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# PRINCIPLES FOR THE DEVELOPMENT AND IMPLEMENTATION OF THE MOIRA COMPUTERISED SYSTEM

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*MOIRA - PROJECT N. F14P-CT96-0036 EUROPEAN COMMISSION*

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*Edited by:*

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## RIASSUNTO

Lo scopo del presente rapporto, composto da articoli preparati dai ricercatori che partecipano al progetto MOIRA (A MODEL-BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING RADIONUCLIDE CONTAMINATED AQUATIC ECOSYSTEMS AND DRAINAGE AREAS), é la descrizione e l'analisi delle metodologie computerizzate che consentono l'identificazione di contromisure atte a ridurre i livelli di contaminazione radioattiva di sistemi acquatici continentali. In particolare vengono descritte la struttura del software MOIRA e i criteri per una valutazione oggettiva dell'impatto ecologico delle contromisure applicate a sistemi lacustri (definizione del "Lake Ecosystem Index" utilizzato nell'ambito del processo di analisi multriattributiva sviluppato per il sistema MOIRA). Vengono altresì presentati i risultati della classificazione delle contromisure in ambiente acquatico in relazione alla loro fattibilità e potenziale efficacia e la classificazione delle principali specie ittiche europee in relazione alla loro importanza in ambito radioecologico. Le metodologie descritte vengono applicate al lago Heimdalsvatn in Norvegia per dimostrare la fattibilità del sistema MOIRA. Nel presente rapporto sono anche inclusi due articoli che descrivono i principi per lo sviluppo di un modello per la previsione del trasporto di radionuclidi dall'ambiente urbano alle acque continentali e di un modello di migrazione dei radionuclidi nei fiumi.

## SUMMARY

The present report is composed of a set of articles written by the partners of the MOIRA project (A MODEL-BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING RADIONUCLIDE CONTAMINATED AQUATIC ECOSYSTEMS AND DRAINAGE AREAS). The project aims to describe and analyse computerised methodologies for the identification of the countermeasures in fresh water systems contaminated by radioactive substances. The structure of the MOIRA software and the criteria for the objective evaluation of the ecological impact of the countermeasures applied to lakes (defined by the "Lake Ecosystem Index", which is used in the Multiattribute Decision Analysis process developed for MOIRA) are described in detail. A preliminary classification of the countermeasures in relation to their potential effectiveness and feasibility and a classification of the main European species of fishes of radioecological importance are presented. The described methodologies have been applied to the Norwegian lake Heimdalsvatn to demonstrate the feasibility of the MOIRA system. Two further articles in the report describe the principles for the development of a model for predicting the radionuclide migration from urban areas to fresh water systems and the migration of radionuclides in rivers.

Keywords: environmental modelling, remedial actions, ecological impact, radionuclides

## SAMMANFATTNING

Denna rapport består av ett antal redovisningar skrivna av medarbetarna i EU-projectet MOIRA (A MODEL-BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING RADIONUCLIDE CONTAMINATED AQUATIC ECOSYSTEMS AND DRAINAGE AREAS). Projektets huvudsyfte är att med matematiska modeller insatta i ett användarvänligt datorbaserade expertsystem beskriva och analysera olika åtgärder (t.ex. pottaskebehandling och kalkning) för att reducera miljöeffekter av radioaktiva ämnen i akvatiska ekosystem. Denna rapport beskriver nya struktur för datorbaserad experthjälp som utvecklas i MOIRA projektet. Det gäller hur olika åtgärder påverkar ekosystemet i stort (definieras genom "Lake Ecosystem Index") och hur detta sätts in i ett vidare system där även sociala och ekonomiska förhållande vägs in (systemet för "Multiattribute Decision Analysis"). Vidare presenteras i rapporten en preliminär klassificering och genomgång av hur olika åtgärder kan relateras till de viktigaste europeiska fiskarterna i radioekologiska sammanhang. De beskrivna metoderna testas i ett scenario för sjön Heimdalsvatn, Norge. Syftet är att visa hur MOIRA systemet kan användas i praktiken. Två andra redovisningar i denna rapport behandlar en modell (principer och processer) för att prediktera transport av radionuklider från urbana områden till sötvattensystem och en modell som beskriver transport av radionuklider i vattendrag.

## SAMMENDRAG

Denne rapporten består av en serie artikler skrevet av deltagerne i MOIRA-prosjektet ("A MODEL-BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING RADIONUCLIDE CONTAMINATED AQUATIC ECOSYSTEMS AND DRAINAGE AREAS"). Prosjektet har som mål å beskrive og analysere edb-baserte metoder for å identifisere mulige tiltak i ferskvannssystemer som er forurenset med radioaktive stoffer. Rapporten beskriver i detalj oppbygging av programvare og valg av kriterier for objektiv evaluering av økologiske virkninger av tiltak i innsjøer. Dette er definert gjennom "Lake Ecosystem Index", som er anvendt i "Multiattribute Decision Analysis", og utviklet innenfor MOIRA. En forløpig klassifisering av tiltak i forhold til deres potensielle effektivitet og anvendbarhet, og en klassifisering av de viktigste europeiske fiskeartene når det gjelder radioøkologi er presentert. Evalueringsmetodene har vært anvendt på den norske høyfjellsjøen Øvre Heimdalsvatn, for å demonstrere anvendbarheten av MOIRA-systemet. To andre artikler i rapporten beskriver prinsippene for utvikling av en modell som forutsier transport av radioative stoffer fra urbane strøk og ut i ferskvann og transport av slike stoffer i elver.

## RESUMEN

Este informe incluye un conjunto de artículos escritos por los socios del proyecto MOIRA ("A MODEL-BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING RADIONUCLIDE CONTAMINATED AQUATIC ECOSYSTEMS AND DRAINAGE AREAS"). El proyecto tiene como objetivo el desarrollo de un sistema computerizado para la identificación de estrategias de restauración aplicables a ecosistemas de agua dulce contaminados con sustancias radiactivas. En el informe se incluye una descripción detallada de la estructura del software MOIRA. Se presentan los criterios seguidos para la evaluación objetiva del impacto ecológico de medidas correctoras de la contaminación aplicables en lagos (mediante el llamado "Lake Ecosystem Index" que se utiliza en el proceso de análisis de la decisión multiatributos desarrollado para MOIRA). Se incluye una clasificación preliminar de las medidas correctoras de acuerdo a sus posibilidades de aplicación y a su efectividad potencial; también se establece una clasificación de las principales especies de peces en Europa y su importancia radioecológica. Las metodologías descritas han sido aplicadas al lago noruego Øvre Heimdalsvatn a fin de demostrar el funcionamiento del sistema MOIRA. El informe se completa con otros dos artículos que describen, respectivamente, las bases del desarrollo de un modelo de migración de radionucleidos desde zonas urbanas contaminadas hacia los sistemas acuáticos y el transporte de radionucleidos en ríos.

## SAMENVATTING

Deze rapportage is samengesteld uit een aantal artikelen geschreven door de deelnemers van het project MOIRA ("A MODEL BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING RADIONUCLIDE CONTAMINATED AQUATIC ECOSYSTEMS AND DRAINAGE AREAS").

Het doel van het project is het beschrijven en analyseren van rekenmethodes voor de identificatie van tegenmaatregelen in met radioactiviteit verontreinigde zoetwater-systemen. De structuur van de MOIRA software, en de criteria voor een objectieve evaluatie van de ecologische impact van de tegenmaatregelen voor meren (gedefinieerd door de "Lake ecosystem index", die wordt toegepast in de "Multi-attribute Decision Analysis" procedure die voor MOIRA ontwikkeld is) zijn in detail beschreven.

Een voorlopige classificatie van de tegenmaatregelen gerelateerd aan de mogelijke effectiviteit en haalbaarheid, en een classificatie van de voor consumptie belangrijkste vissoorten in Europa, worden in dit rapport gepresenteerd. Ter illustratie van de toepasbaarheid van MOIRA, is de beschreven methode toegepast voor Øvre Heimdalsvatn, een meer in Noorwegen. In twee andere artikelen van het rapport worden de modelprincipes voor het transport van radionucliden vanuit stedelijke gebieden (via de afwateringssystemen) naar het oppervlaktewater en voor het transport van radionucliden in rivieren beschreven.

THIRD MOIRA MEETING, ROME, ITALY, 29-31, MAY 1997

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# FOREWORD

MOIRA (A MODEL-BASED COMPUTERISED SYSTEM FOR MANAGEMENT SUPPORT TO IDENTIFY OPTIMAL REMEDIAL STRATEGIES FOR RESTORING RADIONUCLIDE CONTAMINATED AQUATIC ECOSYSTEMS AND DRAINAGE AREAS) is a project financed by the European Commission (Contract N° FI4P-CT96-0036) within 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 environments. The following Institutions participate in the MOIRA project: ENEA (Italy), KEMA (The Netherlands), University of Oslo (Norway), UPM (Universidad Politecnica de Madrid, Spain), Studsvik Eco & Safety AB and University of Uppsala (Sweden). The present report was prepared during meetings held in Rome, 29-31 May, 1997 and in Arnhem, 3-6 December, 1997

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## PREFACE

The main aim of the present report is the description of the principles for the development and the implementation of the MOIRA computerised system. The MOIRA software system is intended to be a user-friendly tool for decision makers that are responsible for the application of optimal restoration strategies to contaminated fresh water systems. Obviously, the user of such a software system is not necessarily an expert in environmental modelling.
Therefore, MOIRA software must be synthetic, accessible, flexible and practical (Appelgren et al. 1996). It is of primary importance to develop a suitable software framework, which is designed to encompass the mathematical simulation models, the preparation and the management of the available input data and the user interface, to achieve such features.
One of the article presented here describes, in detail, the principles for the design of the MOIRA software framework which has being developed according to the most effective, suitable and updated techniques for the realisation of reliable and user-friendly computer codes.
As emphasised in the first MOIRA report (Appelgren et al. 1996), the primary need of a decision support system for choosing optimal remedial strategies in the aquatic environment is the prediction of radionuclide behaviour in fresh water ecosystems. In the present report the principles for developing models for predicting radionuclide migration in rivers and in urban drainage systems are described.
Two of the papers published in this third MOIRA report are of paramount importance for the achievement of the MOIRA goals:

a) The articles on the definition and application of the Lake Ecosystem Index (LEI) concept;
b) The case study based on the application of the Multiattribute Analysis criteria developed for the identification of optimal remedial countermeasures to the Norwegian lake Heimdalsvatn.

Indeed, these articles clearly demonstrate the feasibility and the applicability of the MOIRA methodologies to real cases.
The Lake Ecosystem Index is a holistic parameter which reflects the ecological status of the lake. It may be used to rank the countermeasures according to their impacts on the environment.
The present report also provides tables on the classification of European fish species of radioecological importance. Such tables are an initial attempt to put the fish species into a logical framework that can be incorporated into the MOIRA system.
A review of potential countermeasures in aquatic ecosystems is given. The countermeasures have been classified according to their approach (chemical, physical, biological, etc.), their mode of action (reduction of radionuclide runoff, adsorption of radionuclides, reduction of secondary load, etc.) and the water body where they may be applied. The present state of our knowledge and experience is also reported for each countermeasures. Such a review is an important help in evaluating the feasibility of the most important countermeasures for the restoration of radionuclide contaminated fresh water ecosystems.

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# REVIEW OF POTENTIAL COUNTERMEASURES IN AQUATIC ECOSYSTEMS AND THEIR DRAINAGE AREAS

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In the event of radioactive contamination of aquatic ecosystems and their drainage areas there are a number of options available to management. The development of a model-based computerised system for management support in order to select the optimal action is the main objective of the MOIRA project. The options are wide-ranging, from specific chemical treatment of water bodies to fishing bans and even the movement of communities. The potential actions can be broadly grouped into three main categories, chemical, physical and social. In some cases a combination of actions may be the optimal strategy. A further option is of course not to take any remedial actions, although this may also have significant socio-economic repercussions.

Most of the countermeasures are applicable to all types of aquatic ecosystems, although their efficiency may vary considerably, depending on the sensitivity of the system to radioactive fallout (Håkanson et al. 1996, IAEA- in press). For example, many of the chemical measures will be most effective in lakes with long water renewal times and correspondingly ineffective in rivers or coastal areas with rapid renewal of the water masses. The production level of the contaminated ecosystem will also influence the effectiveness or otherwise of chemical countermeasures. For instance, the fertilisation of a eutrophic lake would be ineffective as a countermeasure. Addition of fertilisers would also increase the risk of undesirable eutrophication effects in lakes near the transition between different productivity states.

In a large-scale study of Swedish lakes, different chemical remedial measures against radiocaesium were experimentally studied (Håkanson & Andersson 1992). The measures studied were lake liming, drainage area liming, lake liming plus selenium treatment and lake liming plus potash treatment. The measures generally gave the intended response in water chemistry. However, there was no rapid and clear reduction in radiocaesium concentrations in fish. Nevertheless, a reduction in the ecological half-life of radiocaesium was apparent, which can be important in a long-term perspective.

In managed aquatic ecosystems, such as regulated rivers, which easily allow the increase or decrease of water renewal and flow-though rates, the transport of radionuclides can be considerably modified. For example, the opening of dams in reservoir cascades can effectively increase downstream transport of radionuclides, although a knowledge of stratification conditions is a prerequisite (Voitsekhovitch 1996). Removal of contaminated aquatic sediments may also be facilitated by the opportunity to lower water levels in for example reservoirs. The use of sediment traps in the Pripyat River after the Chernobyl accident proved ineffective (Voitsekhovitch 1996), but may be more useful in less turbulent rivers, reservoirs and lakes.

Runoff from catchments, especially during floods caused by snowmelt or heavy rainfall events, can give rise to increased radionuclide transfer to aquatic ecosystems, especially from contaminated river floodplains. As well as being important in the initial phase of the contamination, this can be particularly important as a long-term source of radionuclide contamination. In such situations, the application of ecological engineering concepts can be a countermeasure option. For example, the building of dykes between a river and its floodplain may prevent runoff of contaminants, especially during the spring floods when much of the ground in the catchment may be frozen and runoff rates are high (Voitsekhovitch 1996). Note should be taken of the fact that snowmelt or heavy rainfall in the upper parts of the catchment, perhaps a considerable distance away, may frequently gives rise to flooding in the contaminated lower parts and thus cause remediation problems. The removal of grazing domestic animals from contaminated floodplains may be an alternative remedial strategy to reduce the dose to man.

In areas with cold climates, any radionuclide deposition during winter will remain in the snow pack until the spring. This provides the opportunity to remove contaminated snow and ice from lakes and rivers, thereby reducing the primary load on the aquatic ecosystem. It may also be prudent to remove ice jams that will give rise to flooding of contaminated floodplains and other land areas.

Lake and river sediments often constitute a major sink for radionuclides in aquatic systems. The availability of such radionuclides will depend on several factors, of which the sediment $K_d$ and the sedimentation rate are important. For example, if sedimentation rates are high, the contaminated sediments may be rapidly covered by non-contaminated sediments, rendering them unavailable. However, in turbulent rivers and wind-exposed shallow lakes, resuspension of radionuclide contaminated sediments can represent a major long-term source of radionuclides for the aquatic ecosystem. It is possible to remove radionuclide-contaminated sediments by dredging, although care must be taken to prevent their remobilization and transport downstream.

In areas where fish constitute a significant part of people's diet, restrictions or bans on fishing may be an effective countermeasure. In certain circumstances, an alternative to, or in association with restrictions to fishing, are recommendations for food preparation. A significant reduction in radionuclide content can be achieved by certain methods of food storage and preparation. The same applies to drinking water, which may be treated by radionuclide sorbents (Remez 1996).

In cases of severe contamination of aquatic systems and their catchments, it may be imperative to restrict people's access to and utilisation of certain areas, as in the case for much of the Techna River. It may also be necessary to even relocate whole communities, as in the upper and middle reaches of the Techna River and after the Chernobyl accident. In such situations, the use of alternative water sources both for drinking water and irrigation can also significantly reduce the dose to man (Berkovsky et al. 1994, Romanov & Drozhko 1996).

Certain countermeasures are site specific data with regard to the costs involved and the actions that are feasible. These are indicated in the table. They include wetland liming, decreased catchment runoff and control of water flow through rates. Without further information, such countermeasures cannot be costed by MOIRA, although a subjective cost estimate can be provided. However, the user will undoubtedly posses the required site-specific data.

## Countermeasures and their mode of action

| APPROACH | ACTION | LOCATION OF ACTION | MODE OF ACTION | EXPERIENCE, MODELS & DATA |
|---|---|---|---|---|
| Chemical | Potash treatment | Lakes, reservoirs, rivers, coastal | Chemical "dilution" | Some experience and data for lakes. Models available. |
| Chemical | Direct liming | Lakes, reservoirs, rivers, coastal | Changes pH, which influences for example biouptake | Considerable experience in relation to acidification. Models available. |
| Chemical | Wetland liming | Catchment | Reduction in radionuclide runoff | Site-specific. Considerable experience in relation to acidification. Models available. |
| Chemical | Fertilisation | Lakes, reservoirs, rivers, coastal | Increases biomass, "biological dilution" | Considerable experience in relation to eutrophication. Models available. |
| Chemical | Adsorption dykes | Rivers, reservoirs, catchment | Chemical adsorption of radionuclides (e.g. zeolite). Also prevention of floodplain contamination | Site-specific. Some experience after Chernobyl, but ineffective. Can be modelled. |
| Biological | Fishing (removal of shellfish) | Lakes, reservoirs (coastal) | Removal of fish (shellfish) biomass/changes in ecosystem structure | Considerable experience for economic species. Models available. |
| Biological | Fish removal | Lakes, reservoirs (coastal) | Removal of fish biomass using fish poisons such as rotenone | Some experience. Models available. |
| Physical | Modification of catchment runoff | Catchment | Reduced secondary load; build flood dykes, etc. | Site-specific. Some experience and models. Limited success after Chernobyl. |
| Physical | Decontaminat -ion of urban runoff | Catchment | Reduced secondary load | Site-specific. Considerable experience and models available. |
| Physical | Control water flow through rate | Lakes, reservoirs, rivers, polders | Change water retention time; open dams, fill reservoir, etc. | Site-specific. Limited experience, limited success. Models available. |
| Physico-chemical | Removal of contaminated sediments | Lakes, reservoirs, rivers, coastal | Reduction in active sediment layer and/or direct exposure to man | Little experience. Can be modelled. |
| Physico-chemical | Sediment traps | Lakes, reservoirs, rivers | Collection of radionuclides associated with particles | Tried after Chernobyl but unsuccessful. Can be modelled. |

| | | | | |
|---|---|---|---|---|
| Physico-chemical | Removal of contaminated snow and ice | Lakes, reservoirs, catchments, rivers, coastal | Reduction in source term and/or direct exposure to man | Site-specific. No experience, but can be modelled. |
| Physico-chemical | Treatment of drinking water | Lakes, reservoirs, rivers | Reduction in dose from drinking water | Some experience after Chernobyl. Can be modelled. |
| Chemical/ social | Food preparation | All ecosystems | Reduction in dose through food | Some experience. Can be very effective. |
| Social | Bans on fish consumption | Lakes, reservoirs, rivers, coastal | Reduction in dose through food | Some experience. Can be effective. Models available. |
| Social | Alternative drinking water sources, e.g. groundwater | Lakes, reservoirs, rivers | Reduction in dose from drinking water | Site-specific. Some experience. Effective. Can be modelled. |
| Social | Dietary changes (e.g. use of aquaculture where non-contaminated food is given) | All ecosystems | Reduction in dose through food | Some experience, can be effective. Can be modelled. |
| Social | Irrigation bans/ restrictions | Lakes, reservoirs, rivers | Reduction of uptake in crops | Some experience, can be effective. Can be modelled. |
| Social | Restricted areas | All ecosystems | Reduction in dose to population | Site-specific. Some experience; can be effective. Can be modelled. |

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# EUROPEAN FISH SPECIES OF RADIOECOLOGICAL IMPORTANCE

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Fish are a major food source in aquatic ecosystems, both in inland and coastal waters. Contamination of aquatic ecosystems by radionuclides can therefore give rise to significant doses to man. Thus, among other things, MOIRA aims to predict radionuclide concentrations in specific target organisms. It is therefore necessary to choose certain key fish species. Such species must fulfil a number of criteria. Firstly, they should be widespread in all or parts of Europe. Secondly, they should be of major importance for human consumption. There are also certain other fish species, which although they are not consumed by man, are either important prey species or have a central role in ecosystem dynamics. Examples are the roach and the minnow. In European coastal areas shellfish are also a potentially important source of dose to man and they also should be included in the MOIRA system.

After the initial fry stage, many fish species occupy a typical food niche. They may either feed on plankton (planktivore), benthos (benthivore), or fish (piscivore). However, many species change their habitat and/or diet during their life, often in relation to body size. For example, perch, a widespread and abundant European fish species, begins life as a planktivore, consuming zooplankton. As it grows it changes to be a benthivore, while larger individuals consume fish. Pike also undergo similar changes, but it is only a matter of weeks before the pike becomes an obligate piscivore. In contrast, only certain strains of brown trout become piscivorous. Another example, of such a change is the Atlantic salmon, which in addition to changing its diet, also migrates from freshwater to the marine environment and back again during its life cycle.

The following tables are an initial attempt to put the fish species into a logical framework that can be incorporated into the MOIRA system. It will undoubtedly need to be modified with regard to the radioecological data available on each particular species and also as a result of internal discussions within the MOIRA project as to the most appropriate species.

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## Rivers, lakes and reservoirs

| Species | Typical weight range kg | Target weight kg | Trophic range | Habitat | Food habit | Significance |
|---|---|---|---|---|---|---|
| PIKE | 0.5-3 | 1 | oligotrophic -eutrophic | benthic, littoral | piscivore | consumed by man |
| PIKE-PERCH | 0.5-3 | 1 | oligotrophic -eutrophic | pelagic | piscivore | consumed by man |
| PERCH - stage I | 0.01-0.1 | 0.1 | oligotrophic -eutrophic | benthic, littoral | planktivore | early stage of consumed fish |
| PERCH - stage II | 0.1-0.3 | 0.2 | oligotrophic -eutrophic | benthic, littoral | benthivore | consumed by man |
| PERCH - stage III | 0.3-0.6 | 0.5 | oligotrophic -eutrophic | benthic, littoral | piscivore | consumed by man |
| BROWN TROUT - stage I | 0.05-0.1 | 0.1 | oligotrophic - mesotrophic | benthic, littoral | planktivore | early stage of consumed fish |
| BROWN TROUT II | 0.1-0.3 | 0.2 | oligotrophic - mesotrophic | benthic, littoral | benthivore | consumed by man |
| BROWN TROUT - stage III | 0.5-1.5 | 1 | oligotrophic - mesotrophic | benthic, littoral | piscivore | consumed by man |
| ARCTIC CHARR | 0.01-0.2 | 0.1 | oligotrophic - mesotrophic | pelagic | planktivore | consumed |
| MINNOW | 0.001-0.01 | 0.01 | oligotrophic - mesotrophic | benthic, littoral | omnivore: benthos, algae | prey species |
| SMELT | 0.01-0.05 | 0.01 | oligotrophic-eutrophic | pelagic | planktivore | prey species |
| WHITEFISH | 0.1-1 | 0.3 | oligotrophic - mesotrophic | pelagic | planktivore | consumed by man |
| BURBOT | 0.1-1 | 0.5 | oligotrophic | benthic, profundal | benthivore | consumed by man |
| ROACH | 0.05-0.2 | 0.1 | oligotrophic-hypertrophic | benthic/ pelagic, littoral | omnivore: | prey species |
| RUFFE | 0.005-0.02 | 0.01 | oligotrophic -eutrophic | benthic, littoral/ profundal | benthivore | prey species |
| BREAM | 0.5-2 | 1 | mesotrophic - eutrophic | benthic, littoral | benthivore, detritivore | consumed by man |
| CARP | 0.5-3 | 1 | mesotrophic - eutrophic | benthic, littoral | benthivore, detritivore | consumed by man |
| EEL | 0.1-1 | 0.5 | oligotrophic-eutrophic | benthic | omnivore: benthivore, piscivore | consumed by man |

## Coastal areas

| Species | Typical weight range kg | Target weight kg | Trophic range | Habitat | Food habit | Significance |
|---|---|---|---|---|---|---|
| PIKE | 0.5-3 | 1 | oligotrophic - eutrophic | benthic, littoral | piscivore | consumed by man |
| PERCH - stage I | 0.01-0.1 | 0.1 | oligotrophic - eutrophic | benthic, littoral | planktivore | early stage of consumed fish |
| PERCH - stage II | 0.1-0.3 | 0.2 | oligotrophic - eutrophic | benthic, littoral | benthivore | consumed by man |
| PERCH - stage III | 0.3-0.6 | 0.5 | oligotrophic - eutrophic | benthic, littoral | piscivore | consumed by man |
| SEA TROUT - stage I | 0.05-0.1 | 0.1 | available spawning areas limit distribution | benthic, littoral | planktivore | early stage of consumed fish |
| SEA TROUT - stage II | 0.1-0.5 | 0.2 | available spawning areas limit distribution | benthic, littoral | benthivore | early stage of consumed fish |
| SEA TROUT - stage III | 0.5-1.5 | 1 | available spawning areas limit distribution | benthic, littoral | piscivore | consumed by man |
| ATLANTIC SALMON - stage I | 0.05-0.1 | 0.05 | available spawning areas limit distribution | pelagic | planktivore | early stage of consumed fish |
| ATLANTIC SALMON - stage II | 1-4 | 2 | available spawning areas limit distribution | pelagic | piscivore | consumed by man |
| COD | 1-3 | 1 | oligotrophic - mesotrophic | benthic | benthivore | consumed by man |
| HADDOCK | 1-3 | 1 | oligotrophic - mesotrophic | benthic | benthivore | consumed by man |
| WHITEFISH | 0.1-1 | 0.3 | oligotrophic - mesotrophic | pelagic | planktivore | consumed by man |
| ROACH | 0.05-0.2 | 0.1 | oligotrophic-hypertrophic | benthic/ pelagic, littoral | omnivore | prey species |
| SCULPIN | 0.01-0.05 | 0.04 | oligotrophic-eutrophic | benthic | omnivore | prey species |
| HERRING | 0.05-0.2 | 0.1 | oligotrophic – mesotrophic | pelagic | planktivore | consumed by man/ prey species |
| ANCHOVY | 0.01-0.05 | 0.05 | oligotrophic – eutrophic | pelagic | planktivore | consumed by man/prey species |
| PLAICE | 0.1-1 | 0.5 | oligotrophic – mesotrophic | benthic | benthivore | consumed by man |
| EEL | 0.1-1 | 0.5 | oligotrophic – eutrophic | benthic | omnivore: benthos, fish | consumed by man |

# A CASE STUDY OF MULTIATTRIBUTE DECISION ANALYSIS FOR RANKING COUNTERMEASURES IN THE MOIRA PROJECT

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## INTRODUCTION

For ranking the alternative strategies, a Multiattribute Decision Analysis (MDA) methodology has been designed *(Ríos-Insua et al., 1997)*, which basically includes five steps. First, the identification of the contaminated site, supplying all relevant geographical and ecological information. Then, the construction of a hierarchy of objectives, with attributes for lowest level objectives. Using the GIS and the various models of the system, the feasible countermeasures are identified, together with their impacts. Preferences for each consequence are then modelled with component utility functions, which are finally aggregated via a weighted sum, which allows ranking the alternatives. As an additional step, multiparametric sensitivity analyses (both with respect to weights and values) provide insights into the problem.
An application of the MDA methodology to a real case is described, with the intention of illustrating and testing the methodology through a practical example which, although limited, can demonstrate the possibilities of the MOIRA system.

