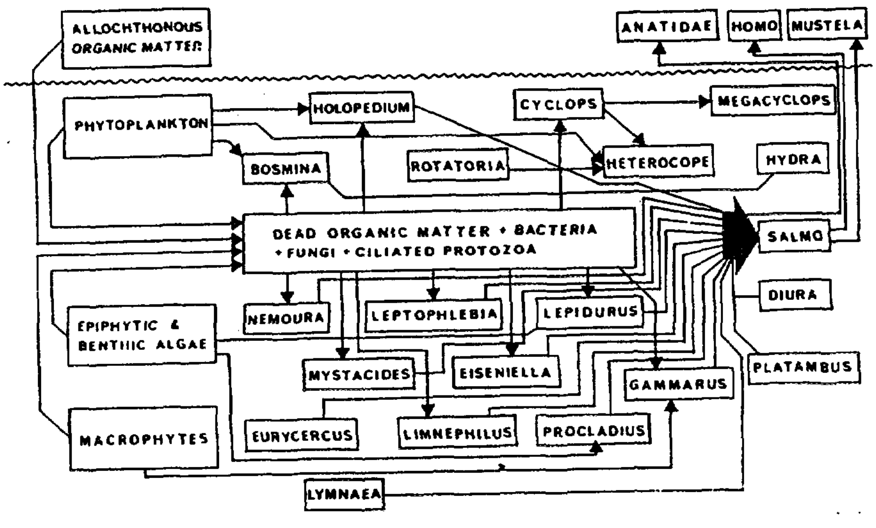

Figure 7 - Selected pathways in the food web documented for the Norwegian subalpine lake, Øvre Heimdalsvatn. The primary produced material is grouped into allochthonous organic matter, phytoplankton, epiphytic and benthic algae and macrophytes, while the secondary producers are generally taken to the generic level. The lake surface is indicated and planktonic organisms are placed nearer the surface than the typically benthic organisms (From Larsson et al., 1978).

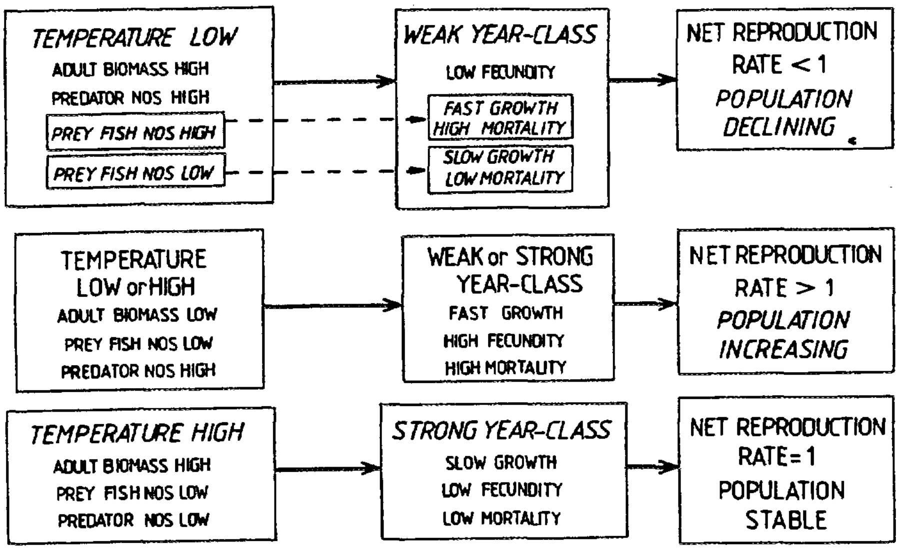

Figure 8 - Influence of environmental factors and initial population parameters on the reproductive performance of the perch (*Perca fluviatilis*) in Lake Windermere (NOS = Number Of Species). (Craig, 1980)