## A REAL CASE BASED SCENARIO

The site chosen for testing the MDA methodology of MOIRA is the lake Øvre Heimdalsvatn located in the county of Oppland, in Norway. It is a small subalpine lake with a mean depth of 4.7 m, maximum depth 13 m, a surface area of 0.78 km$^2$, and a catchment area of 23.6 km$^2$. The highest point of the catchment is 1843 m a.s.l., while the lake itself is at 1090 m a.s.l. The mean annual precipitation is 800 mm. This lake has been thoroughly studied and a very complete information can be found in *Vik (1978).*

There is no permanent settlement within the catchment, but a scientific field station is located by the lake, with an occupation of about 600 man-days per year. During the summer, a herdsman looks after calves and sheep and there is some fishing activity, mainly for recreation. Scientists, heardsman and tourist/anglers use the lake for drinking water, but no other uses are made for irrigation or supply of drinkable water to towns or villages.

After the Chernobyl accident, the lake was contaminated with a fallout of 130 kBq/m$^2$ of $^{137}$Cs, which, in principle, required no countermeasures. Also, the low utilization of the lake by people makes individual and collective doses very low, and any countermeasure result would not be cost-effective, since the doses that could be averted are always very low. However, an evaluation of some alternative strategies can be made with the objective of testing the MOIRA system and, in particular the MDA methodology.

*Objectives and attributes of the analysis.*

Once the contaminated site is defined and characterised, the second step of the methodology is the construction of an objectives tree, which, for this specific case, can be a simplification of the more general hierarchy tree proposed for MOIRA *(Ríos-Insua et al., 1997a).*

Obviously, the three general objectives of minimising the environmental, social and economic impact can be maintained for any given scenario for which MOIRA can be applied. However, taking into account the peculiarities of every case, the general objectives can be split in to more or less branches of sub-objectives, with attributes for the lowest level of the tree. For the particular case studied here, the hierarchy tree of objectives is given in Figure 1.

The process was essentially creative and alterations were made after discussions to arrive at a reasonably comprehensive set of objectives. The next step involved specifying attributes to measure the degree to which these objectives were met.

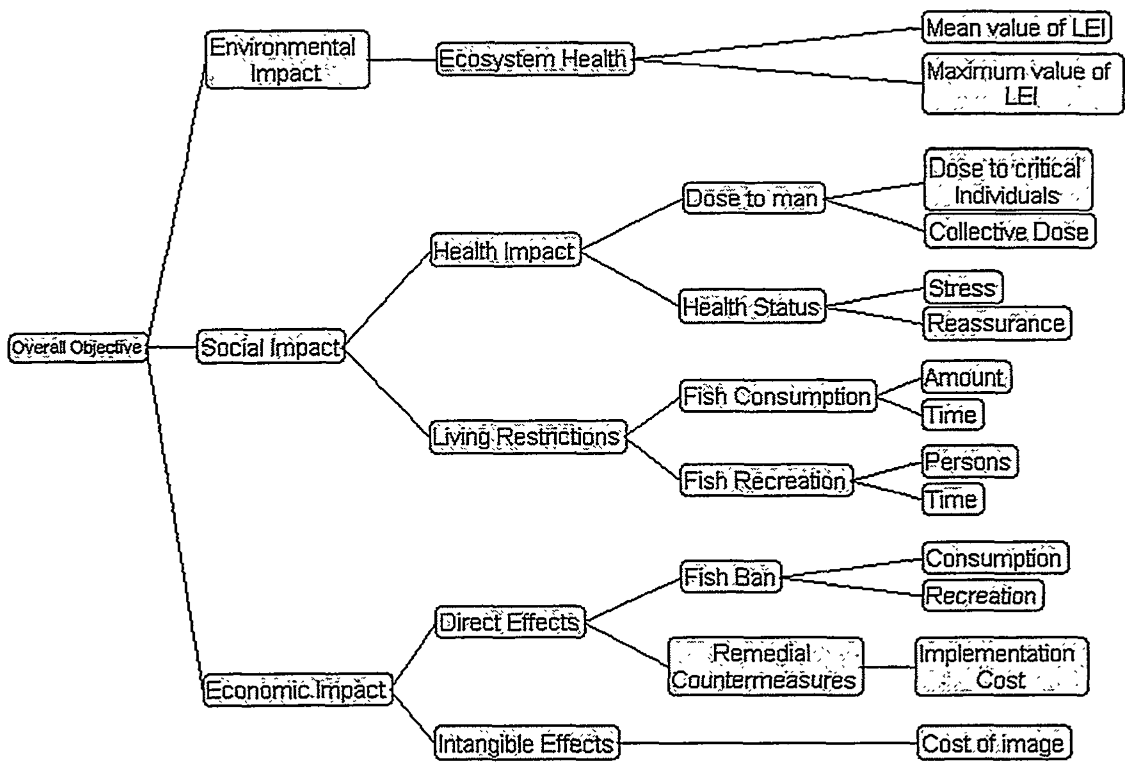


*Figure 1. - Reduced hierarchy tree of objectives for the case studied.*

In the case of the lake Øvre Heimdalsvatn, given the low level of contamination, there will be no threats to biota, and the environmental impact can be identified with the ecosystem health, which can be measured by the so-called *Lake Ecosystem Index, LEI (Håkanson, 1993, 1997)*, from which we will take two attributes: the mean improvement during the evaluated period and the maximum improvement at any given time.

With respect to the social impact, there are two important objectives: minimising impact on health and on living conditions. Radiation dose to man is an obvious factor affecting health, with dose to critical individuals -which should never receive levels above thresholds for early health effects- and collective dose which induces a linear increase in the risk of developing serious latent effects, mainly cancer. Overall health status of people can also be affected by countermeasures, due to less specific reassurance and stressing effects. In the "living restrictions" branch, the only action considered effective for this case is a fish ban, which can reduce the dose by ingestion. Attributes for it can be the duration of ban, the amount of fish banned and the number of persons disturbed, in this case because of the prohibition of accessing the lake for fishing.

Finally, the economic impact can be divided into intangible effects, linked to the loss-of-image and adverse market reactions for the affected area, and direct effects, more amenable to a quantification, like those associated with a fish ban or to the application of remedial chemical countermeasures. Components of the economic impact of a fish ban will be the cost of the fish banned for consumption and the subjective cost of the recreation lost. For chemical countermeasures, the cost of implementing can be well known and is a useful attribute.

*Table 1.- Objectives hierarchy with levels and attributes for evaluation of countermeasures.*

| LEVEL | | | | Measure | Value | |
|---|---|---|---|---|---|---|
| **Four** | **Three** | **Two** | **One: Attributes** | **(Units)** | **Worst** | **Best** |
| Environmental Impact | Ecosystem Health | | $X_1$: average improvement LEI | % LEI | 0 | 21.2 |
| | | | $X_2$: maximum improvement LEI | %LEI | 0 | 80.8 |
| Social impact | Health impact | Dose to man | $X_3$: dose to critical individuals | mSv | 2.47 | 1.86 |
| | | | $X_4$: collective dose | mSv x man | 72.30 | 54.00 |
| | | Health status | $X_5$: stress effects | Constructed Scales | 100 | 0 |
| | | | $X_6$: reassurance effects | | 0 | 100 |
| | Living restrictions | Fish consumption | $X_7$: amount of fish banned | kg | 445 | 0 |
| | | | $X_8$: duration of fish ban | months | 12 | 0 |
| | | Fish recreation | $X_9$: persons affected by fish recreation ban | No. of persons | 30 | 0 |
| | | | $X_{10}$: duration of fish recreation ban | months | 12 | 0 |
| Economic impact | Direct effects | Fish ban | $X_{11}$: cost of fish consumption ban | kSEK | 22.25 | 0.00 |
| | | | $X_{12}$: cost of fish recreation ban | kSEK | 120.00 | 0.00 |
| | | Remedial countermeasures | $X_{13}$: implementation cost | kSEK | 702.00 | 0.00 |
| | Intangible effects | | $X_{14}$: cost of image | Constructed scale | 100 | 0 |

The attributes for each objective of the lowest level will be used as a measure of the effectiveness of each strategy. They can be measured either in "natural" or "constructed" scales. Natural scales are always preferred for objectives for which the system incorporates models or data available on which the predictions of impact for each strategy can be based. On the other hand, some objectives are of a more imprecise or intangible nature, and the evaluation of scores for each strategy will be directly assigned by the experts on a constructed scale, normally qualitative or ranging between, for example, 0 and 100. The list of final attributes for evaluating countermeasures in the present case is shown in Table 1, which also includes their units and the range of variation obtained from the alternatives analysed below.

*Table 2. - Matrix of alternative strategies analysed and their impacts*

| | | Attributes[1] | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Strategy | Description | $X_1$ | $X_2$ | $X_3$ | $X_4$ | $X_5$ | $X_6$ | $X_7$ | $X_8$ | $X_9$ | $X_{10}$ | $X_{11}$ | $X_{12}$ | $X_{13}$ | $X_{14}$ |
| $S_1$ | No action | 0 | 0 | 2.47 | 72.3 | 50 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 100 |
| $S_2$ | Fish banning | 0 | 0 | 2.18 | 63.7 | 0 | 50 | 445 | 12 | 30 | 12 | 22 | 120 | 0 | 75 |
| $S_3$ | Lake liming | 17.5 | 80.8 | 2.34 | 68.3 | 100 | 50 | 0 | 0 | 0 | 0 | 0 | 0 | 141 | 50 |
| $S_4$ | Liming+ fish ban | 17.5 | 80.0 | 2.1 | 61.3 | 0 | 100 | 445 | 12 | 30 | 12 | 22 | 120 | 141 | 25 |
| $S_5$ | Potash treatment | 15.7 | 17.2 | 2.04 | 59.5 | 100 | 50 | 0 | 0 | 0 | 0 | 0 | 0 | 702 | 50 |
| $S_6$ | Potash + fish ban | 15.7 | 17.2 | 1.86 | 54 | 0 | 100 | 445 | 12 | 30 | 12 | 22 | 120 | 702 | 25 |
| $S_7$ | Fertilising | 21.2 | 59.2 | 2.45 | 71.5 | 100 | 50 | 0 | 0 | 0 | 0 | 0 | 0 | 125 | 50 |
| $S_8$ | Fertilising + fish ban | 21.2 | 59.2 | 2.16 | 63 | 0 | 100 | 445 | 12 | 30 | 12 | 22 | 120 | 125 | 25 |

[1]For a description of each attribute and its units, see Table 1.

### *Alternatives and impacts*

The next step of the decision process is to identify feasible alternative strategies, including the no-action option, and describe their impacts in terms of each attribute. With that objective, a set of eight strategies has been analysed, combining chemical countermeasures with fish banning in order to reduce the radiological and environmental impact. They are listed in Table 2, together with their impacts. The basic countermeasures tested are:

- A total ban on fish consumption and capture during the first year after the accident (the fishing season extends from June to October and the lake was contaminated at the beginning of May).
- Lake liming treatment applied at the beginning of July each year, with 40 tonnes the first year and 25 tonnes the five following years.

- Potash treatment applied at the beginning of July each year, with 40 tonnes the first year and 25 tonnes the five following years
- Fertilisation of the lake with 0.1 tonne of phosphorus every month from May to October all the six years.

The matrix in Table 2 summarises the results of running the different MOIRA sub-models for each strategy with the data and assumptions for the calculations which are described in Annex I:

- The MOIRA lake model *(Håkanson et al., 1997)*, to obtain the time variation of the concentrations of $^{137}Cs$ in the water of the lake and in fish as a function of time. The lake model includes sub-models for the countermeasures analysed *(liming; potash treatment; fertilising)* as well as a model for assessing the Lake Ecosystem Index *(Håkanson, 1993, 1997)*.
- The MOIRA dose model *(Jiménez and Gallego, 1997)* to obtain individual and collective doses for each strategy.

For attributes number 5 (stress effects), 6 (reassurance effects) and 14 (cost of image), the values were assigned by direct guess of the team members making the assessment. The social and economic impacts of a fish ban for 1 year (attributes 7 -amount of fish-, 9 -persons affected-, 11 -cost of fish consumption lost- and 12 -cost of recreation lost-) were externally quantified based on information about the site (database information). The cost of the implementation of chemical countermeasures was obtained directly from database on past experience *(Håkanson, 1997a)*.
It is worthwhile extracting some results from the calculations performed. Figure 2 reflects the results of running the lake model for the natural evolution of the lake, no action alternative, and the three chemical countermeasures being tested. As can be observed, the concentration of $^{137}Cs$ in water is very insensitive to the different treatments, which mainly affect the remobilisation from sediments (lake liming), the uptake of cesium by aquatic organisms (potash treatment), or which produce a "biological dilution" (fertilising). Consequently, and looking to the low contamination of the water, no action on direct uses of water has been considered useful. On the other hand, the impact on the LEI indicates that, while the continuous fertilisation during summer

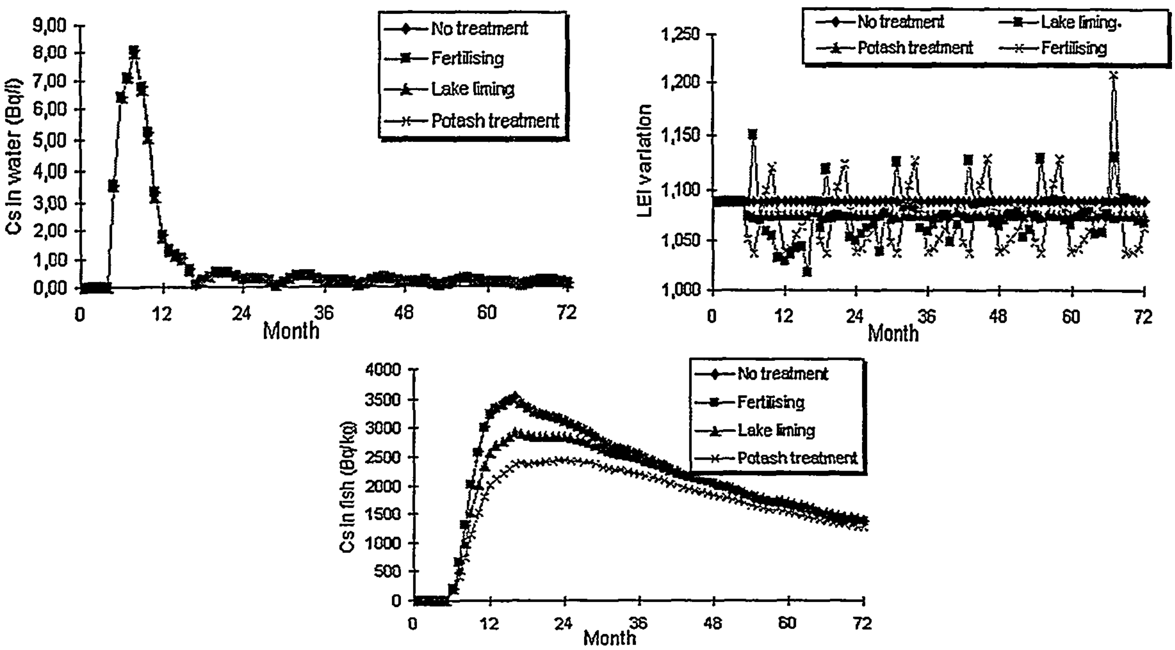


*Figure 2.- Results of the lake model for four different alternatives.*

results in a very stable condition, the annual treatments (with lime and potash) produce yearly fluctuations of the LEI, from which the mean and the maximum improvement (variations towards the ideal value 1.0) have been considered as attributes (level one objectives). Finally, the simulation suggest a good performance for potash in reducing cesium in fish, followed by a lower efficiency of liming, and a negligible effect due to fertilising.

With respect to the results of the dose model, it is worth noting that the ingestion dose from fish represents more than 99% of the total dose for this scenario. Since high values of $^{137}Cs$ in fish are observed for several years, a ban on fish lasting only for the first year will be not very efficient, as the results of the dose model suggest (see Table 2, attributes $X_3$ - maximum individual dose in mSv, and $X_4$ - collective dose in mSv x man). This low efficiency will be reflected in the resulting ranking of countermeasures.

### *RANKING THE ALTERNATIVES*

To conduct the multiattribute analysis, we determine a utility function, which combines the multiple evaluation measures into a single measure or utility of each strategy to provide a ranking. The form of this function used here, which we consider a valid approximation (see *Raiffa, 1982; Stewart, 1996*), is an additive utility function

$$u(S_l) = w_1 u_1(x_1) + w_2 u_2(x_2) + \ldots + w_{14} u_{14}(x_{14})$$

where $x_i$ is a specific level of attribute $X_i$. Thus, to determine such function we need to specify:

- Single dimensional utility functions $u_i$ for each evaluation measure (attribute).
- Weights or scaling constants $w_i$ for each single dimensional utility function

The assessment of a utility function is a process in which the analyst asks the decision-makers a series of questions about his or her preferences. From the responses, the analyst constructs the decision-makers' utility function $u_i$. Then, the individual functions $u_i$ will be aggregated in the form of the equation above, where again, from questions about preferences the weights are assigned. Next, we will see the procedures to carry out the above determinations.

#### *Elicitation of utilities*

The method to determine single dimensional utility functions is based on the combination of two standard procedures. The details of methodology can be found elsewhere *(Ríos-Insua et al., 1997)*. Several authors (*e.g., Hershey et al., 1982, McCord and de Neufville, 1986, Jaffray, 1989)* have agreed that, generally, constructed utility functions are method dependent and bias and inconsistencies may be generated in the elicitation process. To avoid such problems in the MOIRA system, we propose using two methods jointly: the probability equivalent method (PE) and the certainty equivalent method (CE). Furthermore, instead of assessing only one number in probability questions, we have assessed a class of utility functions. This is less demanding because we ask the decision-makers to provide only incomplete preference statements providing intervals instead of unique numbers and allowing easily performed consistency checks.

With this procedure we obtain ranges of utility functions for each method instead of just one utility function for each one. Thus, the intersection of both ranges gives the range where preferences elicited by the two above methods are consistent. Hence, if such intersection was empty for some intervals, the decision-makers would be inconsistent and they should re-evaluate his assessment. The process ends when a consistent range is provided. It is interesting to remark that the resulting ranges in the utility functions can be used later in sensitivity analysis to gain insight in the ranking of the strategies and help the decision-makers to discard certain strategies, which may not never be good. This will be an additional function of the system, not shown in this case.

The next step is obtaining the component utility functions $u_i$. To obtain such functions, we have fitted piecewise exponential functions of the form, $a + b\, e^{-cx}$, using minimum least squares to the midpoints of each range of the class of utility functions, obtained as the intersection of both ranges. Table 3 shows all the fitted component utility functions for all the attributes

*Table 3.- Proposal for the single-attribute utility functions.*

| Attributes | $u_i$ | Range |
|---|---|---|
| $X_1$: average improvement LEI | $u_1(x_1) = 1.53 - 1.53 \exp(-.04998\ x_1)$ | [0, 21.2] |
| $X_2$: maximum improvement LEI | $u_2(x_2) = 1.198 - 1.198 \exp(-.0222\ x_2)$ | [0, 80.8] |
| $X_3$: dose to critical individuals (µSv) | $u_3(x_3) = .3847 + .000602 \exp(-.006179\ x_3)$ | [1860, 2200] |
| | $u_3(x_3) = .5876 - 5.027.10^{-7} \exp(.00565\ x_3)$ | [2200, 2470] |
| $X_4$: collective dose | $u_4(x_4) = -2.015 + 10.01 \exp(-.0222\ x_4)$ | [54, 72.3] |
| $X_5$: stress effects | $u_5(x_5) = -1.192 + 2.192 \exp(-.00609\ x_5)$ | [0, 100] |
| $X_6$: reassurance effects | $u_6(x_6) = 1.784 - .784 \exp(.00822\ x_6)$ | [0, 100] |
| $X_7$: amount of fish banned | $u_7(x_7) = .2477 + .7523 \exp(-.00464\ x_7)$ | [0, 225] |
| | $u_7(x_7) = .9143 - 1.733 \exp(.003737\ x_7)$ | [225, 445] |
| $X_8$: duration of fish ban | $u_8(x_8) = 1.516 - .5161 \exp(.1175\ x_8)$ | [0, 7] |
| | $u_8(x_8) = -.4072 + 1.757 \exp(-.1218\ x_8)$ | [7, 12] |
| $X_9$: persons affected by fish recreation ban | $u_9(x_9) = .1232 + .8768 \exp(-.04778\ x_9)$ | [0, 12.75] |
| | $u_9(x_9) = 1.809 - .8976 \exp(.02336\ x_9)$ | [12.75, 30] |
| $X_{10}$: duration of fish recreation ban | $u_{10}(x_{10}) = 1.397 - .3973 \exp(.1523\ x_{10})$ | [0, 4] |
| | $u_{10}(x_{10}) = 1.349 + 2.464 \exp(-0502\ x_{10})$ | [4, 12] |
| $X_{11}$: cost of fish consumption ban | $u_{11}(x_{11}) = -2.015 + 3.015 \exp(-.01811\ x_{11})$ | [0, 22.25] |
| $X_{12}$: cost of fish recreation ban | $u_{12}(x_{12}) = 1.225 - .2252 \exp(.01826\ x_{12})$ | [0, 70] |
| | $u_{12}(x_{12}) = -.1608 + 3.459 \exp(-.02557\ x_{12})$ | [70, 120] |
| $X_{13}$: implementation cost | $u_{13}(x_{13}) = 2.192 - 1.192 \exp(.000867\ x_{13})$ | [0, 702] |
| $X_{14}$: cost of image | $u_{14}(x_{14}) = 5.508 - 4.508 \exp(.002003\ x_{14})$ | [0, 100] |

*Determining the weights*

As explained above, to obtain the global utility function, it is also necessary to assess the weights or scaling factors for each attribute at the lowest-level of the objectives hierarchy. Furthermore, it would be useful to assign weights to higher level objectives to make possible global sensitivity analyses over any one of the objectives.

The procedure followed *(Ríos-Insua et al., 1997)* begins by assessing the weights to the attributes and then to the objectives of higher level, because the assessed values in the attributes are later used to obtain the remaining values. In the different levels of the hierarchy the weights are assessed based on trade-offs among the corresponding attributes of the lowest-level objectives which are arising from the same objective. In each intersection of the tree, the decision-maker is asked to determine probabilities for which she or he is indifferent to a lottery and a sure consequence. In essence, the questions are of the type:

- You are playing a game: In one side you can obtain all the best values for all the attributes with a probability $\lambda_i$; in the opposite, you can obtain all the worst values for all the attributes with the complementary probability $(1 - \lambda_i)$. What value should have $\lambda_i$ to make you indifferent to this lottery when it is compared with the certainty of getting all the worst values for all the attributes but for $x_i$, for which you would get the median value?.

As in the case of utility elicitation, we assume imprecision in the sense that it is very demanding for the DM to provide a unique value $\lambda_i$ and the decision-makers will provide instead of such value, an interval. Using the properties of the utility functions, and a procedure for normalising the weights, the values indicated in Table 4 were obtained. They represent the normalised weights for the lowest level objectives (attributes) and their ranges of variability, which can be further used to perform sensitivity analyses.

The same procedure for determining weights, once extended to all the branches of the objectives tree, was the basis for completing the data in Table 5, which includes the normalised weights and ranges of variation for each level of the tree.