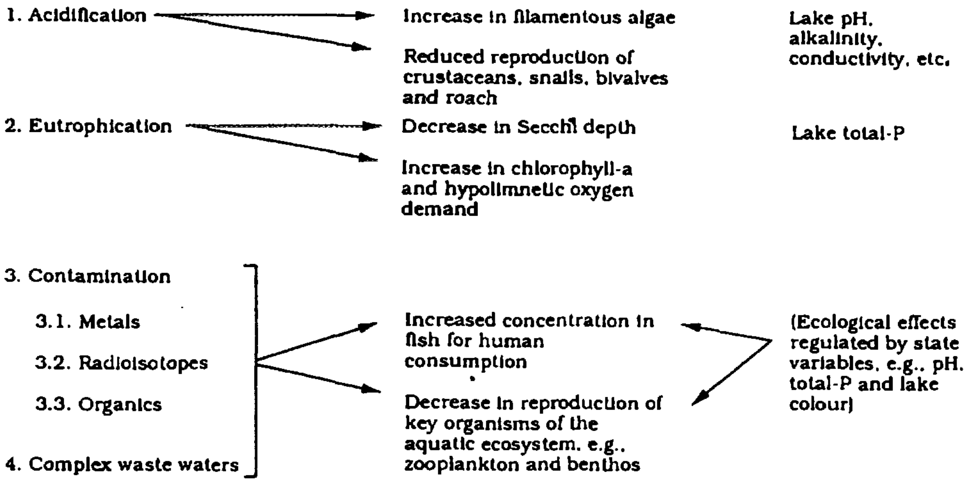

Figure 9 - Various "chemical" threats, some examples of ecological effects or their indices, and state variables. There are also many "physical" threats to limnetic ecosystems, like the building of dams, piers and marinas, and many biological threats, like the introduction of new species and the over-exploitation of established ones.

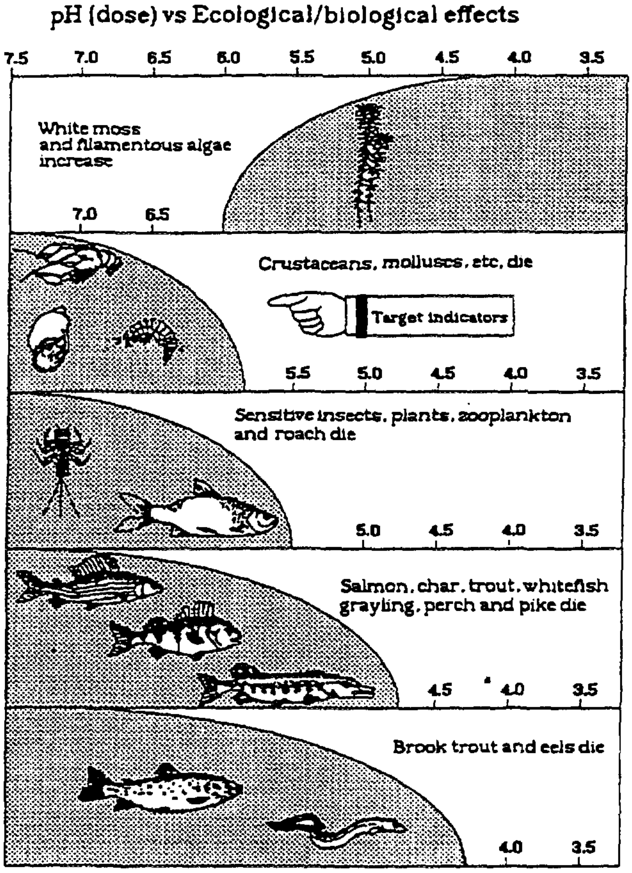

Figure 10 - Illustration of target organism using the example of biological and ecological effects of lake acidification. In predictive limnology, ecology and environmental sciences, a central objective is to find out which organisms are the most sensitive to a given contaminant, whether this is an acidifying substance (like S or N), a nutrient (like P or N) or a poison (like many metals, organics or radioisotopes). To emphasise their importance, the figure shows examples of key functional groups and target organisms for acidification. Crustaceans react rapidly to changes in pH, whereas certain fish such as brook trout and eels do not die until acidification is far advanced. White moss (e.g. Sphagnum) and filamentous algae should not be found in these lakes under normal conditions. So, the abundance of such species also indicates ecological effects of acidification. This figure raises many questions: Since pH is a variable, which measure of pH should be used for which organisms? Since all organisms vary spatially and seasonally in lake, how could indicators of the effect be operationally defined? What would they represent?