*Table 4.- Normalised average values and interval weights for the attributes.*

| Attributes | Normalised weight | Range |
|---|---|---|
| $X_1$: average improvement LEI | .429 | [.214, .644] |
| $X_2$: maximum improvement LEI | .571 | [.380, .761] |
| $X_3$: dose to critical individuals | .067 | [0.0, .134] |
| $X_4$: collective dose | .933 | [.861, 1.00] |
| $X_5$: stress effects | .452 | [.258, .646] |
| $X_6$: reassurance effects | .548 | [.365, .730] |
| $X_7$: amount of fish banned | .379 | [.284, .473] |
| $X_8$: duration of fish ban | .621 | [.518, .725] |
| $X_9$: persons affected by fish recreation ban | .372 | [.278, .464] |
| $X_{10}$: duration of fish recreation ban | .628 | [.524, .733] |
| $X_{11}$: cost of fish consumption ban | .221 | [.147, .294] |
| $X_{12}$: cost of fish recreation ban | .779 | [.719, .839] |
| $X_{13}$: implementation cost | 1 | [1, 1] |
| $X_{14}$: cost of image | 1 | [1, 1] |

*Table 5.- Normalised weights and intervals for the objectives in the hierarchy.*

| Level | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Four | | | Three | | | Two | | |
| Objective | Weight | Range | Objective | Weight | Range | Objective | Weight | Range |
| Environmental Impact | .173 | [.115, .231] | Ecosystem Health | 1 | [1, 1] | | | |
| Social impact | .585 | [.537, .631] | Health impact | .791 | [.718, .862] | Dose to man | .656 | [.524, .787] |
| | | | | | | Health status | .344 | [.229, .458] |
| | | | Living restrictions | .209 | [.139, .279] | Fish consumption | .504 | [.403, .605] |
| | | | | | | Fish recreation | .496 | [.396, .594] |
| Economic impact | .242 | [.217, .266] | Direct effects | .855 | [.683, 1.0] | Fish ban | .766 | [.638, .893] |
| | | | | | | Remedial counter-measures | .234 | [.117, .351] |
| | | | Intangible effects | .145 | [.096, .193] | | | |

***Determining the overall utility and ranking the strategies***

The final step is now obtaining the overall utility of each alternative being evaluated, using the formula given at the beginning of this chapter:

$$u(S_l) = w_1 u_1(x_1) + w_2 u_2(x_2) + \ldots + w_{14} u_{14}(x_{14})$$

Thus, for a given strategy $S_l$ producing a set of consequences $(x_1^l, x_2^l, \ldots, x_{14}^l)$, the individual utilities for each consequence are calculated from the utility functions in Table 3, and they are combined in this additive function, using normalised weights $w_l$ which are obtained multiplying the weights obtained for each attribute by those of their respective objectives of the upper levels in the hierarchy. Hence, the utility function will be

$$\begin{aligned} u(x_1, \ldots, x_{14}) = {} & .074\, u_1(x_1) + .098\, u_2(x_2) + .022\, u_3(x_3) + .283\, u_4(x_4) + .072\, u_5(x_5) + .087 \\ & u_6(x_6) + .023\, u_7(x_7) + .038\, u_8(x_8) + .023\, u_9(x_9) + .038\, u_{10}(x_{10}) + .036\, u_{11}(x_{11}) \\ & + .123\, u_{12}(x_{12}) + .048\, u_{13}(x_{13}) + .035 u_{14}(x_{14}) \end{aligned}$$

In the utility function, it can be observed the importance that the decision-maker in this particular case has assigned to every attribute. The most valued are number 4 (collective dose), 12 (cost of recreation lost by fish ban), 2 (maximum improvement in LEI), followed by 6 (sanitary impact of reassurance), 1 (average improvement in LEI) and 5 (stress effects).

It is also quite surprising that, against what is common in classical cost-benefit analysis, the high importance given to the direct cost of implementation of remedial countermeasures (attribute 13) results in low significance here, when a more wide perspective of the problem is taken. This is one of the main assets of MDA.

Finally, if we represent the overall utility for the average weights obtained, ranking presented in Table 6 for the eight studied strategies is obtained.

*Table 6.- Evaluation of the alternative strategies (average values)*

| Strategy | Description | Overall utility $u(S_l)$ | Rank |
|---|---|---|---|
| $S_1$ | No action | .5032 | 4 |
| $S_2$ | Fish banning | .3897 | 8 |
| $S_3$ | Lake liming | .5040 | 3 |
| $S_4$ | Liming + fish ban | .4563 | 5 |
| $S_5$ | Potash treatment | .6066 | 1 |
| $S_6$ | Potash + fish ban | .5561 | 2 |
| $S_7$ | Fertilising | .4482 | 6 |
| $S_8$ | Fertilising + fish ban | .4230 | 7 |

A sensitivity study could show how sensitive is the final ranking to the imprecision on the utility functions and on the weights expressed by the decision-makers. The methodology for performing such sensitivity studies is being completed, and it will be implemented in the MOIRA system in an interactive module, of the type existing in the well known commercial software V·I·S·A *(Belton et al., 1995)*. A V·I·S·A screen of the case analysed in this report is included below in Figure 3.

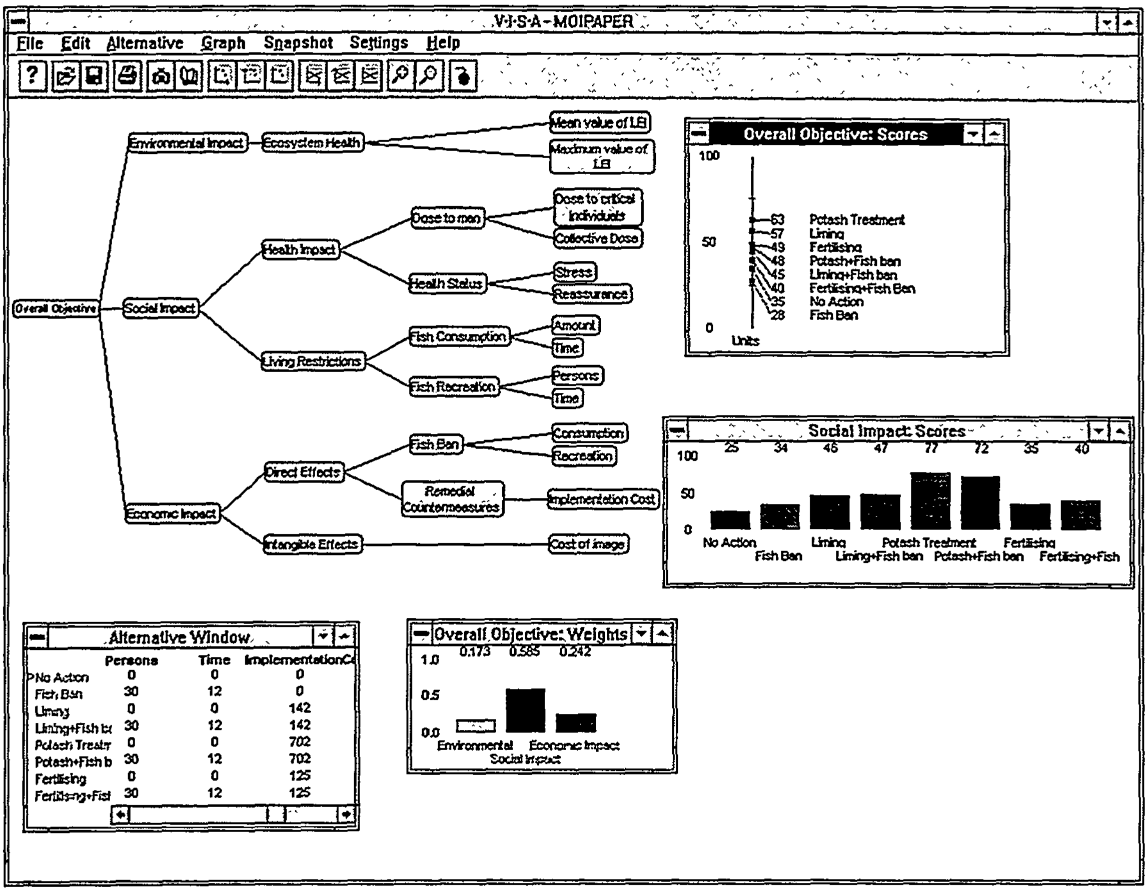

*Figure 3.- V·I·S·A screen for interactive sensitive analysis.*


ACKNOWLEDGEMENTS

The contribution of the following members of the MOIRA team is deeply acknowledged: Dr. John Brittain (Univ. Oslo) and Dr. Per Strand (Norwegian Radiation Protection Institute) provided the definition of the scenario and descriptive data of lake Øvre Heimdalsvatn; and Prof. Lars Håkanson and Mr. Otto Abrahamsson (Univ. Uppsala) ran the lake model.

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*ANNEX: assumptions and data used to run the MOIRA models*

For the Lake Model:

| | |
|---|---|
| Catchment area: | 23.4 $km^2$ |
| Lake area: | 0.78 $km^2$ |
| Mean depth: | 4.7 m |
| Maximum depth: | 13.0 m |
| Fallout: | 130 $kBq/m^2$ of $^{137}Cs$ in the $5^{th}$ month |
| Precipitation: | 800 mm/yr |
| Altitude: | 1090 m a.s.l. |
| Latitude: | 61 degrees North |
| Sedimentation rate: | 70 g/(m x yr) |
| Fish weight: | 50 g (i.e. trout) |
| Primary production: | 27 g C/($m^2$ x yr) |

For the dose model, the following assumptions have been made:

- The annual production of trout from the lake is 445 kg/year *(Vik, 1978)*. This amount is used to calculate the collective dose from ingestion of fish.
- The consumption of freshwater fish by a specially affected individual has been set at 16 kg. The average in Norway is 2.6 kg/year and the local is 8.7 kg/year. All the freshwater fish consumed are assumed to be trout from the lake. This consumption takes place during the period the lake is not covered by ice, i.e. from the beginning of June to mid-October.
- These special individuals consume 2 litres/day of water from the lake during the whole year. The collective dose from ingestion of water is calculated assuming that there are three such individuals.
- The calculation of external doses from the lake are made assuming a 8 hours working day spent boating (summer) or on the ice (winter).

The analysis comprises 72 months, starting in January; the contamination of the water body takes place at month 5.

# DESIGN AND DEVELOPMENT OF THE MOIRA SOFTWARE FRAMEWORK

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## INTRODUCTION

The software system for MOIRA project is designed to cover three main tasks:

1) mathematical simulation (simulation of sequences of radioactive contamination taking into account countermeasures selection and performing of mathematical tasks with respect to optimal strategy selection);

2) input data preparation;

3) making the process of decision making as easy and efficient as possible for the user.

To achieving these goals the MOIRA software system consists of

- software realization of mathematical models;
- geographical information system (GIS) and data bases;
- MOIRA user interface;
- MOIRA operating system making connection between all parts described above.

This report describes the principles of MOIRA software framework design and the way of integration of application tasks, GIS, databases and software framework into the MOIRA software system.

## STRUCTURE OF MOIRA SOFTWARE SYSTEM

The structure of the MOIRA software system is shown in Figure 1.
We will call the software designed to satisfy item 1 — application tasks, items 2 and 3, the MOIRA software framework.

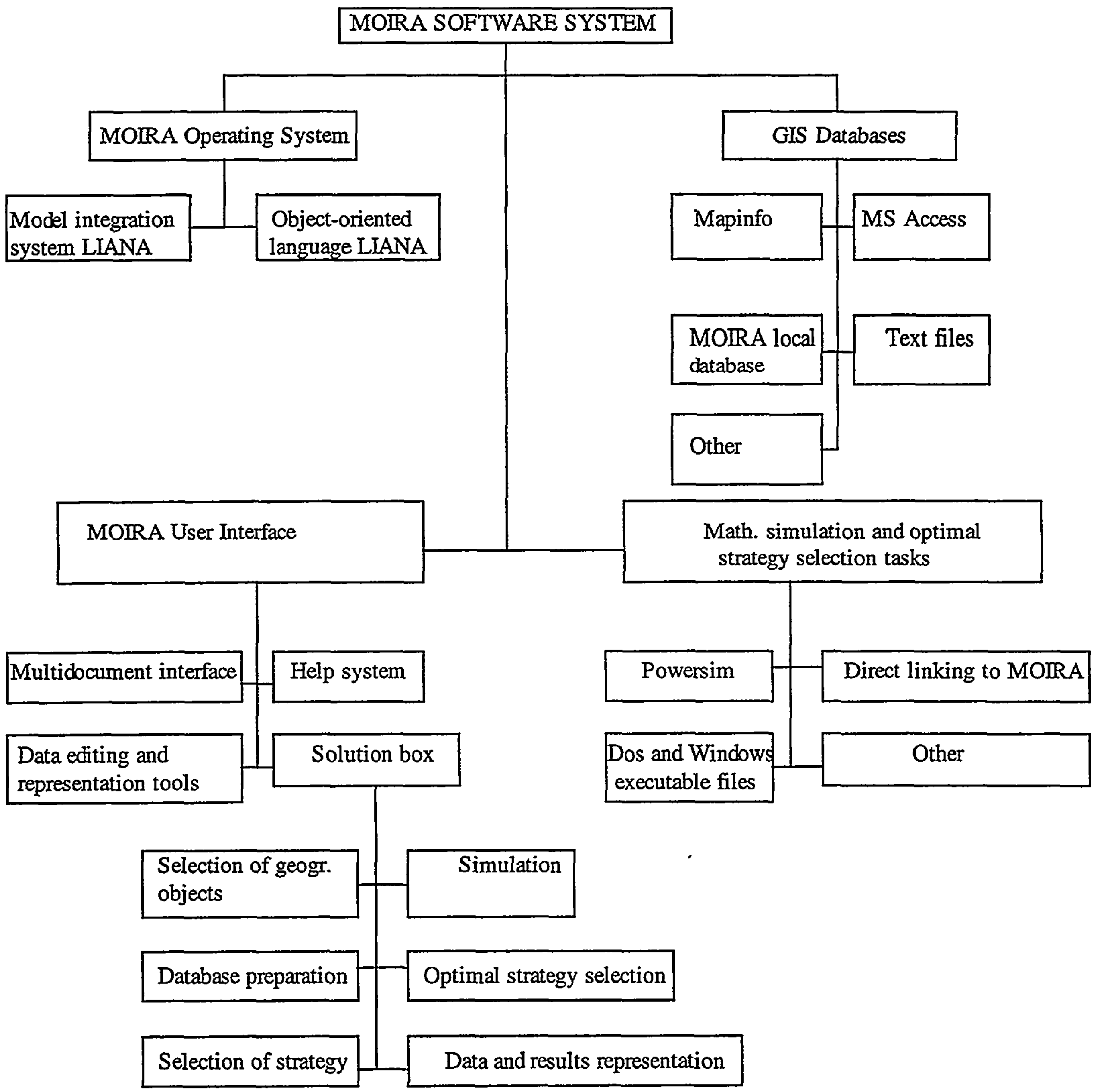


Figure1- Structure of MOIRA software system.

• The software implementations of mathematical simulation and optimal strategy selection tasks are being designed by different participants of MOIRA project [1,7,8,9]. The framework software helps developers to make integration of application tasks into the MOIRA system in a quicker and more efficient way. The framework does not require any special demands for the application task except to produce output in the form of text files. Application can be designed using PowerSim (.sim module ) or any program language like C or FORTRAN (.exe) module. The tasks' demands (input data, interface elements, intertask communication) are described by the task developer using LIANA language (see section "Integration of application tasks, GIS and databases into MOIRA system") and automatically provided by the framework before running of corresponding task.

- The databases and GIS used in MOIRA are intended to store different kinds of permanent and temporary data. The MapInfo system is used as GIS for MOIRA project. MS Access is proposed as the database. The special data storage for the data set used by the framework has been developed (see section "Data sets in MOIRA")

- The user interface for MOIRA is designed to give the user the possibility to make an optimal solution by the easiest and shortest way. For these purposes all layout elements and tools are made user friendly and help the decision maker to do all operations in a more natural and expected way. The user interface works under the Windows 95 operating system and follows all standards used in Windows 95 programs. For detailed description of user interface see section "User interface for MOIRA system."

- The MOIRA operating system will be designed on the basis of the LIANA software system developed by D. Hofman in IPMMS, Kiev, Ukraine 1992-1996 [2-6] and will include:

  - tools for integration of application tasks, interface elements, GIS and databases into the decision support system;

  - use of the object-oriented language LIANA for description of task data and interface demands, format of input and output files, results produced by the task;

  - a visual editor to make the description of the task interface in the LIANA language easy and natural;

  - a core of LIANA software system making the task of execution in the optimal order, data preparation, communication between different parts of software framework;

  - advanced interface elements;

  - facilities for intertask communication in different ways (files, OLE Automation);

  - facilities for management of data storage.

(See section "Integration of application tasks, GIS and databases into MOIRA system" for a detailed description of the MOIRA operating system and the LIANA software system.)

## DATA SETS FOR THE MOIRA SYSTEM

The key notion of MOIRA software framework is the data set. To make simulations the application task needs some input information representing both time-dependent data (for example radioactive contamination) and time-independent data (for example half-time of decay for the simulated nuclide). These data are data sets. The results of simulation are data sets too. Another example of a data set is a description of MapInfo tables used for the current map.

All data sets are kept on the disk as files in the directories corresponding for different solutions (see section "Solution box"). For application tasks data sets will be prepared by the MOIRA operating system in the format described by task developer (using LIANA language ) during the integration of application task into the MOIRA system..

Inside the user interface each data set is represented as an icon (see "Solution box"). Different types of icons are used for different types of data sets. Empty icons are provided for expected but not completed data sets (for example for a new set of countermeasure). Clicking on an empty icon results in calling of the action described during the integration of application tasks or interface tool for creating of corresponding data set (editing data set, calling simulation program, establishing connection with MapInfo system).
If the data set is empty and the action for the creation of data set is the editing of a Table, a tool is called wich loads a sample data set automatically created by MOIRA operating system by analyzing the LIANA language description of this data set. Clicking on an icon which is not empty and corresponds to an existing data set, results in calling of the action described during the integration of application tasks for editing of data sets (it could be Table tool [see section "Table tool"], MapInfo map [see "Integration of application tasks, GIS and databases in MOIRA" ] or other actions).

## THE TABLE INTERFACE TOOL

To edit input data and to represent output from application tasks the Table interface tool was designed. The table interface tool can be used for any type of data both in numerical and string format. Working with tables, the user can edit data already represented in the table, add and delete rows. It is possible to change the width of columns to preview table in a more convenient way.

*Including and deleting rows into the table.* Some table allows row adding and deleting. The example of such a table is a table representing the data set for radioactive contamination in the water (month/concentration) [8].To include a row into the table the user needs to click "+" button in the table toolbar. A row will be added after the last row of table. By default the first column of new row will contain the number of the row. A zero value will be placed into the other numerical cells of the new row. In the next versions it will be possible to describe other default values or to fix the maximum number of rows during the integration task description. The "+" toolbar button is inactive and represented by gray color if the type of table does not allow rows to be included or if the table already has maximum number of rows. To delete rows the user needs to click "-" button in the table toolbar. The last row will be deleted. The contents of row will be lost. The "-" button is inactive if type of table does not allow deleting og rows or number of rows is equal to 0.

*Editing data into tables.* To edit data into the table the user needs to click on the small icon "boxes" placed on the left side of each row. The corresponding dialog window containing

data from row will appear. Editing data in this window followed clicking "OK" changes data in the corresponding row of the table.
The example of the table interface tool is shown in Figure 2.

*Graphical tools for data editing and representation*

Graphical representation of data in the table columns and rows as 2D graphs, bar- and pie-charts will be implemented in the future.

It is planned to design a graphical source editor and attach it to the table used for editing time-dependent data.

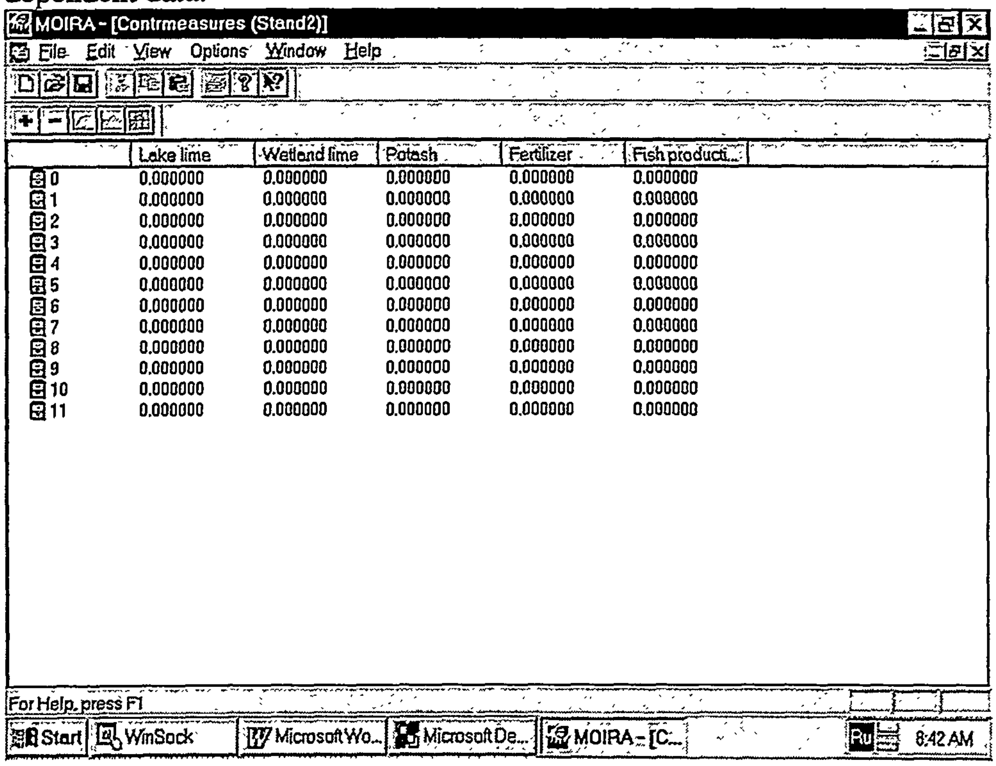


Figure 2 - Table interface tool

## THE MOIRA USER INTERFACE

The key idea for development of the MOIRA user interface is to make the process of decision making for MOIRA users as easy and efficient as possible. We can split future users of MOIRA system into 3 groups:

- Experts. Experts in the field of radioactive contamination, economy, etc., who use MOIRA system permanently as a working instrument. Need detailed information about results produced by every model included in MOIRA as tables and graphs.

- Decision makers. Work with MOIRA temporarily and often use data prepared by experts. Need information from the MOIRA System in the form of advanced reports.

- Software experts. Help experts and decision makers to work with MOIRA system. Prepare all input data working with GIS and data bases directly and via MOIRA system. Advanced users of MOIRA system.

Very often one person can combine some of these roles.

To fulfil the requirements of all groups of users MOIRA interface should:

- provide different levels of detailisation for different user levels - “expert”, “decision maker”,”software expert”, but at the same time allowing the extract to take all the detailed information he or she wants;

- give the user the possibility of making decisions in the “automatic” mode, step-by-step or in a non predefined order;

- be easy to learn and comfortable for users;

- make advanced reports;

- help users at each step in working with the interface.

To achieve these goals the MOIRA user interface has the structure shown in Figure 3.

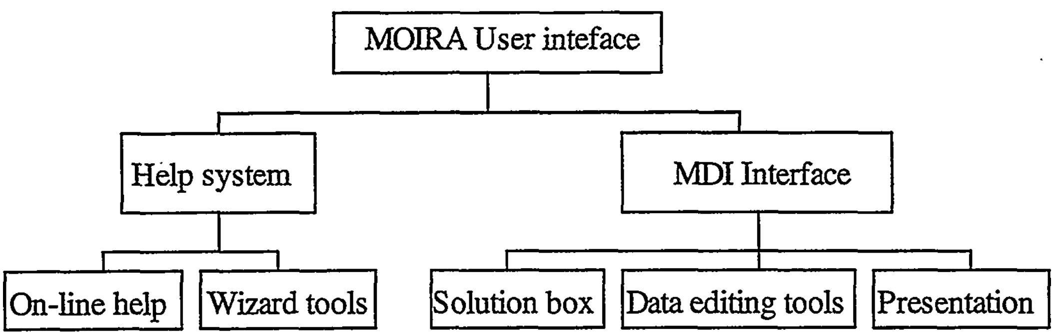


Figure 3 - Structure of MOIRA user interface.

*Solution box*

The solution box realizes the metaphor of the working table of the expert or the decision maker. The user can have so many solution boxes as he or she needs or one solution box can

be used every time. All data sets (maps, input data, countermeasures set, reports) are shown in the different pages of the solution box as different kinds of icons.
Examples of solution boxes are presented in Figures 4 and 5.

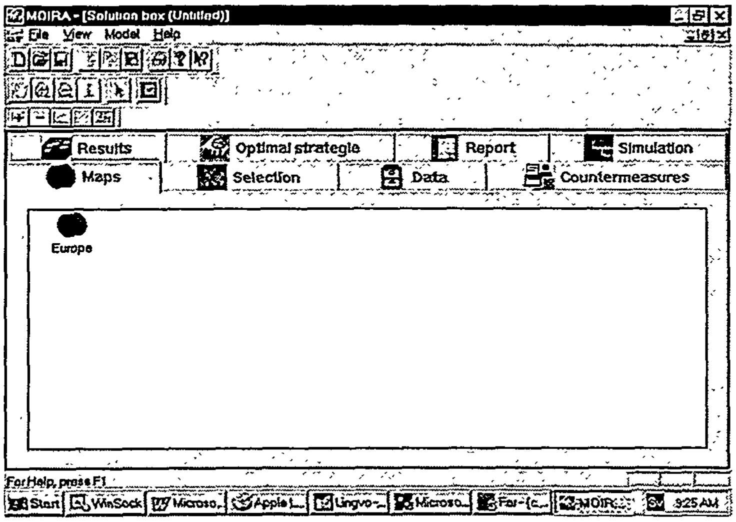


Figure 4 - Solution box. Maps window.

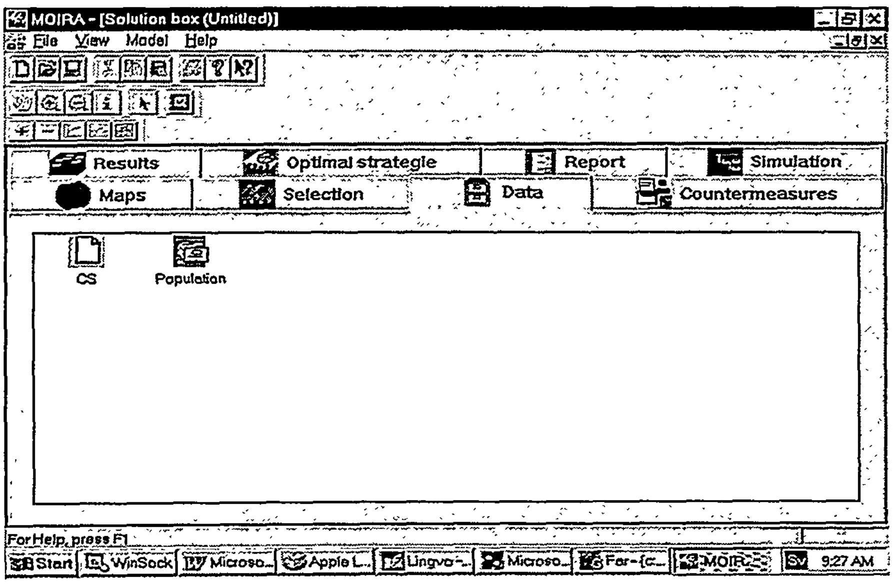


Figure 5 - Solution box. Prepared (Population) and not prepared (Cs) data sets in the data window.

The main structure of the solution box is shown in Figure 6 and corresponds to the decision making process. The gray squares represent the steps of decision making. The double-framed white ones represent MOIRA user interface elements (solution box windows, data editing and representation tools). Shadowed white boxes represent the data that are proposed to distribute together with MOIRA system. The single-framed boxes represent connection with other software packages.

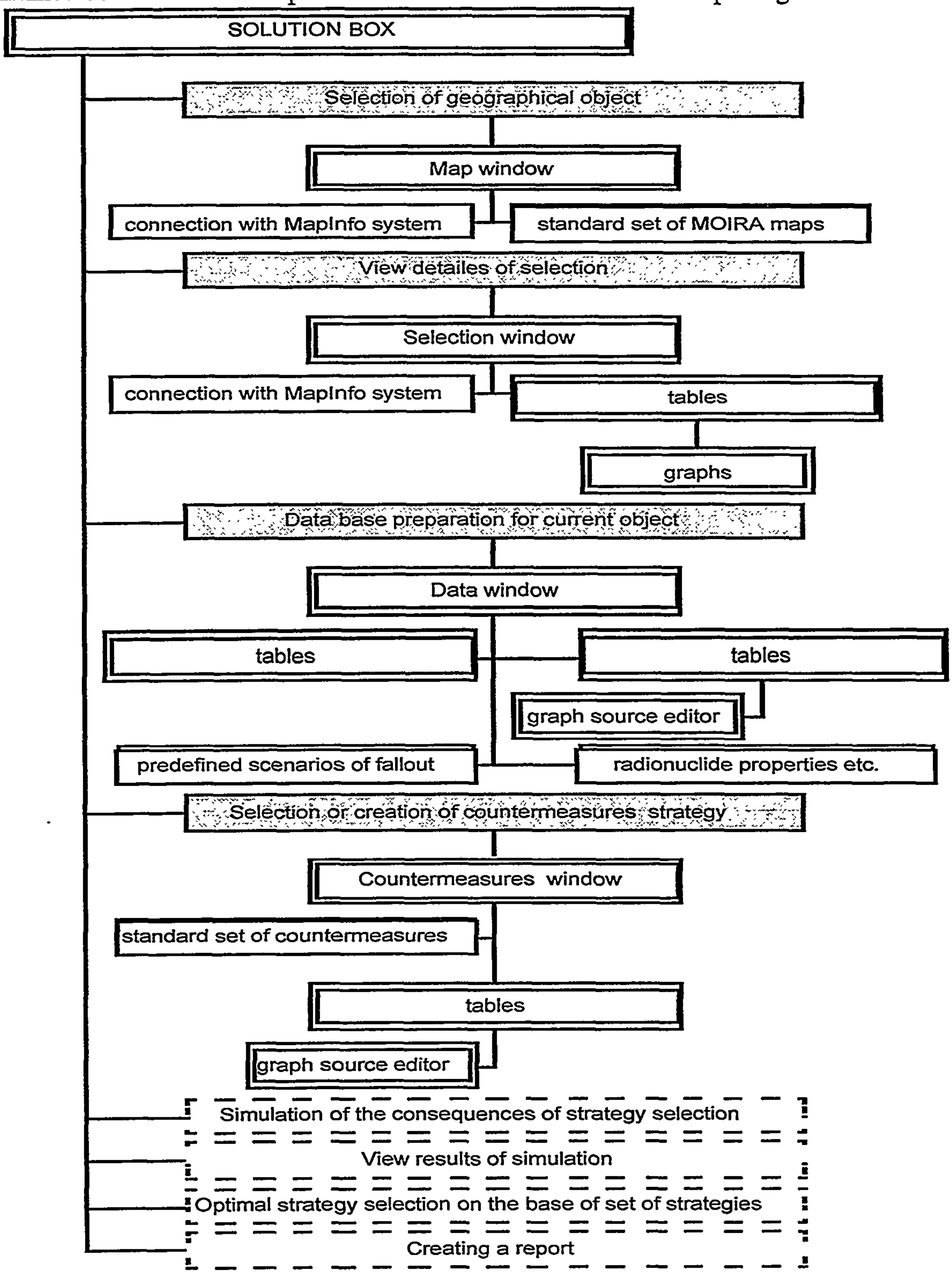


Figure 6 - Decision making steps and structure of solution box.

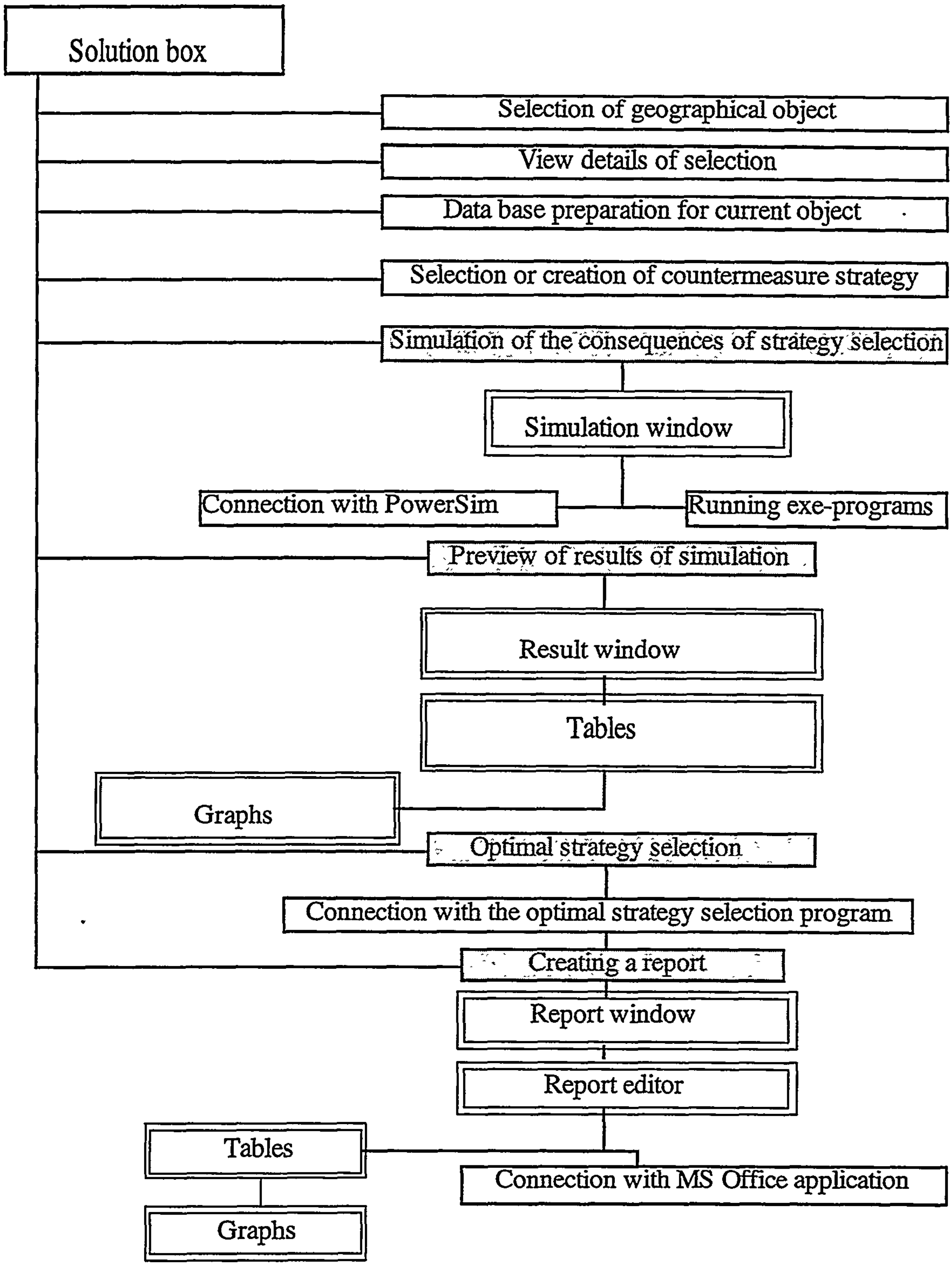


Figure 6 (continued)

*Solution box windows*

♦ Map window. The Map window contains the standard and local set of maps. The user can work with a map clicking on the corresponding icon. The standard set of MapInfo operations (*select, move, zoom in, zoom out, get information*) are accessible with corresponding buttons of the map window toolbar. For a detailed description of MapInfo operations, see [10]. The user selects geographical objects through the Map window. Selection is made by selecting an of object on the map and clicking *Apply selection* button in the Map window toolbar. To provide Map window connection functionality with MapInfo, GIS based on OLE automation is used.

♦ Selection window. The Selection window contains the map selected by the user via the Map window. All operations provided for Map window except Apply selection are accessible in the Selection window. An empty selection window means that no selection was made via the Map window. To provide Selection window connection functionality with MapInfo, GIS based on OLE automation is used. It is planned to give the user possibility to use all information about selected object through the Selection window. For these purposes Table tool (connected with MOIRA data storage) and connection with MS Access could be used.

♦ Data window. This contains all input data sets, which should be prepared by the user. To provide Data window connection functionality with the MOIRA data storage and Table interface tool are made as described in section "Data sets for the MOIRA system" and "The Table interface tool".

♦ Countermeasures window. This gives the user the possibility of creating and selecting a set of countermeasures. To create a new countermeasure set, the user needs to use *Create new* item from pop-up menu accessible by right mouse button clicking. It is planned to use both some predefined set of countermeasures and contermeasures created by the user. Countermeasures sets can be edited by the Table interface tool and saved in the MOIRA data storage. For Countermeasures window functionality, the MOIRA operating system will provide connection with the MOIRA data storage and Table interface tool. For making optimal solutions, the user will have possibility to select some countermeasures.

♦ Simulation window. This allows the user to run simulation programs. The simulation programs makes simulation for objects selected via the Map window, using input data prepared via the Data window and data for countermeasures prepared and selected via the Countermeasures window. The progress dialog to show the simulation process progress is provided. The connection with PowerSim based on OLE Automation is used. It is planed to use other ways of model running (.exe files running ) too.

♦ Results window. This shows the results of simulations made via the Simulation window. It uses Table interface tools and will use Graph interface tool (see "Table interface tool"). it provides a separate icon for each simulation with a different countermeasure set. To provide Result window connection functionality with MOIRA, data storage is used. It is planned to use an empty icon in the Result window the same way as in Data window to call simulation process automatically based on models description during integration (see sections "Data sets in MOIRA" and "Integration of application tasks ..."). In the future, Result and Simulation windows could be joined for complete representation of the process of simulation.

♦ Optimal strategy window. It is planned that user will open the optimal strategy selection program via this window. The optimal strategy selection will be based on the group of results of simulation selected in the Countermeasures window.

♦ Report window. It is planned that advanced reports reflecting all steps in the decision making and the final decision will be created via this window. To give user possibility to export this report into the MS Excel and MS Word, OLE and DDE technology could be used.

The MOIRA user interface will provide possibilities to save solution boxes (with all information this solution box contain) on the hard disk and load it again. Solution boxes will be placed in the separate directories on the hard disk. These directories will contain all files - data sets for current solution (see chapter "Data sets for the MOIRA system"). It is planning to make solution box in the Wizard mode (with "Back", "Next" and "Auto" buttons) to give user possibility make decision in an automatic mode or step-by-step.

*Interface tools*

For data preparation and presentation the Table interface tool is used in the MOIRA user interface. It is planned to attach different kinds of graphical tools to the Table interface tool. For a detailed description of Table tool see section "Table interface tool"

*Help system in the MOIRA user interface*

The MOIRA user interface will provides the advanced help system. Use of short tool tips for each button and status line information is available in the present version of the interface

In the next versions it will be possible for the user :

- to obtain help information for all interface elements - buttons, toolbars and windows, clicking the help button on the main toolbar followed by clicking on the interface element;

- to obtain detailed on-line help for the MOIRA software system elements - solution box windows, data sets icons, MapInfo maps and so, by pointing the cursor to a specified element and clicking the F1 button;

- to find help topic by Help-Search tools;

- to use wizard tools to follow the user step-by-step from the beginning of working to making the final optimal solution.

## INTEGRATION OF APPLICATION TASKS, GIS AND DATABASES IN MOIRA

### *Connection with the MapInfo system*

GIS is an important part of the MOIRA system, because the selection of geographical object and obtaining its properties is the starting point that join the application models' level of the MOIRA system with the user levels. MapInfo GIS was selected as the geographical information system used in MOIRA. To make the connection between MapInfo and MOIRA, the OLE Automation method is used. The usage of class DMapinfo provided by MapInfo API [11] makes it possible to run any MapBasic command [10,11] from the MOIRA framework and reflect the results of command execution in any window. The MapInfo functions are:

- Showing the maps contains certain levels;
- Selection of a geographical object;
- Obtaining information for a geographical object in the standard MapInfo "Information" window;
- Obtaining information by fetching tables;
- Move tool;
- Zoom-in tool;
- Zoom-out tool.

The user obtains access to standard set of maps via the Map window of the Solution Box (see "Solution box" subsection in " The MOIRA user interface"). The selection made by the user is accessible via the Selection window of the Solution Box.

The tables included in the each map from standard set of maps are described in the configuration files.

### *Connection with Powersim system*

The Powersim system [12] designed by the Powersim Corporation is very useful for design and development of software realizations for application tasks performing time dependent mathematical simulation. It gives the user the possibility to construct and validate software realization of mathematical models using perfect visual editing tools. Powersim was selected as one of the recommended tools to produce software realization of mathematical models in the MOIRA project. Some application tasks included in MOIRA are already designed in Powersim [8].

Powersim API [13] includes class CPSConnections giving applications the possibility to transfer data to and from Powersim and to manage process of simulation. The CPSConnection provides functions which give possibilities:

- to set start and finish times of the simulation;
- to set the simulation process;
- to start and stop the simulation process;
- to make a pause in the simulation process;
- to transfer data to the application task;
- to get data fromt he application task.

It is also possible to transfer data to and from Powersim with text files. These can be "time series" of data or sets of a single parameter. If some input text files are included in the "time series" and times are differ in different "series", Powersim makes automatic interpolation of data. The transformation via text files is selected to transfer data from the appliction task to the MOIRA operating system and from the MOIRA operating system to the application. The advantage of this method is the possibility to prepare and receive all data automatically based on the LIANA language description of the application task. The transformation via text files gives the possibility to run Powersim-based modules,. exe modules or modules based on other software packages with the same method of data exchange. The OLE Automation connection will be used in the next versions of MOIRA software framework to transfer start and stop time values, transfer timestep value and to manage the process of simulation.

*The key ideas of integration of the application task, GIS and databases into the model-based decision support systems used the LIANA software system*

Each application task:

- *Prepares* some data for another tasks running;
- *Needs* input data made by another tasks or prepared by the user;
- Can only run if all input data are ready.

The decision making process can be represented as a tree in the terms of data preparation corresponding to goals. An example of such a representatation is shown in Figure 7.

Each data set can be represented by a description of its contents. The format of file containing this data, the application task that made this data set and the input data must be prepared before this task can run. The example of such a representation for the LAKECO model is shown in Figure 8

Each data set can be represented in the LIANA language as a class including:

- *Produces* section - describes the output of class accessible from all other application tasks;
- *Needs* section - describes data that must be prepared before task can be run;
- *Realization* section - describes the type of task producing output data (.exe module, Powersim-based and s.o.). Could include another statement describing action before and after task running.

- *Represented* section - describes the text file(s) format contain output data
- *Private* section - describe variables which are used internally in the class

An example is shown in Figure 9. A visual editor will be designed to support this description making.

The decision making process shown in Figure 7 can be described and represented as a hierarchy of such classes .

This decision making process can be made automatic by parting this tree from the top "Solution" by the following procedure:

- Check class corresponding to output data set (section Prepares);
- Check if all input data described in the Needs section are prepared;
  - If *yes*, call Realization to make output data set;
  - If *no*, repeat procedure for class corresponding to not prepared input data set;

Class could include interface elements (like Table tools), connection with data bases, GISes, Powersim, other software packages or running of .exe modules in the Realization section.

The application must produces result in the form of text files. The format of files should be described in Represented section. Framework accept these data from files and store them in the corresponding class variables.

Application obtains input data from the framework in the form of text files. If a variable of a certain class is placed in the Needs section the framework makes the text file in correspondance with the Represented section of this class.

Such a method of model integration makes a decision support system very flexible. It gives the possibility to change models without influencing other models, and to exclude and include different kinds of models without changing the software framework.

*Example: Integrating DOSE and LAKE models into MOIRA*

To make an optimal solution in MOIRA we need to evalute doses received by population of a certain region. This calculation is made by the DOSE model [9].

The DOSE model needs data about time (in the units of days), concentration in the water, in sediments and in the fish. It accepts these data from file "INPUT.DAT" and this file has the following format:

| | | | |
|---|---|---|---|
| 10. | 1.2e+8 | 1.2e+6 | 1.2e+7 |
| 50. | 1.0e+8 | 1.0e+6 | 1.0e+7 |
| 100. | 1.0e+8 | 1.2e+6 | 1.5e+7 |
| 150. | 6.0e+7 | 8.0e+5 | 3.9e+6 |
| 200. | 3.0e+7 | 9.0e+5 | 1.5e+6 |
| 300. | 1.4e+7 | 6.4e+5 | 1.0e+6 |
| 400. | 5.4e+6 | 5.4e+5 | 1.3e+6 |
| 500. | 1.0e+7 | 1.0e+5 | 1.0e+6 |
| 600. | 1.0e+7 | 1.0e+5 | 7.0e+5 |
| 800. | 6.0e+6 | 1.1e+5 | 4.0e+5 |

The LAKE model [8] produces data for concentration in water, in phytoplankton, in prey fish and in predatory fish. The time unit of the LAKE model output is one month. After some transformation this data could be used to make the DOSE model simulation.

We can formally describe the sentences written above using LIANA language. After this description is accepted by the LIANA interpreter, the MOIRA software framework will be able the to run DOSE and LAKE model in correct order (first LAKE than DOSE), automatically preparing all files for their run including data transformation between LAKE model output and DOSE input.

The LAKE model produces data in the form of a text file made by PowerSim. For this example we can assume that name of file is "CONC.OUT". This file contains a "head string" - "Time Conc_in_water Conc_in_phytoplankton Conc_in_prey Conc_in_predator" and 5 columns represents these values. The example of these file is:

| Time | Conc. in water | Conc. in phytoplankton | Conc. in prey | Conc. in predator | |
|---|---|---|---|---|---|
| 1 | | 0 | 0 | 0 | 0 |
| ................................................ | | | | | |
| 4.75 | 0.181686 | 0 | 0 | 0 | |
| 5 | | 0.7065049 | 0 | 0 | 0 |
| 5.25 | 1.707413 | 1.57628 | 1.086569 | 2.470301 | |
| 5.5 | 3.289105 | 8.452218 | 5.980514 | 13.30849 | |
| ............. | | | | | |

To integrate the LAKE model into MOIRA we need to create the class which describes this model. Assuming the name of class is LAKE.

The Produces section of this class should be described as:

*REAL* T( );
*REAL* WaterConc( );
*REAL* PhytoConc( );
*REAL* PreyConc( );
*REAL* PredConc( );

The Represented section should be:

```
AS "CS.TXT"
INTEGER I;
<<"Time Conc_in_water Conc_in_phytoplankton Conc_in_prey Conc_in_predator">>
 REPEAT
            I=1
            <<T(I)," ",Water_Conc(I) >>
UNTIL(NOT(EOF))
<<EOF>>
```

The Realization section should look like:

```
RUN "LAKEMODL.SIM";
```

To make the transformation between LAKE model output and DOSE model input files we need to describe some new class. Let the name of this class be PrepDOSE.

In this case the Needs section of PrepDOSE class should contain a string like:

```
LAKE lake;
```

The Represented section of class PrepDOSE should look like:

```
AS "SOMENAME.DAT"
REAL T1( );
REAL ConcBot( );
REAL ConcFish( );

INTEGER NumTimes,I;

NumTimes=DIM(lake.T,1);

FOR I=1,NumTimes
{
ConcBot(I)=0;
ConcFish(I)=(lake.PreyConc(I)+lake.PredConc(I))/2.;
#These formulas are only examples
#should be of course another correct formulas
#to calculate concentration in the bottom and in fish

<< lake.T*30," ", ConcBot(I)," ",ConcFish(I)>>
}
<<EOF>>
```

The Realization section should be:

```
RUN;
```

The Produces section could be empty;

Lets class DOSE represents DOSE model.

In this case to make INPUT.DAT automatically before the dose model is run we only need to include in the Needs section of class DOSE one string like:

PrepDOSE prDose("input.dat");

The Realization and Produces sections should contain corresponding output variables and file format for the output information of the dose model.

The Realization section should contain :
RUN "LAGOS.EXE";

During the simulation the framework first checks files described in the Needs section of class Dose. In our case it checks if file INPUT.DAT (for current strategy) exists and has correct time (not late then SOMENAME.DAT). If the file exists the simulation for the dose model will be immediately performed. If the file doesn't exist or time of file creating is not correct the procedure will be repeated for Needs section of PrepDOSE class - for file CONC.OUT , then for Needs section of LAKE class and so on. When all information described in the Needs section of class LAKE is ready the system will run LAKEMODL.SIM via the connection with POWERSIM. After finishing the run the data from file CONC.OUT will be stored in corresponding arrays described in Produces section of the class LAKE. Then a transformation file SOMENAME.DAT will be created which uses these values in the format described in the Represented section of PrepDOSE class. This file will be renamed INPUT.DAT . After the preparation of all other data described in Needs section of DOSE class LAGOS.EXE will run. The data described in the Produces section of class DOSE will be stored and could be used in the other classes (for example to describe economic models, optimal solution selection model, etc. )

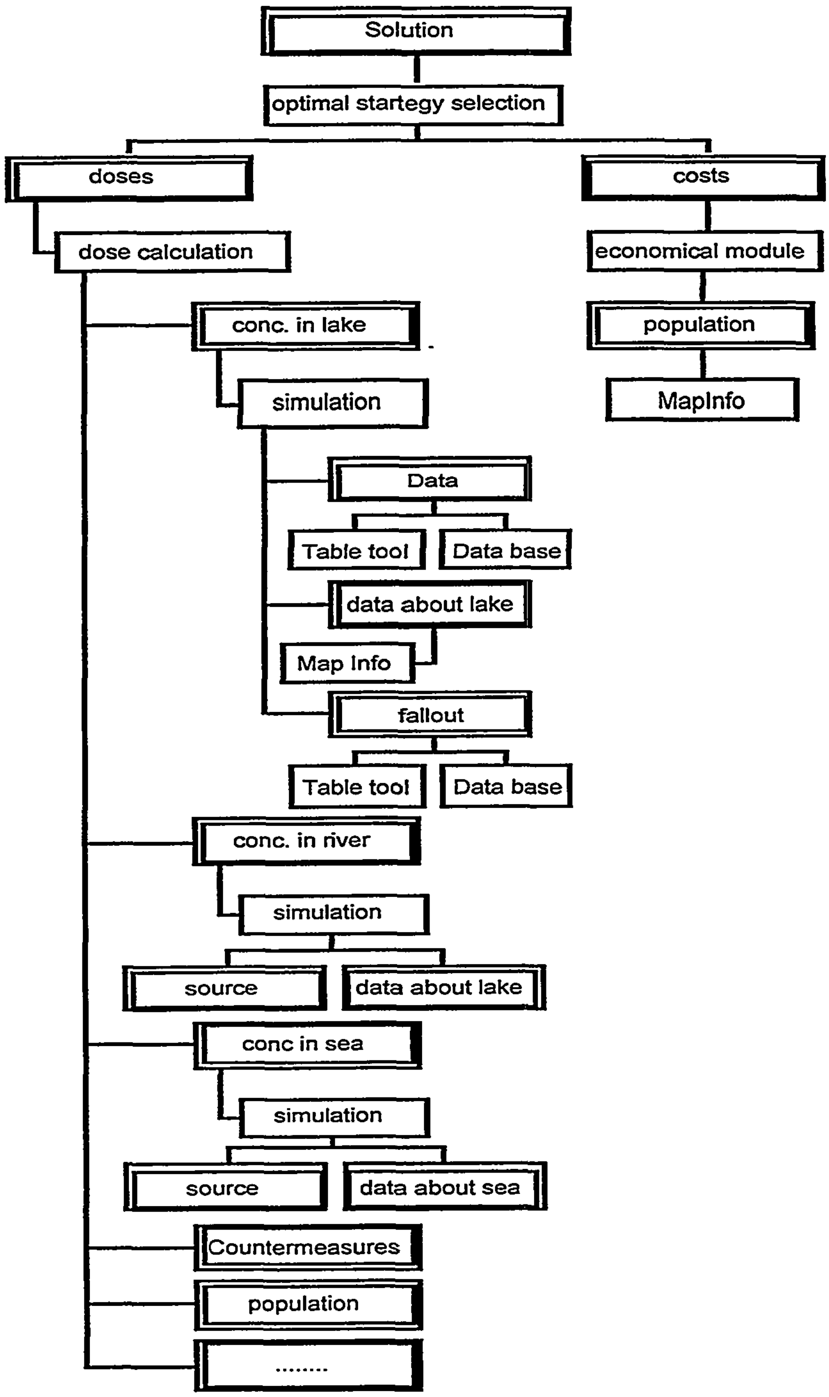


Figure 7 - Representation of the decision making process.