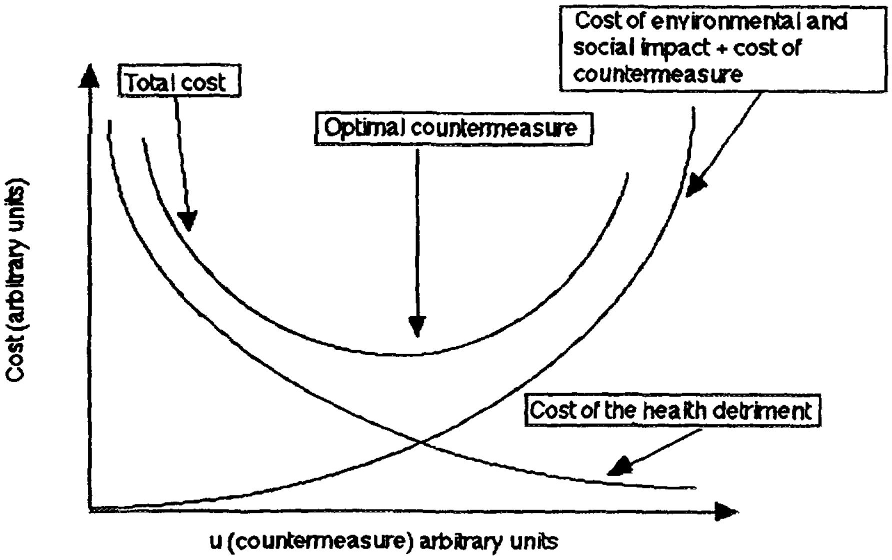

Figure 11 - Costs as function of the countermeasure intensity. The optimal countermeasure is achieved when the total cost reaches the minimum value.