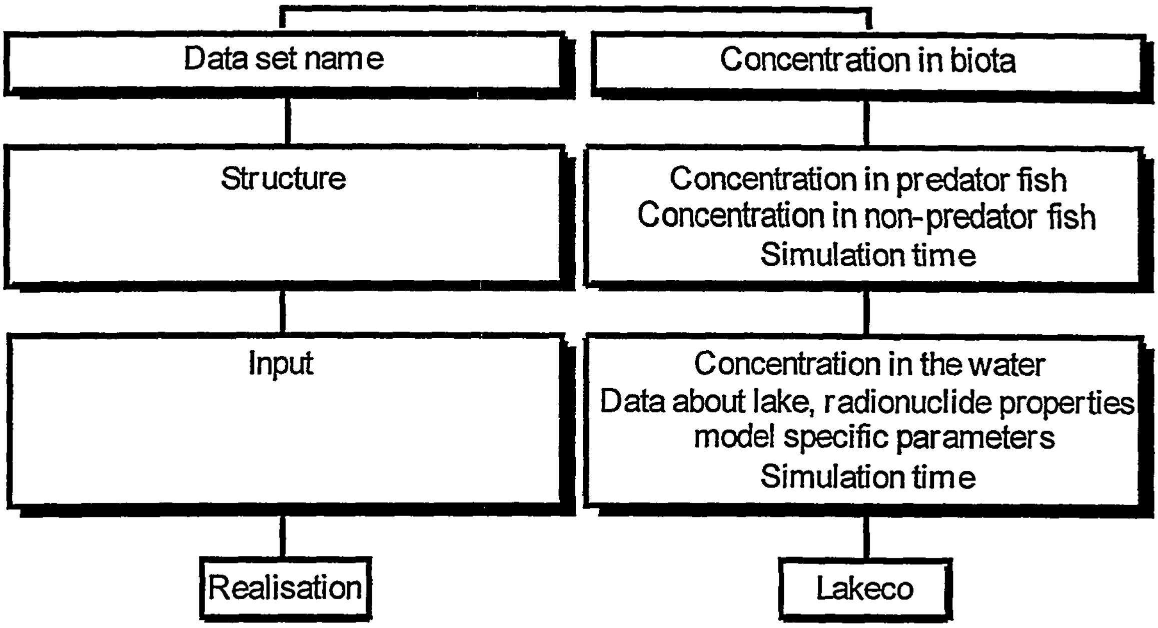


Figure 8 - Example of the conceptual representation of the "Concentration in biota" data set produced by LAKECO model [7].

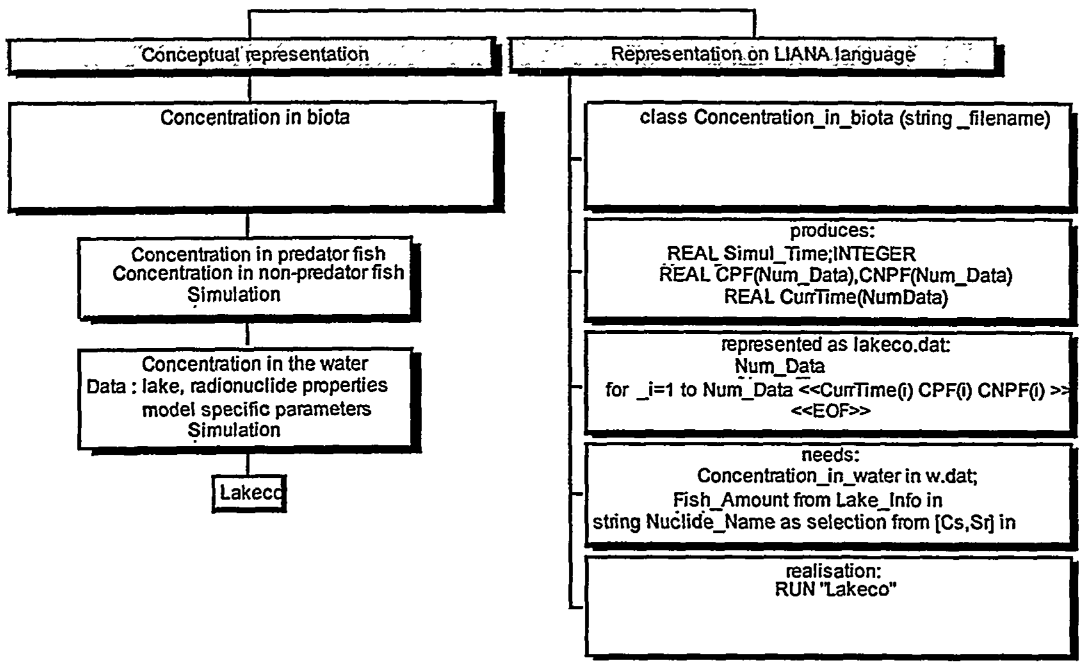


Figure 9 - Representation of the "Concentration in biota" data set in LIANA language.

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# THE APPLICATION OF THE LAKE ECOSYSTEM INDEX IN MULTI-ATTRIBUTE DECISION ANALYSIS IN RADIOECOLOGY

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## INTRODUCTION

The aim of this work is to present a new version of the lake ecosystem index (LEI; see Håkanson and Peters, 1995; Håkanson, 1996 for further information about LEI) to be used within the framework of a multi-attribute decision support system in aquatic radioecology (see Rios-Insua et al., 1996 and Appelgren et al., 1996; Monte etal., 1997 for further information about this multi-attribute analysis system).

Within this overall objective, there are three specific goals of this paper:

Firstly to give a rationale for this work, secondly to motivate the modifications and simplifications of the existing LEI-model and to illustrate the practical use of lake ecosystem index, and finally to define a utility function, u(x), which is needed in the multi-attribute analysis to compare different objectives.

Several chemical remedial measures have been tested to try to decrease the concentrations of radiocesium in lake fish (see Håkanson and Andersson 1992 for further details):

- Wetland liming and full-scale catchment area liming have been tested to reduce the transport of cesium from land to water through the modification of soil chemical processes to reduce cesium mobility.
- Lake liming was tested to increase the proportion of cesium deposited on the lake bottom, thereby preventing or delaying its biouptake. The hypothesis is that the binding to particulate matter and the flocculation tendency and sedimentation of the cesium-carrying particles is increased by increasing the pH, alkalinity and hardness of the lake water.
- Potash treatment has been carried out to increase the potassium concentration of the lake waters thereby blocking and reducing the proportion of cesium taken up by fish. Potassium and cesium are taken up in fish in a similar manner (Black 1957; Fleishman 1963; Carlsson 1978). Other ions, such as Ca, Mg and Na, may also participate in different blocking processes. This implies that the different liming measures, which give a general increase in the ionic strength of the water may also have a positive effect in this manner.
- Reducing the fish stock (intensive fishing) or changing the predation pressure so that the nutrient web is changed in such a way that the phytoplankton biomass increases, which could result in lower concentrations of radiocesium in biota at all trophic levels.
- Lakes 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. By means of "biological dilution" this will decrease the concentration of cesium in each individual fish. The method is based on theories involving biological buffering (Jansson et al. 1981).

Within the MOIRA project, new models have been developed to predict realistic water chemical changes (e.g., in lake pH) from lake liming (see Ottosson and Håkanson 1997), changes in K-concentration from potash treatment (Abrahamsson and Håkanson, 1997) and changes in total-P concentrations from lake fertilization (Håkanson et al., 1997). By means of the overall lake model for radiocesium (the modified VAMP-model; see Håkanson et al., 1996) one can then also predict how these water chemical changes influence the concentrations of radiocesium in water and biota (fish).

However, there may also be negative effects on a lake ecosystem from these remedial measures. The lake ecosystem index (LEI) is utilized in this context to account for such potential problems in a simple, rational manner from a radioecological perspective. It should be stressed that the approach to determine the lake ecosystem index is intentionally very simple. Although, ecosystems are very complex (see Håkanson and Peters, 1995) since many chemical, physical and biological factors and processes interact in a such a way that even very extensive models are simplistic. Large models are comparatively easy to built, but are very difficult to validate, especially for many individual ecosystems. The greatest challenge is to built models which are based on the smallest possible number of driving variables (see Håkanson, 1995 for a discussion about the optimal size of predictive models).

This new and simplified version of the original LEI-model is meant to relate directly to the mentioned chemical remedial measures used in the MOIRA project, lake and wet land liming, potash treatment and fertilization. This means that the original sub-models in LEI for lake colour and mercury in fish (see Håkanson, 1996) are omitted in this approach.

## THE STATE VARIABLES - TOTAL-P AND PH

The most important nutrients in aquatic ecosystems are phosphorus and nitrogen. Total phosphorus (TP) has long been recognised as the nutrient most likely to limit lake primary productivity (Schindler, 1977, 1978; Chapra, 1980; Wetzel, 1983). Several compilations of models, theories and approaches to the role of phosphorus in lake eutrophication already exist (Chapra and Reckhow, 1979, 1983; Vollenweider, 1968, 1976). Both experimental and comparative studies of whole lake ecosystems have been used to derive loading models for lake management (Dillon and Rigler, 1974; Schindler et al., 1978; OECD, 1982). A key element in this development was the identification of the simple relationship between sedimentation of phosphorus and water turnover in lakes. The concentration of total phosphorus can be linked to most other biological lake characteristics (table 1). Thus, the concentration of total-P in the lake water is a powerful predictor in limnology and one of the traditional key variables used to classify lakes (see table 2).

There is a significant overlap between the classes ultra-oligotrophic, oligotrophic, mesotrophic, eutrophic and hypertrophic in table 2. This is further shown in fig. 1 for total-P and a key operational variable for eutrophication effects, lake chlorophyll-a concentration. The information about the variability within the different classes in fig. 1 for total-P will be used in this study as an important input for the derivation of the utility function of the LEI to handle changes caused by chemical remedial measures in individual lakes. The information given in OECD (1982) is also compiled in table 3, which gives the actual data on the spread (the standard deviation, SD) for each trophic class. One can note from table 3 that for oligotrophic lakes (there are data from 21 lakes with a range in total-P from 3 to 17.7 µg/l), 1 SD corresponds to ± 0.22 for the logarithmic values, or, for a mean value (MV) of 8 µg/l, one has $10^{0.9-0.22}$= 4.8 to $10^{0.9+0.22}$=13.2 µg/l. From table 3, it can be seen that the mean absolute difference for the three classes is about 0.25. This means that 95% of all total-P values for all the given classes fall approximately within an order of magnitude, since the lower 95% confidence limit is given by $10^{x-2*0.25}$ and the upper 95% conf. limit by $10^{x+2\wedge0.25}$ and the range by $10^{x+2*0.25}/10^{x-2*0.25}$ or $10^{(x+2*0.25-x-(-2*0.25))} = 10^1 =$ 10. This mean that one can be quite certain (a 95% certainty) that if a action causes a change in mean monthly TP-concentration from x to 10*x for a given lake, then it is very likely that there will also be a change in lake trophic level, which entails a change in the abundance and/or biomass of several key functional organisms, like the key fish species illustrated in table 2.

This is crucial information in the definition of the requested utility function, which should, ideally, be set to 1 (or 100%) if there is a total change in the ecosystem structure in such a way that the normal (or reference/initial) key functional groups are replaced by other groups which would be typical for another trophic levels.

Lake pH is also a limnological state variable which influences the entire ecosystem. Fig. 2 shows that many animals accustomed to a circum-neutral pH (pH ≈ 7) cannot reproduce and survive in acidified lakes. Some, like crustaceans and snails, are very sensitive to changes in pH, whereas other animals, like perch (*Perca fluviatilis*) and pike (*Esox lucius*), are less sensitive. The literature on anthropogenic acidification of land and water, its ecological damage and its economical consequences is extesive (e.g., Likens et al., 1979; Ambio, 1976; Overrein et al. , 1980; Monitor, 1981, 1991; and Merilehto et al. , 1988). There are many models and modelling approaches to address the acidification of aquatic and terrestrial environments and to propose remedial measures for acidification (Eliassen and Saltbones, 1983; Sverdrup, 1985; Warfvinge, 1988). There are even entire books of literature references on acid precipitation (e.g., Seip et al., 1989).

In Sweden lake acidification has caused more harm for fish biology, in terms of number of lakes with reduced or extinct number fish species, than all of the other listed threats (fig. 3). Liming as a countermeasure will not solve the acidification problem, but it can prevent severe damage to ecosystems until the sulphur emissions are reduced to levels that natural ecosystems can withstand in the long term. About 8000 lakes in Sweden have been limed and the total annual cost for liming in Sweden was about SEK 200 million (≈20 million ECU) in 1994 (Henrikson and Brodin, 1995). Since there are great costs connected to liming, it is very important that the liming is performed effectively and that the amount of lime in each lake is optimised so that it will give the most suitable change in water chemistry (pH, alkalinity, etc.) at a minimum cost. Different lakes will not respond in the same manner when limed. The response depends very much on the environmental properties of the drainage area, the lake water retention time, etc. This means that it is very important to have a method to calculate the amount of lime needed to achieve a certain pH in a certain lake. The aim of the sub-model for lake liming (Ottosson et al., 1997) is to present realistic pH-responses to lake liming.

## THE LAKE ECOSYSTEM INDEX (LEI) - DEFINITIONS AND SET-UP

The complexities involved in establishing simple, practical and meaningful ecological indices sometimes seem insurmountable. Still, the benefits of even crude environmental indices are so great, that they are worth pursuing. So long as one clearly states the criteria, theories and evidence, then these components can be discussed, tested and improved.

The first step in LEI is to establish normal values (corresponding to natural, initial, pre-industrial or reference conditions) for the state variables. These values may be predicted from empirical models (see Håkanson and Peters, 1995). In the MOIRA expert system they may be derived from algorithms based in GIS-information (e.g., land use, bedrocks, soils, vegetation, precipitation and topography), or obtained from data bases. It is assumed that normal values for lake total-P concentration and lake pH are available (they will be used as reference constants), the actual values of lake total-P concentration and lake pH are calculated from sub-models for liming and fertilization (see fig. 4, which gives an outline of the modified LEI model).

The index reflecting the status of fish, the fish yield ratio, FYR, is defined from the following arguments:

1. Actual and normal fish yield (in mg ww/$m^2$*yr) is first calculated from the equation given in table 1. FYR' = 7.1·TP, where TP is the mean monthly concentration of total-P (mg/$m^3$). The ratio between the measured or predicted actual field yield and the defined normal fish yield constitutes the base of this equation.

2. Lake pH will influence this ratio. For example, the reproduction and abundance of roach (*Rutilus rutilus*) is very sensitive to alterations in lake pH (Håkanson and Peters 1995). This is described by a dimensionless moderator acting upon the fish yield ratio. The moderator is 1 when the measured or predicted actual pH is equal to the normal pH. Since sensitive species like roach cannot reproduce at pH lower than 5, the moderator is calibrated to be zero for pH = 5 when normal pH is set to 7. This gives an amplitude value of +3.5. The moderator is:

$$Y'pH = (1+3.5\cdot(Actual\_pH/Normal\_pH-1) \qquad (1)$$

So, the fish yield ratio is first set to (see also table 4):

$$FYR = (Actual\_TP/Normal\_TP)\cdot Y'pH \qquad (2)$$

It is, however, unrealistic to assume that the changes in fish biomass can be directly related to mean monthly changes in lake TP-concentrations - it takes much longer to change fish biomasses than lake TP-concentrations. To describe this in detail is a most complicated matter related to the size (length/weight/age) and species of the fish, lake temperature and many food-web characteristics. In this simplified approach, this problem is handled by means of an exponential smoothing function (see Håkanson and Peters, 1995). The smoothing function is written as: smth(input, averaging time, initial value), or, using a differential equation:

$$dx/dt = x(t) + (z(t)-x(t))/T^{0.75} \quad (3)$$

where

$z(t)$ = the input (a time series of data);
$x(t)$ = a selected start value plus a time series of data;
T = the mean, characteristic age (a selected time constant in months) of the given organism. The mean age of all types of fish is set to 18 months as a default value. As a comparison, the following corresponding figures could be used as typical values for other organisms: zooplankton 3 months, macrozooplankton and benthos 6 months, planktivorous fish (like small perch) 12 months, benthivores, omnivores and herbivores 18 months, piscivorees 24 months, and large piscivores 36 months.

Fig. 5 illustrates how this delay function approach works in a hypothetical case for the fish yield ratio if the age of the fish is set to 1, 3, 6, 12, 24 and 36 months and if there is a sudden decrease in lake total-P concentration from 60 to 20 µg/l. The idea with this example is to illustrate how eq. 3 works, and the realism of such a sudden drop in total-P concentration will not be elaborated.

In this approach, the input as well as the initial value is given by FYR and the default value for the characteristic age of the entire fish stock in the lake is set to 18 (months). This will smooth the response in the fish yield to changes in lake TP-concentrations. This change (or reaction time) is assumed to by much shorter for the benthic community than for the fish community, and the default characteristic age of benthos is set to 6 months. No smoothing function is used for phytoplankton (or for bacteria or microzooplankton). Since phytoplankton generally have a life span much shorter than 1 month (which is the calculation time used in this model), there is no need for a smoothing function.

The interpretation of the fish yield ratio is that the value is 1 for a normal lake. A ratio of 2 means that the fish yield is 2 times higher than the normal. The same approach is used for the plankton biomass index and the bottom fauna biomass index.

**The phytoplankton biomass ratio** (PBR) is based on the quotient between the actual (from measured or predicted data) and the given normal biomasses of phytoplankton (from table 1), i.e., $30 \cdot actual\_TP^{1.4}/30 \cdot normal\_TP^{1.4}$. This ratio is influenced by a dimensionless moderator for pH using an amplitude value of 2 (see Håkanson, 1996) , such that a lower pH implies a lower phytoplankton biomass.

**The bottom fauna biomass ratio** (BFBR) first gives the ratio between actual and normal biomass of benthos (in mg ww/m$^2$ from table 1), i.e., $810 \cdot actual\_TP^{0.71}/810 \cdot normal\_TP^{0.71}$. This ratio is influenced by a dimensionless moderator for pH using an amplitude value of 3, such that a lower pH implies a lower phytoplankton biomass.

The amplitude values describe the strength of the change in the index when the actual values depart from the normal values. The rationale for setting the highest amplitude values for the influence of pH for the fish yield ratio and lowest for the plankton biomass ratio is that fish (like roach) biomass is assumed to be the most sensitive to changing pH. These assumed values describe the methodological foundation of establishing ecosystem indices.

Any change in state variables will affect the given key functional groups in the ecosystem, and also the ratios. Such changes would also affect other important conditions in a lake, like the concentration of toxins in fish. Alterations in lake pH and productivity would also influence the concentration of radiocesium in fish (Håkanson et al., 1996). However, this is not accounted for by this index, but by the overall lake model, the VAMP-model, in the MAA-analysis.

Finally, the lake ecosystem index (LEI) is defined as:

$$LEI = (FYR+PBR+BFBR)/3 \quad (4)$$

Note that if FYR (or PBR or BFBR ) is < 1, then the model sets the value to 1/FYR. Thus, the LEI-value is always $\geq$ 1. LEI is never allowed to attain negative values, but any departure from 1 is always negative from an ecological point of view because it represents a departure from normal. A compilation of equations used to calculate the LEI-value is given in table 4.

## THE UTILITY FUNCTION

There are several possible ways to define the utility function for the multi-attribute analysis (MAA). The basic idea with the utility function is to give a scientific rational to implement LEI in the MAA-analysis in such a way that environmental, social and economic objectives can be compared. Different users would then arrive at the same results if they follow the given manual of the MAA-analysis. The utility function, u(x), should be directly related to the LEI-value and it should be a value between 0 and 1. Utility means in this MAA-analysis a measure of how good a countermeasure would be to minimise the ecological impact expressed by LEI. Therefore, if the optimal value of LEI is 1, this should correspond with the maximum "utility" u(x) = 1.0; at the other extreme u(x) should be 0.

However at which value for LEI should u(x) approach 0? The information in table 3 and fig. 1 can then used to determine an equation describing the relationship between LEI and u(x)? It is, as stated, evident that if lake TP-concentrations change by a factor of 10, i.e., if the ratio between actual and normal TP changes by a factor of 10, from minus 2*SD to + 2*SD, then, with a 95% certainty, there will be a shift from one trophic category to another, e.g., from oligotrophic to mesotrophic. There will also be associated changes in abundance and biomass of key functional groups. If one starts with a Actual/Normal ratio of 1 for lake TP, and, as all empirical data suggest, the log-normal frequency distribution for lake TP-concentrations (see OECD, 1982; Håkanson and Peters, 1995) is used, then one can define the utility function directly from the probability function for logarithmic TP-values (see fig. 6A). This means that there is a 95% probability that TP-values should lie within 4 and 40 µg/l if the normal TP-value is set to 20; and there is a 52% probability that the TP-values would fall between 13.3 and 30 (see fig. 6B). One can then determine the probability function, set equal to (1-u(x)), , for different ratios of Actual/Normal. In this study, the following ratios 1.1, 1.2, 1.3, 1.4, 1.5, 1.6, 1.7, 1.8, 1.9, 2.0, 2.2, 2.4, 2.6, 2.8 and 3.0 were tested for symmetric divergences from the mean value for log-transformed TP-concentrations.

The relationship between LEI and u(x) is given in fig. 7. One can note that there is an almost perfect fit ($r^2 = 0.9998$) between the values given by the following algorithm and the calculated data derived from the Monte Carlo simulations:

$$u(x) = (1 - 1.21*((LEI-1)/LEI)^{1.3})^2 \quad (5)$$

The basic approach, (LEI-1)/LEI, is logical since u(x) should be 1 for LEI = 1. From fig. 7B one can note that the utility function, u(x), approaches 0 as LEI approaches 5.

## PRACTICAL APPLICATION

Figures 8 and 9 illustrate how the model works, showing changes in lake total-P concentrations (eutrophication, oligotrophication and fertilization as a remedial measure) and changes in lake pH (acidification and liming as a remedial measure), respectively

The driving variable in fig. 8A is the given changes in actual total-P (TP) concentrations (pH is kept constant at 7). First, there is a phase where actual TP is equal to normal TP, then there is a phase of comparatively slow increases in TP (to illustrate "creeping" eutrophication), then there is an instant reduction in TP back to normal values, then a period of adjustment, followed by a period with very low actual TP-values, a period of normal TP and, finally, a period where actual TP is set to 60 µg/l. The normal TP is keep at 20 µg/l throughout the simulation. Note that this simulation is NOT intended to illustrate realistic changes in lake TP-concentrations, but rather how the model works. The results for the fish yield ratio (curve 1), the bottom fauna biomass ratio (curve 2) and the phytoplankton biomass ratio (curve 3) are given in fig. 8B. The instant changes in the driving TP-variable (curve 4 in fig. 8A) cause much slower changes in the fish yield ratio, similar slow changes for the bottom fauna biomass ratio, but rapid changes for the phytoplankton biomass. Also note that a change of 2 in any of these ratios means a double (or halved ) change in the biomass as compared to the normal. The corresponding changes in LEI and the utility function, u(x), are shown in fig. 8C. The natural or reference value for the lake ecosystem index (LEI) is 1 when lake TP is normal (20 µg/l in these simulations). If LEI is geater than 2, u(x) is smaller than 0.26; if LEI > 4, then u(x) > 0.03. If LEI > 4, this implies a very significant alteration of the entire lake ecosystem.

Fig. 9 gives an example for lake pH when TP is kept constant at 20 µg/l. First, there is a phase where actual pH is equal to normal pH (=7), then there is a phase of comparatively slow decreases in pH (to illustrate the acidification process), then there is an instant increase in pH back to normal values (e.g., from lake or wet land liming), then a period of adjustment, followed by a period with very high actual pH-values. The implications of such pH changes are given in fig. 9B for the fish yield ratio (curve 1), the bottom fauna biomass ratio (curve 2) and the phytoplankton biomass ratio (curve 3). In this case, the instant changes in pH also cause also slow changes in the fish yield ratio and the bottom fauna biomass ratio. The corresponding changes in LEI and the utility function, u(x), are shown in fig. 9C.

## CONCLUDING REMARKS

These simulations demonstrate how variability in the two state variables, total-P and pH, influence the lake ecosystem index and the utility function. Very high values of LEI and low values of u(x) are obtained if the actual values of all these two state variables depart significantly from the normal (initial or reference) values. These examples show that the approach has potential in multi-attribute analysis, water management and radioecology because it would facilitate overview and provide an aim and direction to evaluate the ecological impact of remediation.

## ACKNOWLEDGEMENTS

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Table 1. Selected regressions illustrating the key role of lake total phosphorus in predictive limnology. Many biological variables whose determination normally require extensive and expensive field and laboratory work may be estimated or predicted from one key, abiotic state variable, total-P (in $mg/m^3$). Some variables may be predicted with great precision, others with much less. n = number of lakes used in the regression. ww = wet weight. dw = dry weight. From Peters (1986) and Håkanson and Peters (1995).

| y-value | Equation | Range for TP | $r^2$-value | n | Units |
|---|---|---|---|---|---|
| Chlorophyll | $y=0.073 \cdot TP^{1.44}$ | 3-300 | 0.96 | 77 | $mg \cdot m^{-3}$ |
| Max. prim. prod. | $y=20 \cdot TP-71$ | 7-200 | 0.95 | 38 | $mg\ C \cdot m^{-3} \cdot d^{-1}$ |
| Mean prim. prod. | $y=10 \cdot TP-79$ | 7-200 | 0.94 | 38 | $mg\ C \cdot m^{-3} \cdot d^{-1}$ |
| Nanoplankton | $y=17 \cdot TP^{1.3}$ | 3-80 | 0.93 | 23 | $mg\ ww \cdot m^{-3}$ |
| Phytoplankton | $y=30 \cdot TP^{1.4}$ | 3-80 | 0.88 | 27 | $mg\ ww \cdot m^{-3}$ |
| Fish yield | $y=7.1 \cdot TP$ | 10-550 | 0.87 | 21 | $mgww \cdot m^{-2} \cdot yr^{-1}$ |
| Macrozooplankton | $y=20 \cdot TP^{0.65}$ | 3-80 | 0.86 | 12 | $mg\ ww/m^3$ |
| Zooplankton | $y=38 \cdot TP^{0.64}$ | 3-80 | 0.86 | 12 | $mg\ ww \cdot m^{-3}$ |
| Bacteria | $y=0.90 \cdot TP^{0.66}$ | 3-60 | 0.83 | 12 | $mill. \cdot ml^{-1}$ |
| Net plankton | $y=8.6 \cdot TP^{1.7}$ | 3-80 | 0.82 | 23 | $mg\ ww \cdot m^{-3}$ |
| Fish | $y=590 \cdot TP^{0.71}$ | 10-550 | 0.75 | 18 | $mg\ ww \cdot m^{-2}$ |
| Crustacean plankton | $y=5.7 \cdot TP^{0.91}$ | 3-300 | 0.72 | 49 | $mg\ dw \cdot m^{-3}$ |
| Microzooplankton | $y=17 \cdot TP^{0.71}$ | 3-80 | 0.72 | 12 | $mg\ ww \cdot m^{-3}$ |
| Blue-greens | $y=43 \cdot TP^{0.98}$ | 8-1300 | 0.71 | 29 | $mg\ ww \cdot m^{-3}$ |
| Benthos | $y=810 \cdot TP^{0.71}$ | 3-100 | 0.48 | 38 | $mg\ ww \cdot m^{-2}$ |

Table 2. Characteristic features in lakes of different trophic categories. Note that there is a great overlap between the different categories, such that in oligotrophic lakes the concentrations of total-P may vary within a year from very low to high values (modified from OECD, 1982; Håkanson and Jansson, 1983).

A.

| **Trophic level** | **Primary prod.** (g $C/m^2$*yr) | **Secchi** (m) | **Chl-a*** ($mg/m^3$) | **Algal vol.*** ($g/m^3$) | **Total-P** ----($mg/m^3$)---- | **Total-N** | **Dominant fish** |
|---|---|---|---|---|---|---|---|
| Ultra-oligotr. | < 6 | >10 | <1.5 | <0.4 | <5 | <200 | Char, Trout |
| Oligot. | 5-30 | 12-5 | 1-3 | 0.2-0.8 | 4-15 | 150-350 | Trout, Whitefish |
| Mesot. | 25-60 | 7-2 | 2-10 | 0.5-1.9 | 10-30 | 300-500 | Whitefish, Perch |
| Eut. | 40-200 | 4-1 | 6-35 | 1.2-2.5 | 20-100 | 350-600 | Perch, Roach |
| Hypert. | 130-600 | <2 | 30-400 | 2.1-20 | >80 | >600 | Roach, Bream |

* = Mean value for the growing period (May - Oct.)

Table 3. Compilation of results from OECD (1982) concerning mean values and standard deviations for lakes belonging to different trophic classes. The differences (Diff.) are calculated for the logarithmic values (e.g., 0,22=1,12-0,90) for oligotrophic lakes (minus 1*SD - mean value).

| **Total-P in µg/l** | | **Absolute values** | | | **Log. values** | | **Diff. for** | **log. values** | |
|---|---|---|---|---|---|---|---|---|---|
| | **Oligotr.** | **Mesotr.** | **Eutrophic** | **Oligotr.** | **Mesotr.** | **Eutr.** | **Oligotr.** | **Mesotr.** | **Eutr.** |
| **Mean** | 8 | 26,7 | 84,4 | 0,90 | 1,43 | 1,93 | | | |
| **Minus 1 SD** | 13,3 | 49 | 189 | 1,12 | 1,69 | 2,28 | 0,22 | 0,26 | 0,35 |
| **Plus 1 SD** | 4,85 | 14,5 | 48 | 0,69 | 1,16 | 1,68 | 0,22 | 0,27 | 0,25 |
| **Range** | 3,0-17,7 | 10,9-95,6 | 16,2-386 | | | | | | |
| **n** | 21 | 19 | 71 | | | | **Mean abs. diff. of log. values=** | 0,26 | **≈ 0,25** |

Table 4. A compilation of equations used for the Lake Ecosystem Index, LEI.

**1. Fish yield ratio** FYR = SMTH(FYR´, 18, FYR´)

FYR´ = (Actual_fish_yield/Normal_fish_yield)*Y'pH
Fish yield = 7.1*TP (in mg ww/$m^2$*yr)
Moderator 1 for pH: Y'pH = 1+3.5*(Actual_pH/Normal_pH-1)
Averaging time set to 18 months for fish as a default value for the entire fish community.
SMTH = smoothing function.

**2. Phytoplankton biomass ratio** = PER

PBR = (Actual_biomass/Normal_biomass)*Y"pH
Phytoplankton biomass = 30*$TP^{1.4}$ (in mg ww/$m^3$)
Y"pH = 1+2*(Actual_pH/Normal_pH-1)
No smoothing is used for phytoplankton when dt is set to 1 month.

**3. Bottom fauna biomass ratio** BFBR = SMTH(BFBR´, 6 BFBR´)

BFBR´ = (Actual_biomass/Normal_biomass)*Y'"pH
Bottom fauna biomass = 810*$TP^{0.71}$ (in mg ww/$m^2$)
Y'"pH = 1+3*(Actual_pH/Normal_pH-1)
Averaging time set to 6 months as a default value for the entire benthic community..
SMTH = smoothing function.

**LAKE ECOSYSTEM INDEX**, LEI = (FYR+BFBR+PBR)/3

If FYR (or BFBR, PBR and CFHg) < 1, then 1/FYR instead of FYR.
Thus, LEI is always ≥ 1. A departure from LEI = 1 is generally negative.

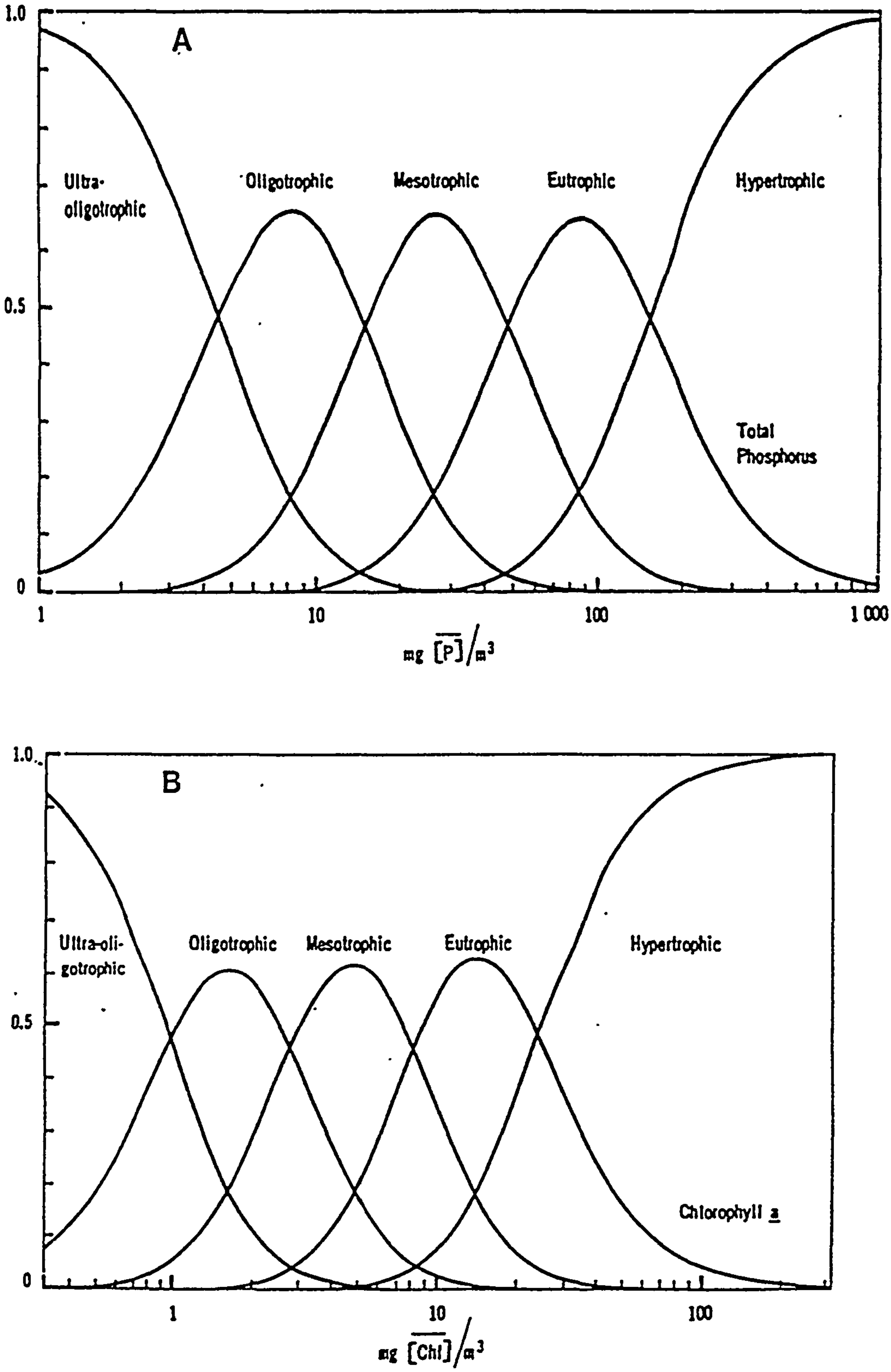


Fig. 1 Probability distributions for different trophic categories. From OECD (1982).

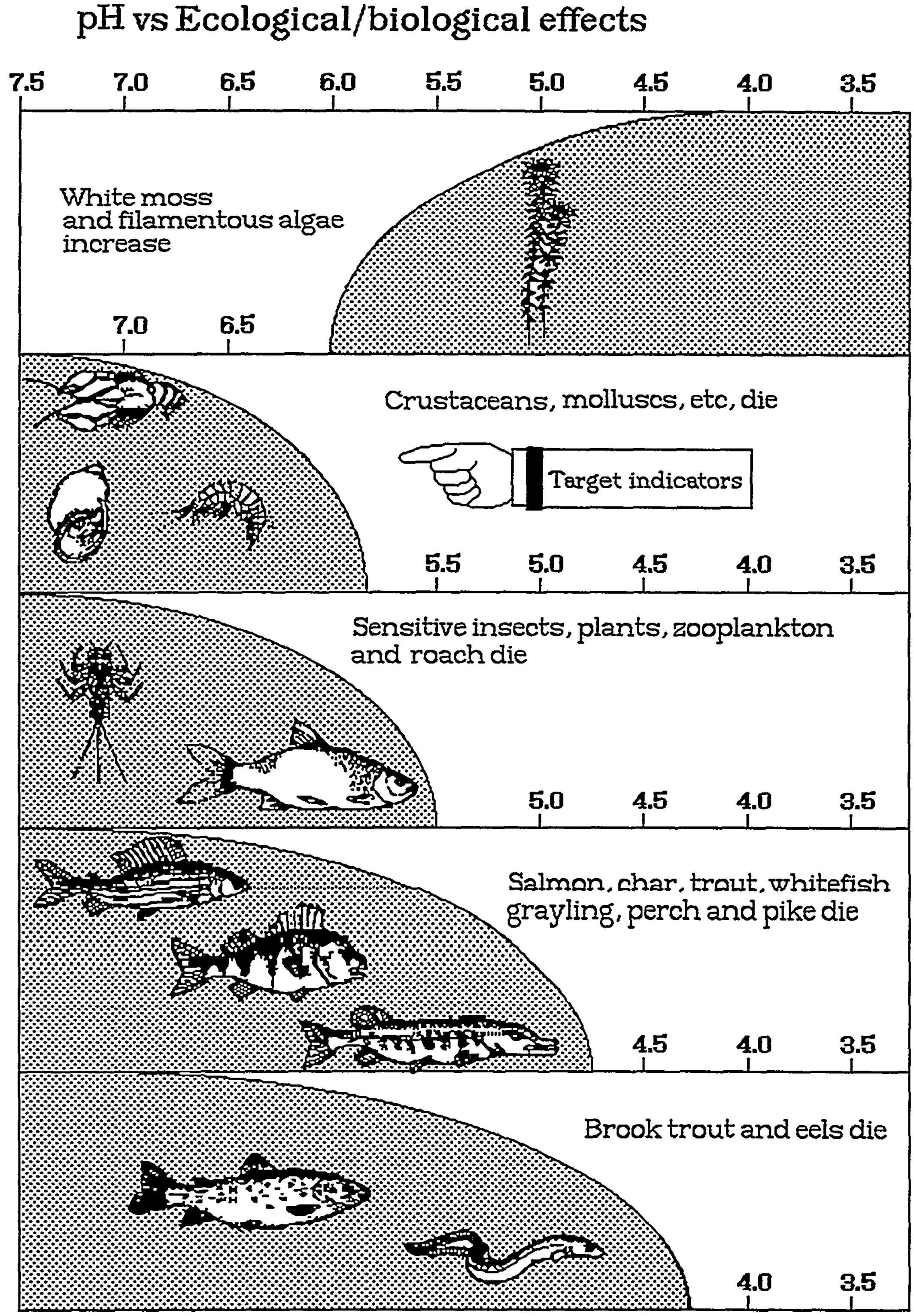


Fig. 2. Illustration of target organisms using the example of biological and ecological effects of lake acidification. The figure shows examples of key functional groups and target organisms for acidification. Crustaceans react rapidly to changes in pH, whereas certain fish such as brook trout and eels do not die until acidification is far advanced. White moss (e.g., Spagnum) and filamentous algae should not normally be found in these lakes and the abundance of such species indicates ecological effects of acidification. From Håkanson and Peters (1995).

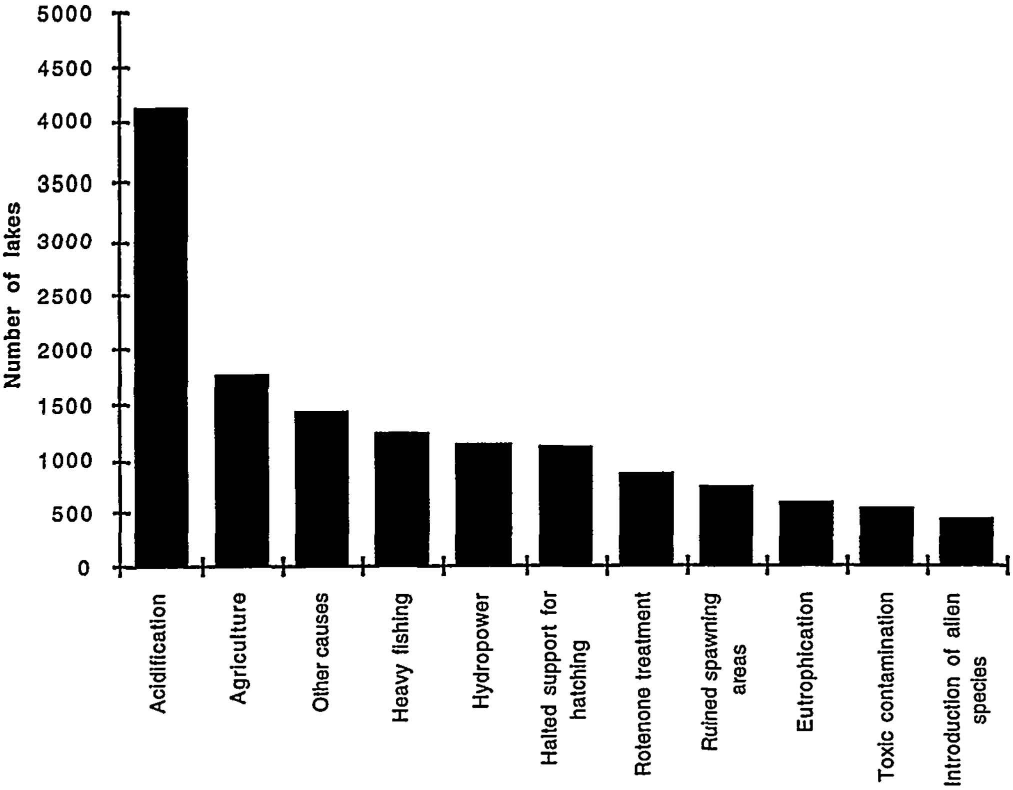


Fig. 3. Different causes for the reduction and/or extinction of fish communities in Swedish lakes. Note that acidification in most important among these causes, but that also eutrophication due to agriculture has given rise to significant changes in the structure of fish communities in Swedish lakes. From EPA (1997).

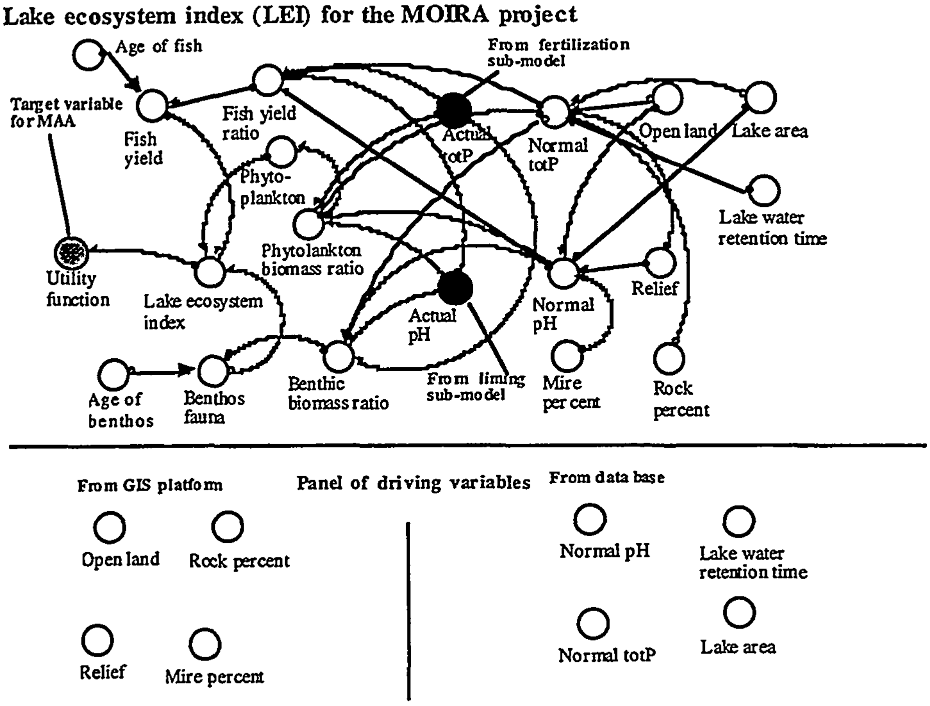


Fig. 4. Interactions among the basic components of the modified lake ecosystem index and its relationship to the state variables and the panel ofdriving variables

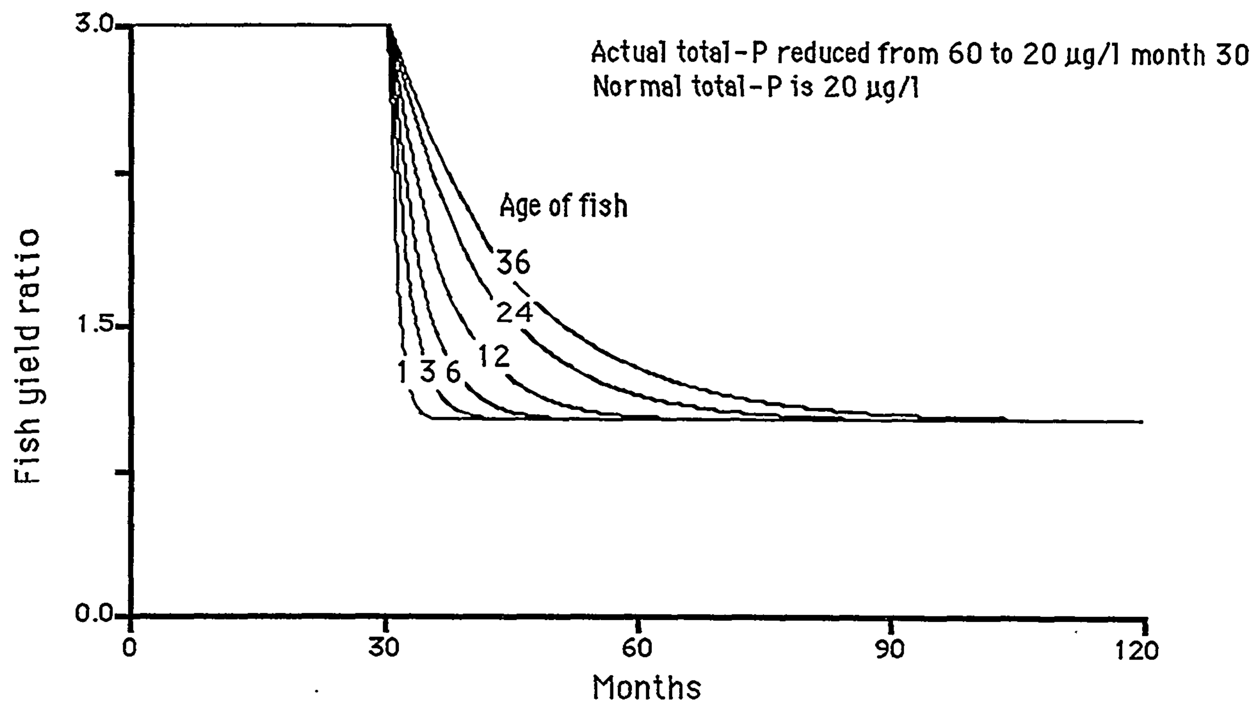


Fig. 5. Illustration of the smooting function for the fish yield ratio using different fish ages (from 1 to 36 months) in a hypothetical scenario when lake total-P concentation is set to decrease from 60 to 20 µg/l month 30.

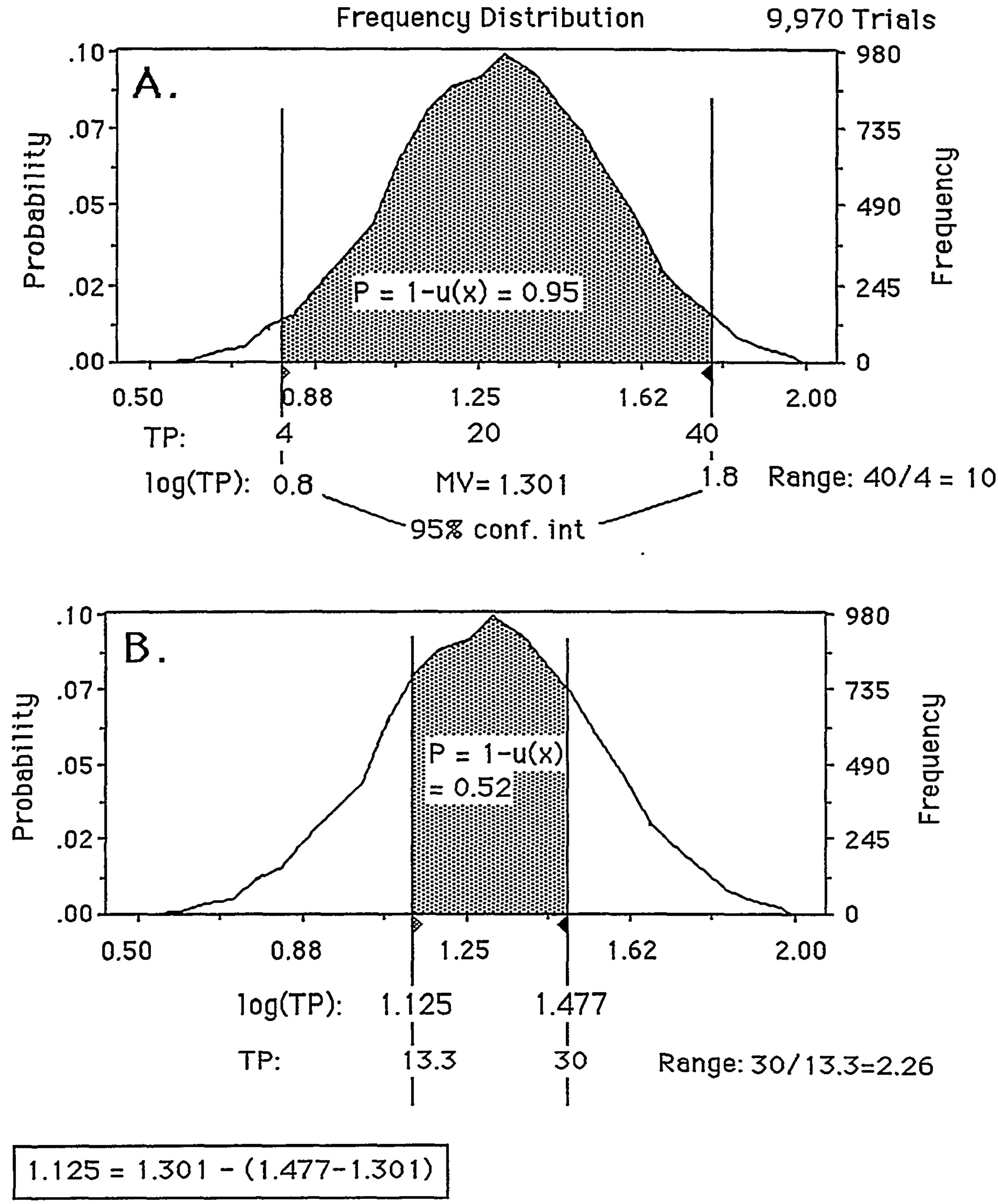


Fig. 6. Monte Carlo simulations to define the utility function, u(x), which is set be be complementary to the probability function.
A. Symmetric definition for the log-normal frequency distribution. Example when lake total-P is set equal to 20. Then there is a 95% certainty that TP-values would fall between 4 and 40 µg/l. u(x) is 1-0.95 = 0.05.
B. Symmetric definition for the log-normal frequency distribution. u(x) = 1-0.52 = 0.48 if TP-values fall between 13.3 and 30 µg/l.