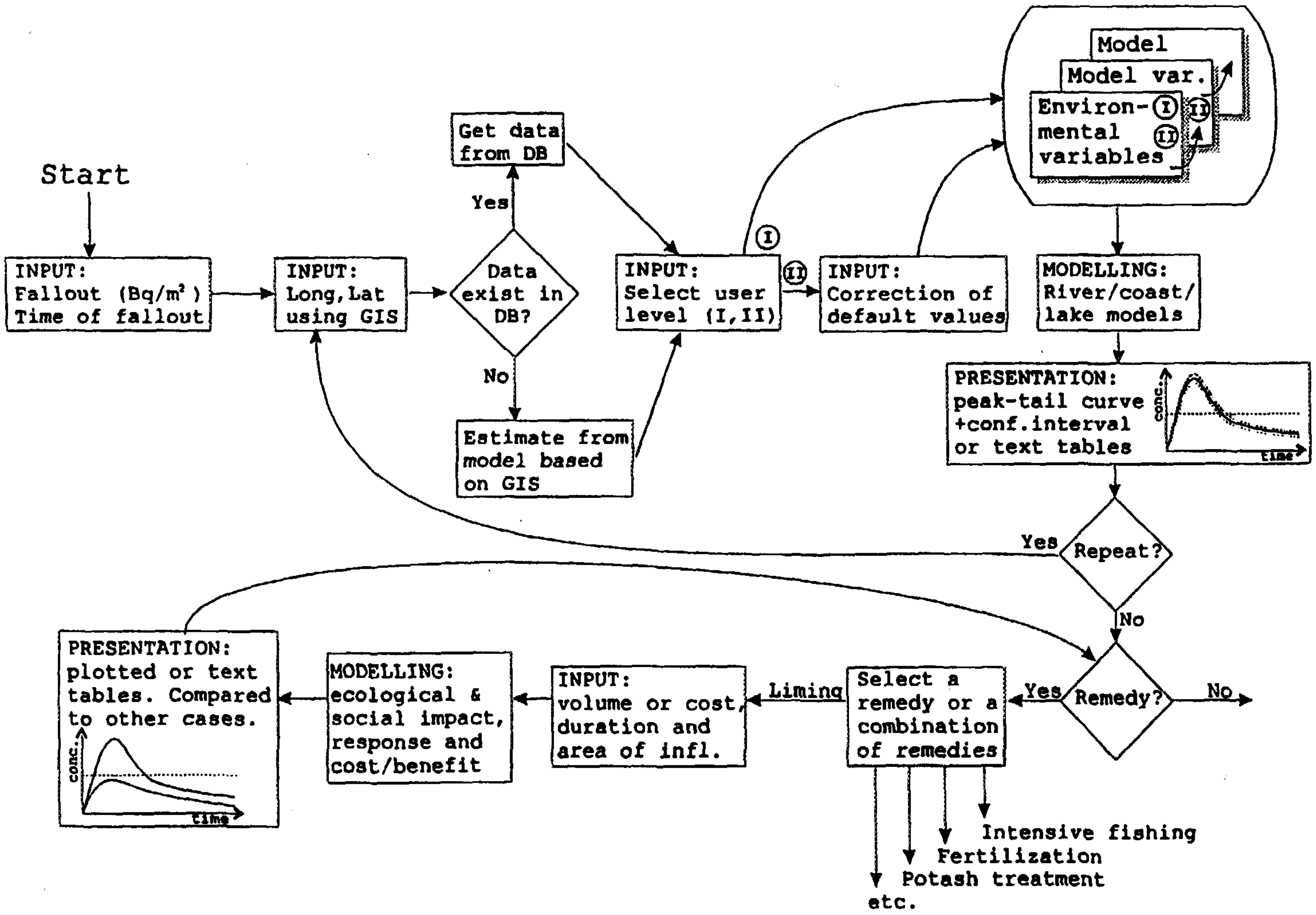

Figure 12 - General structure of the MOIRA project.

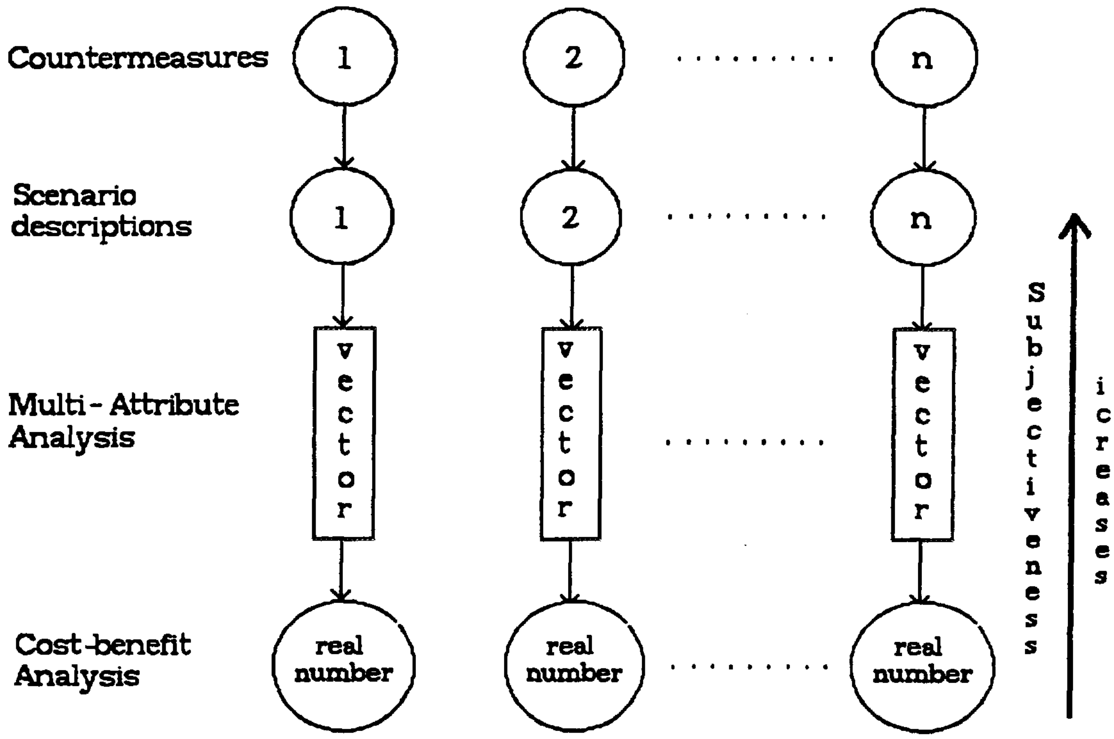

Figure 13 - The subjectiveness of the choice of the optimal countermeasure is low when a general consensus on the evaluation of costs allows to apply a Cost-Benefit Analysis, whereas it is high when only a description of the environmental and social scenario is possible.

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