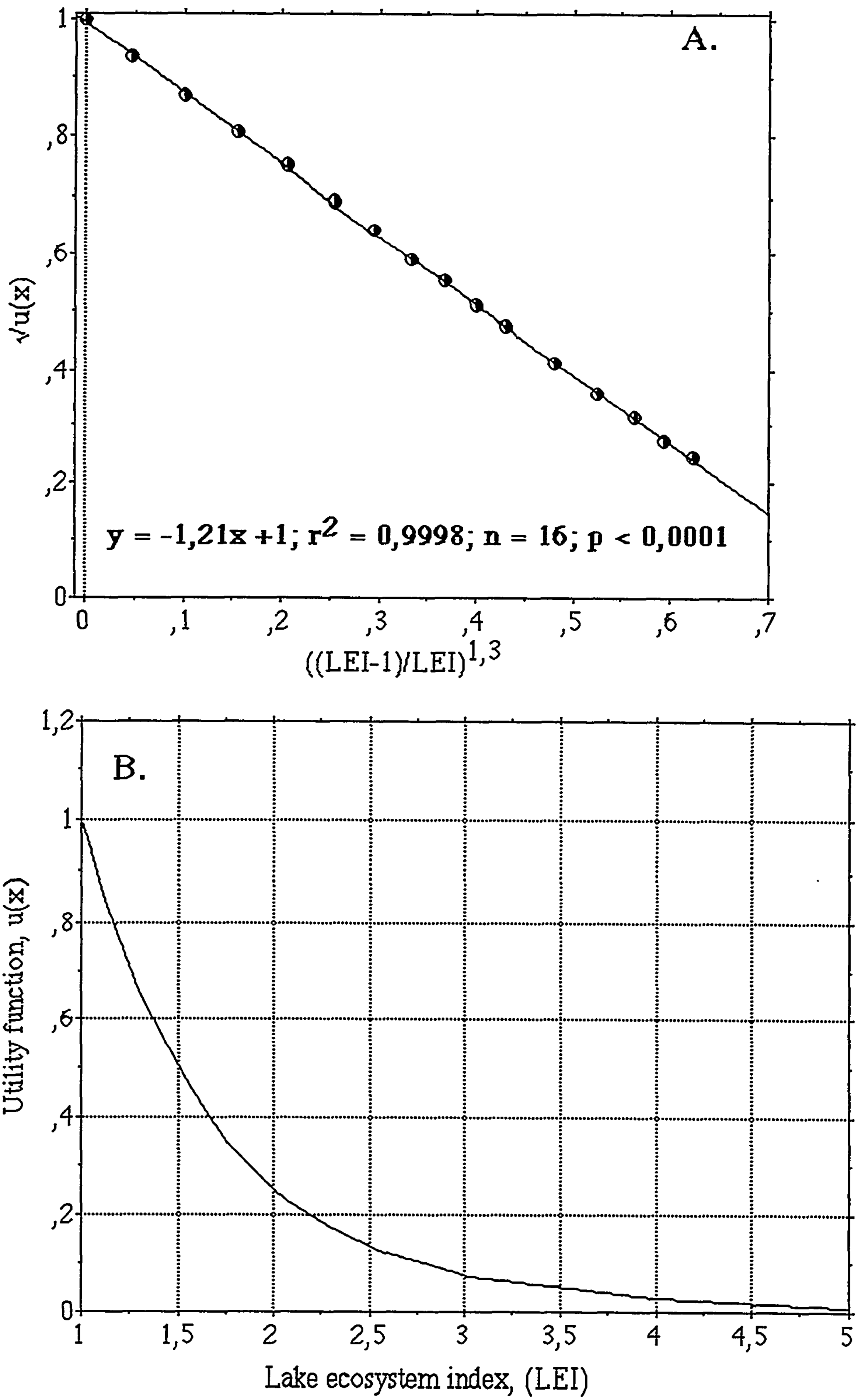


Fig. 7. The relationship between the lake ecosystem index (LEI) and its utility function, u(x).
A.The regression. B. The utility curve

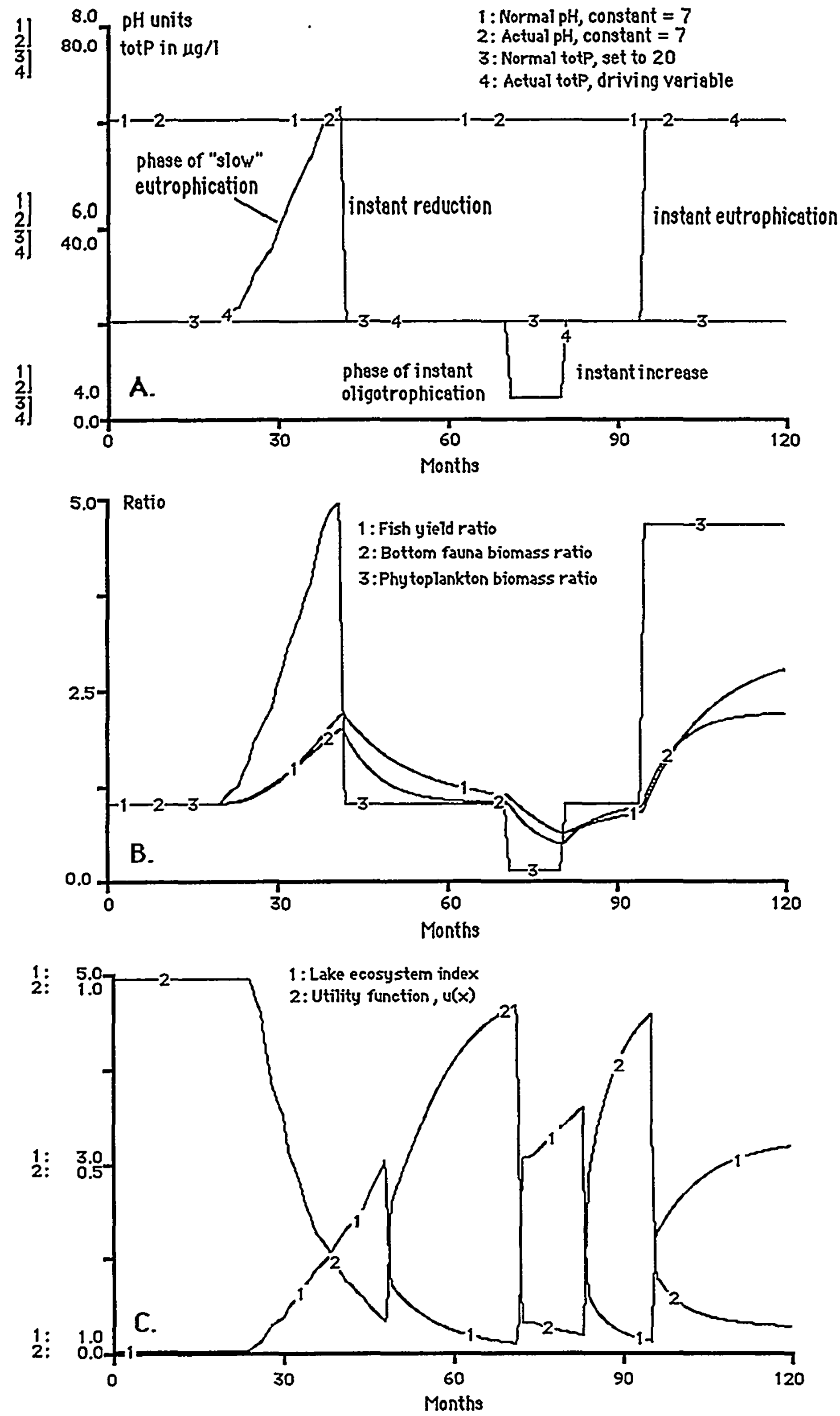


Fig. 8. Simulations and sensitivity tests with the model for the lake ecosystem index for changes in actual lake total-P concentration. Normal values for total-P = 20 µg/l and pH = 7.

A. Illustration of the assumed actual lake TP-concentrations (curve 4; the driving function in these simulations).

B. Response for the fish yield ratio (curve 1), the bottom fauna biomass ratio (curve 2) and the phytoplankton biomass ratio (curve 3).

C. Response for the lake ecosystem index (LEI, curve, 1) and the utility function (u(x), curve 2).

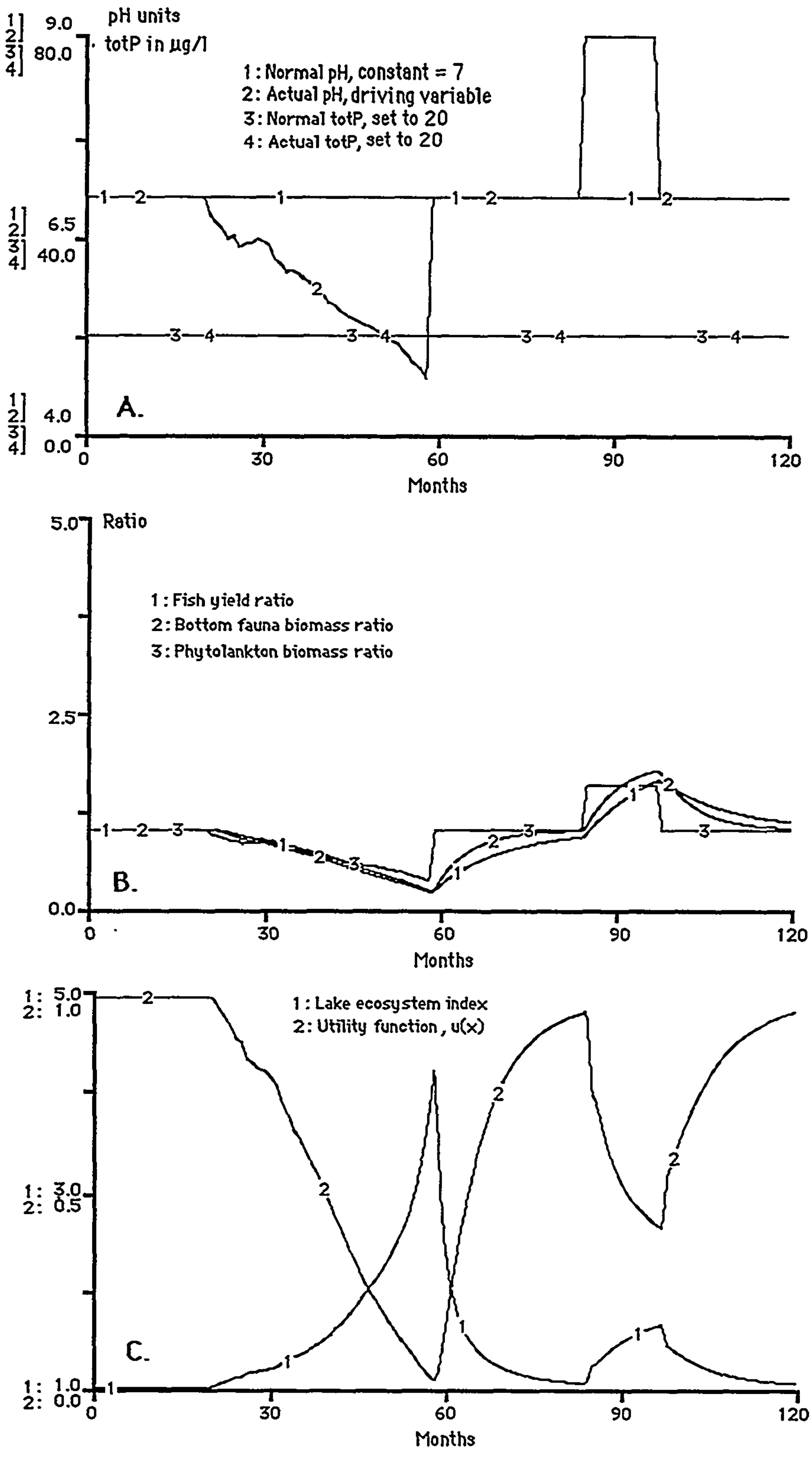


Fig. 9. Simulations and sensitivity tests for the lake ecosystem index for changes in actual lake pH. Normal values for total-P = 20 µg/l and pH = 7.

A. Illustration of the actual pH-values (curve 2; the driving function in these simulations).

B. Response for the fish yield ratio (curve 1), the bottom fauna biomass ratio (curve 2) and the phytoplankton biomass ratio (curve 3).

C. Response for the lake ecosystem index (LEI, curve, 1) and the utility function (u(x), curve 2).

# RADIONUCLIDE TRANSPORT IN URBAN DRAINAGE SYSTEMS

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## OVERVIEW OF URBAN DRAINAGE SYSTEMS

The development of a model for simulating the behaviour of radionuclides in urban drainage systems was a necessity since new potential pathways could originate from the presence of radioactivity in such environmental locations. Between other potential pathways for human exposure, that could result in the contamination of drinking water supplies, fisheries and the marine environment, are irrigation with sewage effluent, the disposal of sewage sludge to agricultural land, which allows radionuclides to enter the food chain and the recycling of sewage effluent to streams and rivers and eventually to the sea.

The aim of the present report is to give an overview of the sewerage systems and sewage treatment, which has been useful for the identification of the most important parameters to be considered for the model: that is, the influence of chemical, physical and biological treatments, together with retention times in any process or operation.

### *Description of urban drainage systems.*

Urban drainage systems generally have two main components: **the sewerage system**, which is preceding the treatment, and **the sewage treatment plant**, where the treatment takes place. Both are described in the following sections.

### *Sewerage systems.*

There are three kinds of systems, namely:

i. *Separate systems.*- It consists of two separate pipe systems. One pipe system carry the water runoff for direct discharge to the watercourse without treatment; the other conveys the waste waters from domestic and industrial sources to the treatment plant.

ii. *Partially separate systems.*- There are also two pipe systems as above, but in this case the waste water pipe also carries some surface water, generally runoff from roofs and paved yards.

iii. *Combined systems.*- In this case there is just one single system of pipes where foul and surface water flows are carried together.

The most common system is the combined system though this varies from one country to another.

*Sewage treatment.*

Sewage treatment is a combination of physical, biological and chemical processes, designed for removing organic matter from solution. In each country, treatment plants have been designed to produce quality effluent concerning the standards recommended by the correspondent regulatory commission on sewage disposal. Today such standards for discharges are set to give a particular environmental quality in the receiving water and so they can vary.

Individual methods are normally classified in unitary physical operations, unitary chemical processes and unitary biological processes.

Unitary operations are those methods in which the application of physical forces prevails. Some of them are: planing, mixing, flocculation, sedimentation, flotation and filtration.

Unitary chemical processes are carried out by means of chemical reactions, such as: precipitation, gas transferring, adsorption and disinfecting. Finally, unitary biological processes are methods in which pollutants, mainly biodegradable organic matter and nitrogen, are removed by biological activity.

The unitary operations and processes may be grouped into the following sets of treatments:

- *Preliminary treatments*, consisting of a set of operations carried out at the beginning of the treatment plants, aimed to avoid hazards to the rest of facilities that constitute the plant. These processes are included into the physical treatments since their working is based on these kind of principles.

- *Primary treatment*, where suspended solid matter, oil and fat or dissolved solids are settled out by strictly physical processes such as difference in density between particulate and water, osmotic pressure, heat transfer, electric transfer, etc.

- *Secondary treatment*, which comprises chemical treatments where the removal of pollutants is carried out by a chemical reaction; and biological treatments, where biodegradable organics contained in the waste water, either industrial or domestic, are separated through biological processes or reactions that involve the action of micro-organisms.

- *Advanced treatment*, which is only used in very specific cases when high quality water is required or water has to have special characteristics; it comprises physical, chemical and biological processes.

- *Sludge treatments*, through the processes described above, an amount of sludge will be produce in a very diluted way. Sludge contains pollutants removed from water by the preceding treatments, so it is necessary to concentrate it before it can be definitively disposed. Most common processes are: dewatering, disinfecting and stabilisation.

Figure 1 displays a diagram for a typical treatment plant.

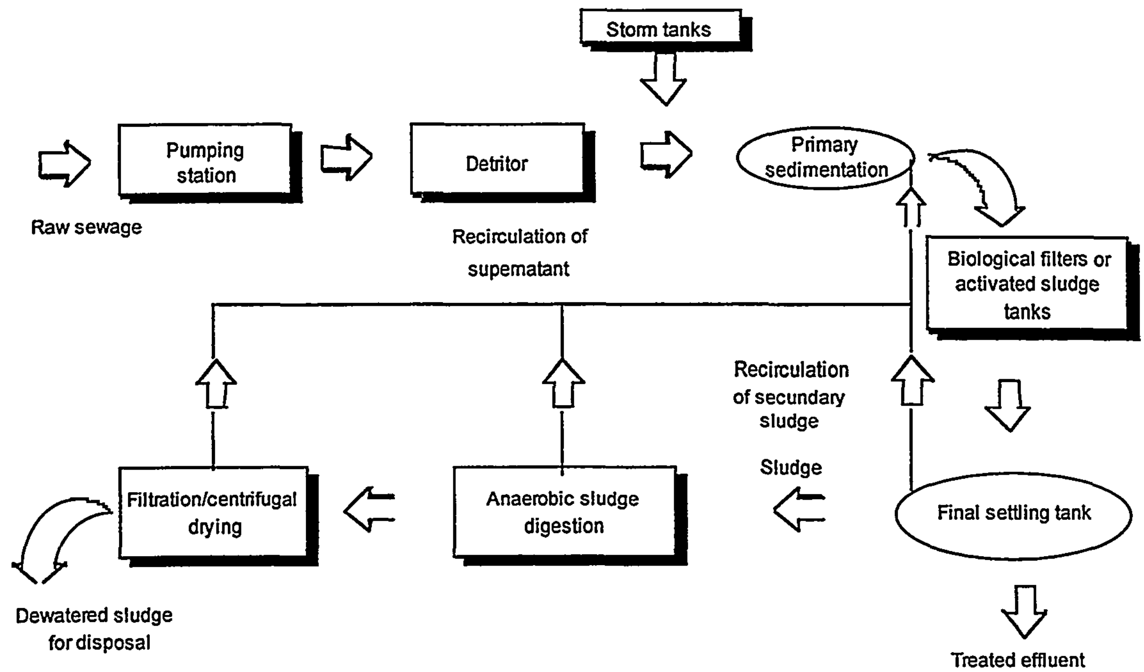


*Figure 1. Flow diagram for typical sewage treatment plant*

*Preliminary and primary treatment.*

Preliminary treatment involves passing the raw sewage through fixed bar metal screens in order to remove large solids such as rugs and fibrous materials, all of them of organic nature. This makes it possible to readily settle out a fairly clean grit and also to separate organic matter from the grit so that the final product is innocuous.

Preliminary treatment is applied to the total sewage flow arriving at the treatment plant. On the other hand, as primary treatment is not designed for treating all the entering flow after preliminary treatment, any excess flow is diverted to the storm tanks, prepared to receive overflows in case of heavy rainfall. They are usually designed to give primary treatment and are very similar to sedimentation tanks.

Sedimentation is performed in rectangular or circular basins, where the waste water is held quiescent to permit particulate matter to settle out of suspension. Primary sedimentation tanks are devised to remove as much as possible of the settleable organic materials present in the sewage, together with any mineral matter not extracted by the detritor. These tanks are often designed to give a retention time of approximately 6 hours, as it is found that, in most cases, the settleable solids are deposited in about 3 hours.

*Secondary treatment.*

After primary treatment, the remaining effluent contains non-settleable suspended solids and dissolved organic matter. Then, the secondary treatment involves bringing the settled sewage into contact with bacteria and other micro-organisms. Therefore, under appropriate conditions, organic matter is removed both from solution and from suspension, leaving a relatively pure liquid. There are essentially two forms of biological treatment:

1. Treatment by percolating filters.

2. Activated sludge process.

A typical percolating filter consists of a bed or granular, evenly-graded, durable materials such as stones, or pieces of rock or coke. The settled sewage is sprinkled over the surface of the bed and trickled downwards continually, leaving part of the void space for air. The micro-organisms, which perform the oxidation process, form a thin film on the surfaces of the filter medium. After a few days, the film is able to remove much of the organic matter from the sewage when the surplus accumulated film becomes detached from the bed and emerges at the base of the filter, suspended in the purified liquid. The suspended matter, known as "humus", is removed by secondary sedimentation. Where synthetic media have been introduced to replace the use of stone, the term *biological towers* is commonly employed.

The activated sludge process is based on the same principle as the percolating filter except that, in this case, the settled sewage is held in aeration tanks, with flocculent cultures of micro-organisms -the activated sludge freely suspended in the liquid. Aeration stimulates the micro-organisms in the sewage to grow into a culture capable of oxidising the organic matter in a period of a few hours. A proportion of these organisms tend to clump together and the flocculated mass can be readily separated by sedimentation. Further quantities of raw sewage can be added and the procedure repeated, until a sufficient concentration of flocculent, activated sludge has been built up to permit operation under continuous flow conditions. After aeration, the sludge is removed from the purified liquid by sedimentation in the final settling tanks. Most of this sludge is recirculated to the inlet end of the aeration unit, while the excess is drawn off for further treatment or disposal. In most cases, as with the percolating filter, the clear supernatant is now suitable for discharge to the watercourse.

*Advanced treatment.*

Advanced treatment is used if a higher quality effluent is required, beyond the standards, as may be the case of water supply for towns being taken downstream of a treatment plant. The main objective is the removal of suspended matter not settled out during primary or secondary sedimentation.

Advanced treatment methods are: activated coal, reverse osmosis and ionic exchange.

*Sludge treatment.*

The objectives of sludge treatment are:

1. To render the sludge more stable, less objectionable and more suitable for disposal;

2. To reduce the volume of sludge produced by separating some water from the solid matter.

Primary sludge, because of their fibrous and coarse nature, thickens and dewaters readily. However, secondary sludge is more difficult to deal with by itself and is usually mixed with primary sludge before treatment. Therefore, the mixed sludge is often subject to gravity thickening in tanks to reduce the liquid volume. Alternatively, activated sludge may be thickened separately using dissolved air flotation.

In the recent years, stabilisation of sludge before dewatering has become more common. Its success is related with the results of the stabilisation process on the volatile portion of the sludge. Pathogen survival, bad odour and putrefaction take place when the micro-organisms grow up on the organic portion of the sludge.

The most extended method of sludge stabilisation is anaerobic digestion. In the absence of air, bacterial action develops and results in some decomposition of the organic material into methane, hydrogen sulphide and hydrogen. Commonly the organic solids contents is reduced by about 50%. The rate at which sludge digestion processes occur depends upon temperature. Mesophilic digestion, where the sludge is heated to 35°C, is the one which is usually employed, particularly for large plants. The retention time in this kind of process is normally 30 days, although a shorter time is sometimes adopted. Another kind of treatment is thermophilic digestion in which the operating temperature is between 45-55°C, which is more rapid and gives greater pathogen reduction. However, this digestion is not frequently used.

Another widespread method is aerobic digestion, which requires retention times of 15-30 days in temperate climates and up to 50 days in cold climates. Alternative methods of stabilisation are treatment with lime and composting, either alone or mixed with a bulking agent such as straw, bark or domestic refuse. Approximately 70% of the sludge in Europe is subject to some form of stabilisation.

Dewatering of sludge is usually preceded by a conditioning stage, in order to improve its dewatering properties. This can be achieved by the addition of chemical compounds, such as iron, aluminium salts or polyelectrolytes.

## RADIONUCLIDE BEHAVIOUR IN URBAN DRAINAGE SYSTEMS.

Radioactivity entering the urban drainage networks may be returned to the 'above ground' systems either in the discharge of treated or untreated effluent to the receiving watercourse or concentrated in sewage sludge which may be disposed of in a number of ways, including application to agricultural land. That is why a quantification of the water entering the plant and the water that is spilled directly to the watercourse, in case of overflow, is very important.

Since most European countries are sewered by a mixture of combined and separate sewer systems, it is necessary to be able to handle both types of system.

For separate sewer systems, the situation is relatively simple. Since this work is only concerned with the fate of radioactive material in surface water runoff, it is only necessary to consider the storm sewer system, which discharges directly to the receiving watercourse.

For combined sewer systems, the situation is more complicated. During rainfall, the combined foul sewage and runoff water can enter different parts of the system, depending on the flow rate. In the majority of cases, all the flow is directed to the sewage treatment plant. However during heavy rainfall, part of the flow may be diverted to the storm tanks and only returned to the main process stream at the end of the storm. If the rainfall is prolonged and the storm tanks become full, then some of the diverted sewage will be spilled to the watercourse. Furthermore, if the flow rate approaches the maximum for the treatment plant, the combined sewer overflows will come into operation. From the overflows, the excess sewage is discharged directly to the watercourse, although there may be provision for part of the excess to be retained in overflow storage chambers. Most storm tanks and some storage chambers are designed to retain some settleable matter. From the treatment plant, some of the radioactivity carried in the sewage will be discharged to the watercourse with the treated effluent, while the remainder will be retained in the sewage sludge. Quantification of radionuclide migration in combined sewers will involve consideration of all these elements.

Ideally, such quantification would take into account the effects of the different processes carried out at the treatment plant, i. e. primary and secondary treatment and sludge treatment. However, there is frequently a significant degree of recalculation within the plant. For example, since activated sludge is difficult to handle by itself, it is often mixed with primary sludge by resettling in the primary sedimentation tanks. Also, it is common for the supernatant extracted during sludge treatment to be returned to the biological oxidation unit for further processing.

Should the discharge of radioactivity in untreated sewage to the watercourse prove to be a significant exposure risk, it will be important to take into account the time variation in the rainfall, since this determines the route and subsequent treatment of the sewage. Averaging of rainfall intensity and runoff rates over long time periods is likely to lead to under estimation of radioactivity entering the watercourse.

*Modelling radionuclide transport.*

Surface runoff entering the sewers will contain a proportion of solid matter. In combined sewers, the surface runoff will be mixed with the waste water which will also contain some solid material. Many radionuclides are easily adsorbed onto street sediments and other particulates. Therefore, the surface runoff will carry away to the sewers some radioactivity previously adsorbed onto surface sediments. Also soluble radionuclides in the runoff water may be taken up by particulate during transport in the sewers. Consequently, of the activity entering the sewers with the runoff, plus the waste water from domestic and industrial use, a proportion will be associated with the particulate material and a proportion will be in solution.

Another possibility is that radionuclides may be adsorbed onto the sewer walls, soil and pipes. The evolution of the radionuclides activity in these surfaces follows the radioactivity decay equation:

$$C(t) = C_0 \cdot e^{-\lambda \cdot t}$$

In case of overflow due to heavy rainfall some effluent will pass directly to the watercourse while the rest will pass to the primary treatment plant.

At the primary treatment plant, raw sewage contains both settleable and non-settleable solids. Modern primary sedimentation tanks are capable of removing up to 90% of the settleable solids that will be incorporated to primary sludge and, in so doing, will remove the radioactivity associated with these solids. The non-settleable solids (and any associated activity) will be carried over to the secondary treatment stage together with most of the soluble activity. The biological treatment processes will remove further activity, which becomes incorporated in the secondary sludge.

A fraction of the primary sludge will be recirculated to secondary treatment in case of activated sludge process.

The treated effluent will spill to the watercourse and the secondary sludge will pass to the sludge treatment plant.

The concentration of radioactivity in the sludge will therefore increase with further treatment. The increase in concentration will be related to the amount of water removed and the reduction in sludge volume. However, since some sludge processes are relatively slow the amount of activity present may be reduced due to radioactivity decay which may be appreciable for short-lived radionuclides.

From the sludge treatment, the supernatant will recirculate to secondary treatment and the final dewatered sludge will be ready for disposal.

For short-lived radionuclides, it is necessary to know the retention times for the treatment processes, in order to take into account of radioactive decay. Where the activity is associated with the liquid phase then the retention time will be of the order of one day. Where the activity is associated with the activated sludge, the mean residence time of the sludge may be estimated by dividing the total mass of activated sludge by the rate of growth. This time period is termed the sludge age and is typically about 8 hours. Sludge treatment is a much slower process and can take from 2 weeks to 2 months or even longer. However, where anaerobic digestion is followed by some form of dewatering, the average retention time is 4 weeks.

Figure 2 shows a compartment representation of the drainage system model for MOIRA system.

Drainage

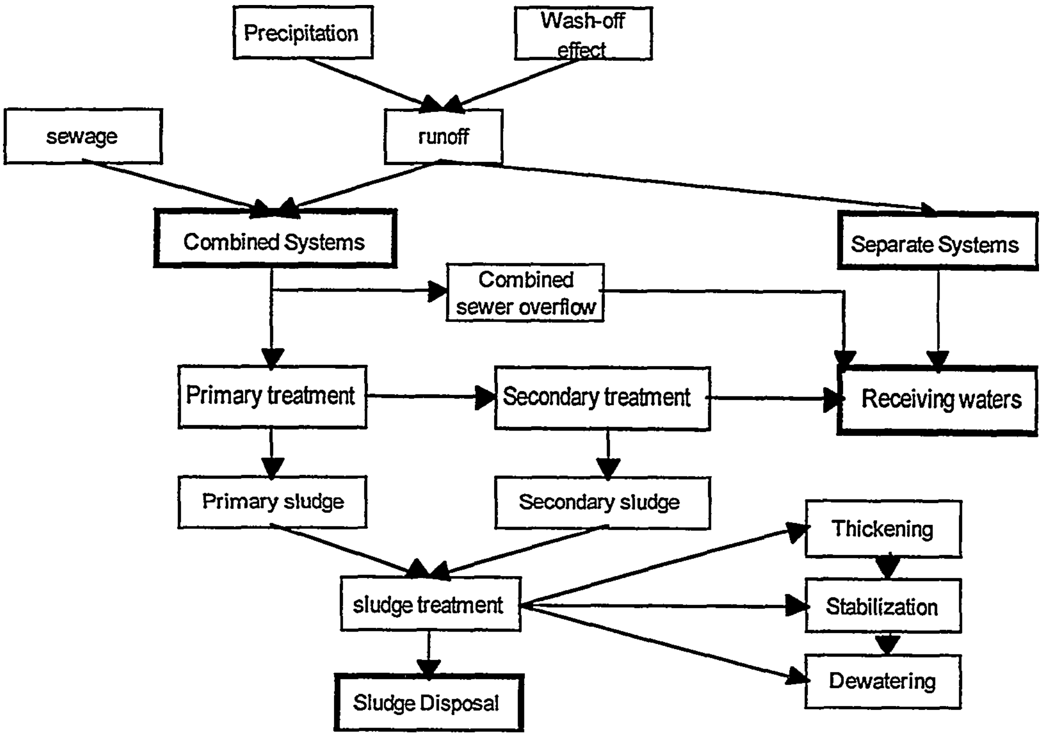


*Figure 2. Compartment representation for the urban drainage system model*

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# MODELLING THE BEHAVIOUR OF RADIONUCLIDES IN RIVERS

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## INTRODUCTION

Rivers are an important resource for human populations. They are often used to supply drinking and irrigation waters. In many cases, fish species living in rivers may constitute a significant dietary component. Therefore, following events involving radioactive contamination of the environment, appropriate actions are necessary to restore the quality of the water to levels suitable for the utilisation of river water resources.

Some experience on the effectiveness of the countermeasures to reduce the radiological risks due to river contamination were gained following the Chernobyl accident. A classification of the countermeasures applied to the Dnieper cascade was made by Voitsekhovitch, 1995. The interventions were classified according to the time of application:

- Emergency phase (2-3 months after the accident)
  - regulation of the flow of contaminated water;
  - water consumption ban;
  - purification of drinking water;
  - search for new non-contaminated groundwater supply wells.

- Early and later intermediate phase (1-5 years after the accident) actions included large scale engineering works such as building protective dikes to catch the contaminated urban runoff, excavating canal bed traps to reduce water velocity and increase sedimentation, building zeolite-containing dikes to adsorb radionuclides from water, building special drainage systems and walls to catch infiltrating contaminated waters and building dikes to reduce radionuclide migration from river flood-plains.

The countermeasures aimed at modifying the chemical characteristics of the waters in order to reduce the radionuclide uptake by fish species, although often appropriate for lacustrine systems, in general, are not effective for rivers due to the short water residence time.

The building of protective dikes may be ascribed to the wide and classical category of the "containment technologies" that may be successfully applied to reduce the migration of contaminants from restricted polluted areas (Smith, at al. 1995). Unfortunately in case of major accidents, such as the Chernobyl incident, these sorts of countermeasures may be very expensive, have a non negligible impact on the environment and do not guarantee useful results.

One of the goal of the MOIRA project will be the analysis of the effectiveness of the applied countermeasures to evaluate what intervention actions may be feasible and successful in practical circumstances.

To carry out such an evaluation it is necessary to take advantage of models for predicting the migration of radionuclides in rivers and the effects on the water contamination levels as a result of the applied countermeasures.

The goal of the present report is to describe a simple model for evaluating the migration of radionuclides in rivers.

## MODEL DESCRIPTION

The model is based on the quantitative analysis of migration processes occurring in an "elementary" water body composed of a sediment bed and of a surrounding water column containing a certain amount of suspended matter (see figure 1).

The main assumption of the model, described earlier in Monte (1993), is that a radioactive substance introduced into the water body dilutes in the water and reacts with the suspended matter and the upper layer of the bottom sediment (the sediment interface layer) in a time interval that is negligible when compared with the observation period and with the time rates of the migration of radionuclide to bottom sediments. It is also assumed that the process of radionuclide absorption on suspended matter and on the sediment interface layer may be modelled according to the well-know "$k_d$ concept" ($k_d$ = partition coefficient "particulate form/dissolved form" [$m^3$ $kg^{-1}$]) based on the hypothesis of a reversible, rapid equilibrium between the dissolved ($C_w$) and the adsorbed phases ($C_s$) of radionuclide

$$\frac{C_w}{C_s} = k_d \tag{1}$$

The migration of a radionuclide from water to sediment may be modelled by the diffusion equation. The interface layer was introduced to account for the boundary condition: the concentration in interstitial water in the upper layer of sediment is equal to the concentration in the water column above the sediment. If W is the total amount of radionuclide dissolved in water, we get

$$\begin{aligned} S_s &= k_d \, w_{sm} \, W \\ S_a &= k_d \, D_{il} \, A_L \, \delta W / V = k_d \, D_{il} \, \delta W / h \end{aligned} \tag{2}$$

In equation (2) $S_s$ is the total amount of radionuclide in the interface layer (Bq), $w_{sm}$ is the concentration of suspended matter in water (kg $m^{-3}$), $S_a$ is the total amount of radionuclide in the interface layer (Bq), $D_{il}$ is the thickness of the interface layer (m), $A_l$ is the surface area of the water body ($m^2$), $\delta$ is the density of the interface layer (kg $m^{-3}$), V and h are, respectively, the volume ($m^3$) and the average depth (m) of the water body.

The time variation of the radionuclide amount (T) in the three compartments - Water, sediment Interface Layer and Suspended Matter (WILSM) - may be simply calculated as a function of the time variation of W

$$\frac{dT}{dt} = R \frac{dW}{dt} \tag{3}$$

where R is

$$R = 1 + k_d \, w_{sm} + k_d \, D_{il} \, \delta / h \tag{4}$$

The variation in time of the total radionuclide amount in the WILSM system is equal to the algebraic sum of the radionuclide fluxes $Q_i$ (Bq $s^{-1}$) *to* and *from* the WILSM:

$$R\frac{dW}{dt}=\sum_i \pm Q_i \tag{5}$$

where we use "+" if the flux is *to* the WILSM system and "-" if the flux is *from* the WILSM. Dividing both members of equation (5) by V we get, after simple calculations,

$$\frac{dC_w}{dt}=\sum_i \pm\frac{Q_i}{R\,A_L\,h} \tag{6}$$

The product Rh may be decomposed into two terms

$$\begin{aligned} Rh &= h + h_\Delta \\ h_\Delta &= (k_d\, w_{sm} + k_d\, D_{il}\, \delta / h)h \end{aligned} \tag{7}$$

where $h_\Delta$ (the "incremental depth") has the dimension of a depth and accounts for the rapid interaction of the dissolved radionuclide with the sediment and the suspended matter. Evaluations of such a parameter for $^{137}Cs$ in some European lakes have been made in a previous study (Monte 1995). Using a value of approximately 6 m it was possible to carry out good approximate estimates of the initial $^{137}Cs$ concentrations in lake water following the Chernobyl accident. As $h_\Delta$ is proportional to $k_d$, the value of $h_\Delta$ for $^{90}Sr$ may be assumed to be negligible compared with the lake depth (measurements carried out in water and in sediment of some Italian lakes suggest that $^{90}Sr$ $k_d$ is of the order of 0.1 $m^3\ kg^{-1}$).
Equation (6) becomes

$$\frac{dC_w}{dt}=\sum_i \pm\frac{\phi_i^*}{h_{eff}} \tag{8}$$

where $h_{eff}$ is $h+h_\Delta$. $\phi^*_i$ are the fluxes of radionuclide *to* the WILSM (or *from* the WILSM) divided by the water body surface (Bq $s^{-1}\ m^{-2}$). Equation (8) states that the first derivative of the concentration of the dissolved radionuclide may be simply calculated as the algebraic sum of input terms, the radionuclide input to water, per square metre, divided by the effective depth. $\phi^*_i$ may be related to other quantities, such as the radionuclide deposited in the sediment and the radionuclide concentration in water, by specific equations. The simplest phenomenological equation relates, by a linear relationship, the radionuclide flux (Bq $m^{-2}\ s^{-1}$) to the concentration of the radionuclide in a compartment or to the radionuclide deposit. The relevant proportionality constants have, respectively, the dimension of a velocity (m $s^{-1}$) or of a rate ($s^{-1}$). Such a formulation of the migration process from the water to the sediment and vice-versa, is conceptually equivalent to the discrete form of the diffusion equation of a substance through the sediment.
Subdividing a river into infinitesimal "elementary" boxes of length dx, we can write the following equation of the radionuclide balance (see figure 2):

$$\frac{\partial C_W(x,t)}{\partial t} = -\frac{1}{h_{eff}(x)l(x)}\frac{\partial}{\partial x}\left[C_W(x,t)\xi(x)\Phi(x,t)\right] - \frac{v\,C_W(x,t)}{h_{eff}(x)} +$$
$$-\frac{v_S\,C_W(x,t)\,k_d\,w_{ssd}}{h_{eff}(x)} + \frac{R(x,t)}{h_{eff}(x)l(x)} - \frac{C_W(x,t)P(x)\xi(x)}{h_{eff}(x)l(x)} + \quad (9)$$
$$+\frac{k_{sw}\,D_{ep}(x,t)}{h_{eff}(x)} - \lambda\,C_W(x,t)$$

where v and $v_s$ are, respectively, the transport velocities of the dissolved and particulate radionuclide to sediment, $C_w(x,t)$ is the dissolved radionuclide concentration (Bq $m^{-3}$) at time t and point x , l(x) is the river width (m). $\Phi(x,t)$ is the water flux ($m^3\ s^{-1}$), $h_{eff}(x)$ is the effective depth (m) of the river, x is the point of observation (distance from the contamination source in m), R(x,t) represents the radionuclide contribution from the catchment per unit length of the river (Bq $m^{-1}\ s^{-1}$), P(x,t) is the withdrawal (e.g. for irrigation purposes) of water from the river in $m^3\ s^{-1}\ m^{-1}$, $\lambda$ is the radioactive decay constant ($s^{-1}$) and $\xi(x)=1+k_d w_{sm}$
The terms in the second member of the previous equation are as follows: the amount of radionuclide migrating from the catchment to the water body divided by the surface area; the radionuclide deposition onto the water body surface; the radionuclide migrating, per square meter and per second, from the bottom sediment to the water; the amount of radionuclide (divided by the surface area) removed, per second, by the downstream transport; the radionuclide migrating, per second and per square metre, from water to bottom sediment. The following equation controls the migration of the radionuclide from water to sediment and through the sediment:

$$\frac{d\,D_{ep}(x,t)}{dt} = v\,C_W(x,t) + v_S\,C_W(x,t)\,k_d\,w_{sm} - k_{sw}\,D_{ep}(x,t) + \quad (10)$$
$$-k_{ds}\,D_{ep}(x,t) - \lambda\,D_{ep}(x,t)$$

Dep is the radionuclide deposit (Bq $m^{-2}$) in the bottom sediment layer, $k_{sw}$ is the migration rate from the sediment to the water and $k_{sd}$ is the migration
The river may be subdivided into a set of boxes. The discrete equations are as follows:

$$\frac{d\,C_W(i,t)}{dt} = -\frac{1}{h_{eff}\,l(i)}\left[C_W(i,t)\xi(i)\Phi(i,t) - C_W(i-1,t)\xi(i-1)\Phi(i-1,t)\right] +$$
$$-\frac{v\,C_W(i,t)}{h_{eff}} - \frac{v_S\,C_W(i,t)\,k_d\,w_{ssd}}{h_{eff}} + \frac{R(i,t)}{h_{eff}\,l(i)} - \frac{C_W(i,t)P(i)\xi(i)}{h_{eff}\,l(i)} + \quad (11)$$
$$+\frac{k_{sw}\,D_{ep}(i,t)}{h_{eff}} - \lambda\,C_W(i,t)$$

$$\frac{d\,D_{ep}(i,t)}{dt} = v\,C_W(i,t) + v_S\,C_W(i,t)\,k_d\,w_{sm} - k_{sw}\,D_{ep}(i,t) + \quad (12)$$
$$-k_{ds}\,D_{ep}(i,t) - \lambda\,D_{ep}(i,t)$$

where the index "i" identifies the order of the box ((first box: i=1; second box: i=2; etc.). The processes of diffusion in water were not included in the model as they are considered to be negligible compared to the transport processes occurring in rivers. Equations (11) and (12) may be numerically solved by a variety of software tools such as Ithink™ (Software from High Performance Systems Inc., 45 Lyne Road. Hanover NH).
The model has been validated in many circumstances. For example, it was applied for predicting $^{137}Cs$ and $^{90}Sr$ migration through the Dnieper cascade reservoir during the VAMP project and through the outlet of contaminated volcanic lakes in central Italy (Monte et al., 1997).
Some outputs of the model compared with experimental data collected in Khakovskoje and Kievskoje reservoirs are also reported in figures 3 and 4.

## CONCLUSIONS

The evaluation of the effectiveness of countermeasures for restoring the economic exploitation of river systems contaminated by radioactive substances requires the use of suitable validated models aimed to predict the migration of radionuclides.
The simple model described above is based on the analysis of the radionuclide transport processes occurring in rivers. The validation shows that the model may be an effective tool for predicting the radionuclide concentrations in water in practical circumstances.

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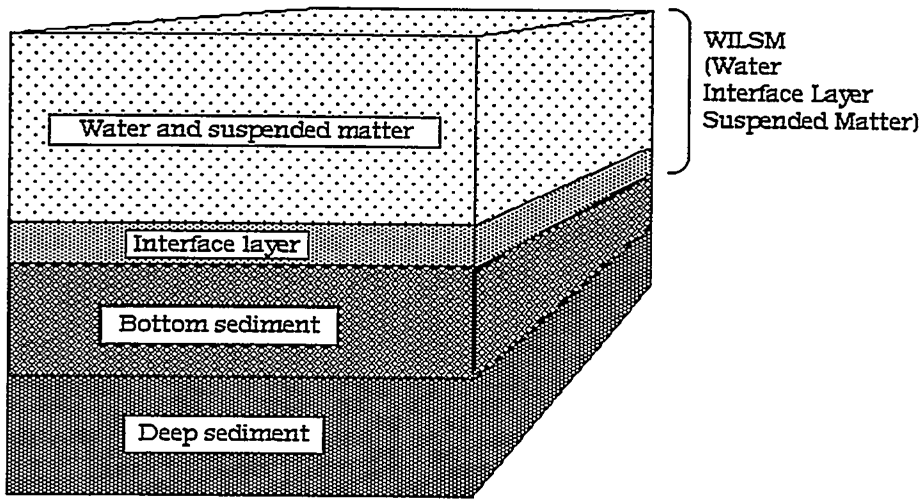


Figure 1 - The elements of the water body box in the river model

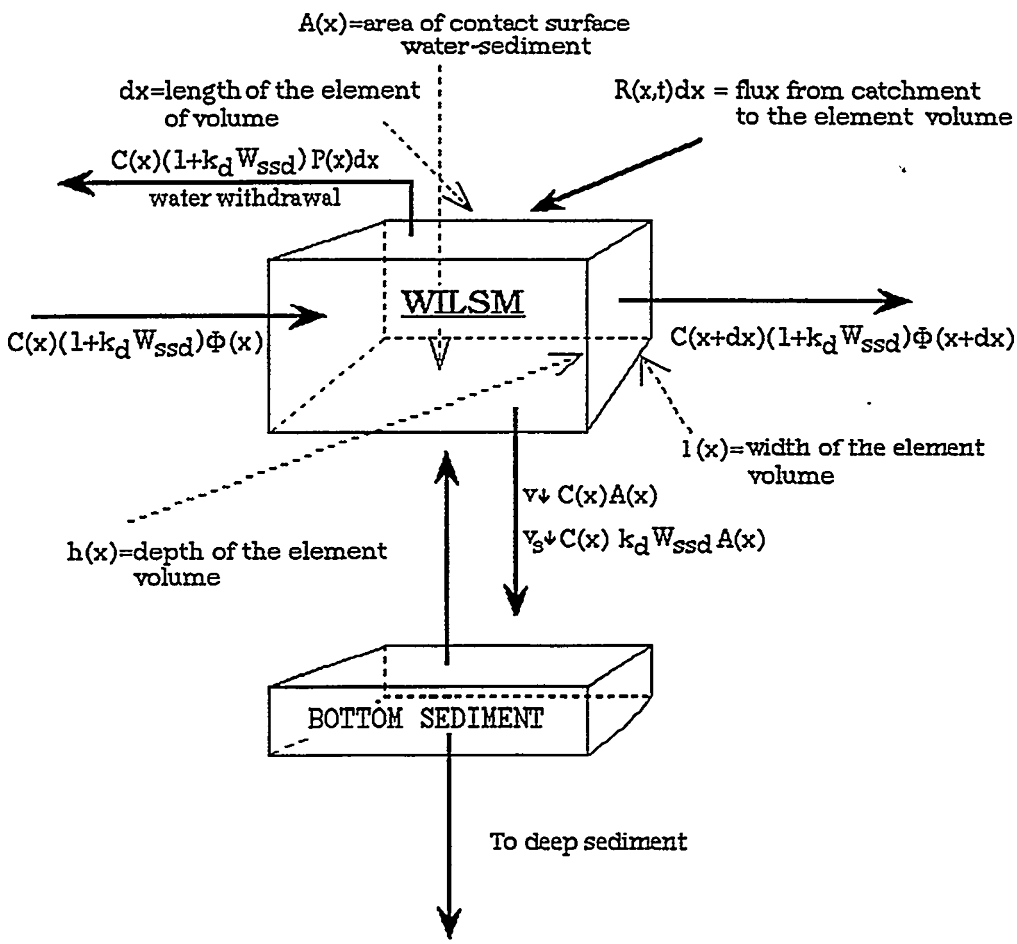


Figure 2 - Radionuclide fluxes in the river model

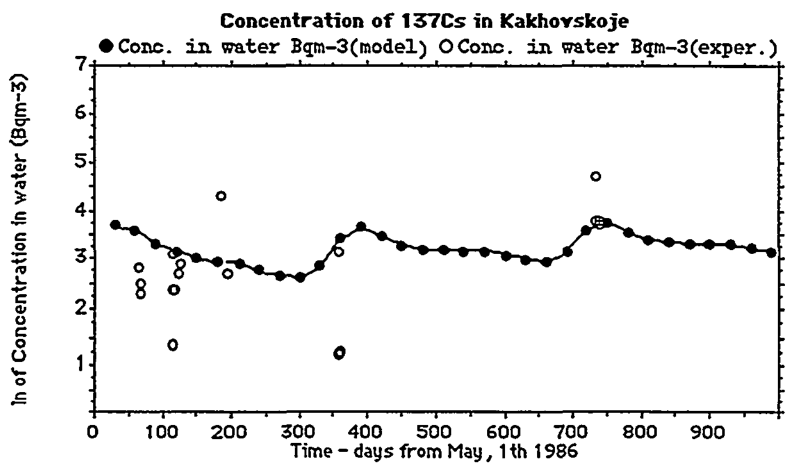


Figure 3 - Model validation. Dnieper reservoir cascade (Kakhovskoje Reservoir)

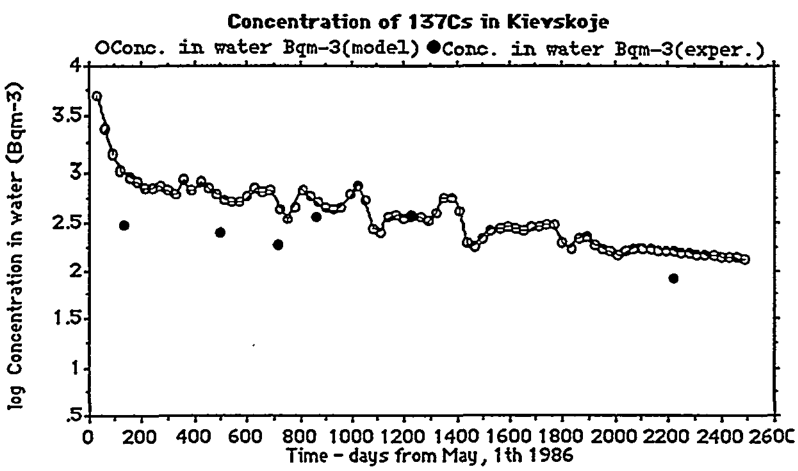


Figure 4 - Model validation. Dnieper reservoir cascade (Kievskoje Reservoir)

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