Journal of Hazardous Materials 61 (1998) 329–336

Risk and decision analysis of groundwater protection alternatives on the European scale with emphasis on nitrate and aluminium contamination from diffuse sources

Lars Rosén [a,*], David Wladis [a], Dominique Ramaekers [b]

[a] *Department of Geology, Chalmers University of Technology, S-412 96 Gothenburg, Sweden*
[b] *TECHNA Consult, 2 Rue du Baleau, B-1342 Limelette, Belgium*

Abstract

Stochastic simulation of nitrate and aluminium contamination of groundwater from diffuse sources on the European scale was performed. The results were used for two purposes in this paper: (1) to describe possible economical implications resulting from the uncertainties in contaminant modelling on the European scale and (2) to perform societal monetary decision analysis on alternatives to reduce contamination risks. The risk was treated as a probabilistic cost for exceeding existing water quality standards. A risk–cost minimization model was used for the decision analysis, considering all investment costs and risks of each alternative. The analysis was performed using the Netherlands as a study area and the cost of failure was based on the value of groundwater as a drinking water resource. The risk reduction alternatives included in the analysis were two emission reduction scenarios given by the IIASA RAINS model: the Current Reduction Plans (CRP) scenario and the Maximum Feasible Reductions (MFR) scenario. The study indicates that the uncertainties result in prediction intervals for the economical outcomes of the two scenarios of 2.7 to 5.1 billion ECUs over a 15-year simulation period. The study also indicates that the CRP scenario is the economically most advantageous alternative, given the economical assumptions made.

Keywords: Risk and decision analysis; Groundwater protection; Nitrate and aluminium contamination

* Corresponding author.

0304-3894/98/$19.00
PII S0304-3894(98)00140-X

1. Introduction

Contamination of soil and groundwater due to landuse practices and acid deposition is of major environmental concern in Europe. Increased nitrate and aluminium concentrations in groundwater result primarily from diffuse source emissions from industry, traffic, and agriculture. High concentrations of these substances can cause problems relating to human health, the environment, and technical constructions, which may all have significant economical consequences. Because of the complexity of hydrogeologic systems, unevenly distributed emission sources, and climatic variations, concentrations of nitrate and aluminium in groundwater on the European scale cannot be predicted with complete certainty. Use of models in environmental decision-making without considering the uncertainties may lead to economically unfavourable decisions.

The purpose of this paper was (1) to analyze possible economical implications resulting from uncertainties in modelling nitrate and aluminium contamination on the European scale and (2) to perform societal monetary decision analysis on alternatives to reduce contamination risks. Contamination is when the concentration exceeds the maximum admissible concentration levels. As a basis for the analysis, stochastic simulation of nitrate and aluminium contamination was performed using a process-oriented hydrological model, SMART2 [1]. The modelling was performed on natural land in the Netherlands and provided results of nitrate and aluminium leaching to phreatic groundwater.

2. Sources of nitrate and aluminium contamination

Increased aluminium and nitrate concentrations in groundwater are due to nitrogen and sulfur deposition causing nitrate leaching and acidification of soil–groundwater systems. If the deposition of nitrogen in a terrestrial system exceeds the capacity of the system for nitrogen uptake in the biomass, nitrate leaching will occur. Nitrate leachate may deteriorate groundwater as a drinking water source and can cause increased leaching of cations from the soil, i.e. acidification [2]. Acidification due to deposition of sulfur dioxides, nitrogen oxides, and ammonia can result in mobilization of aluminium if the pH gets below a value of 4.2. In some countries, corrosion due to acidification has resulted in increased levels of copper of the piping systems [3]. The increased levels of nitrate and aluminium in European groundwater resources clearly causes potential health risks and economical consequences.

3. Groundwater policy issues within the EC and risk reduction alternatives

Groundwater quality is presently regulated at the European level by ‘Directive 80/68/EEC’ concerning the protection of groundwater against pollution caused by certain dangerous substances. Its purpose is to prevent or limit the direct or indirect introduction to the groundwater of certain dangerous substances. This directive is deemed to be repealed when the draft ‘Directive establishing a Framework for Community Action in the field of water policy’ is adopted by the Council. Concerning groundwater, the proposal aims (Article 1) at further preventing deterioration and enhancing the status of water, promoting sustainable water consumption and thereby contributing to the sustainable use of resources.

In addition, an 'action programme' for integrated groundwater protection and management has been adopted by the European Commission in 1996. The action programme, while not legally binding, aims at better integrating water policy into other policy areas such as regional, industrial and agricultural policy. It should be noted that groundwater is indirectly protected by several Community legislations such as the 'Nitrates Directive (91/676/EEC)', the 'Regulation on agri-environmental measures (No. 2078/92)', the 'Limitation of emissions from large combustion plants (88/609/EEC)', the 'Sulfur content of certain liquid fuels (93/12/EEC)', and the 'Integrated Pollution Prevention and Control (IPPC) Directive (96/61/EC)'. The maximum admissible concentrations (MAC) relating to drinking water quality, which were the risk criteria used in this study, are set in 'Directive 80/778/EEC'. MAC for nitrate and aluminium are 50 mg/and 0.2 mg/l, respectively.

The related costs of different reduction actions in terms of sulfur dioxide (SO_2), nitrogen oxides (NO_x) and ammonia (NH_3) emission and deposition rates can be studied using the 'Regional Air Pollution Information and Simulation (RAINS)' model [4] developed by International Institute for Applied Systems Analysis (IIASA) in Laxenburg, Austria. In RAINS, different 'emission scenarios' are defined to study deposition and related costs from different action plans. For SO_2 and NO_x, an emission scenario is composed of an 'energy pathway' and an emission control strategy. For NH_3, the energy pathway is replaced by an 'agricultural activity'. An energy pathway describe the sectoral use of the different fuel types over time. A sector is defined as a group of similar emission sources, which consumes energy (fuel) and releases emissions of SO_2 and/or NO_x. An emission control strategy is a definition of emission control measures applied to the different emission source categories in a country, i.e. an instruction of when, how and how much to reduce emissions (RAINS 7.2 manual).

Two emission control scenarios for NO_x, SO_2 and NH_3 were studied in this paper. The Current Reduction Plans (CRP) scenario is based on an inventory of officially declared national emission ceilings [5]. For NH_3, the CRP scenario is not applicable. Instead, the Netherlands Acidification Abatement Plan (NAP) scenario was used. Regarding SO_2, the CRP emission target level for the year 2000 is already reached in the Netherlands, and consequently, the reduction rate was set to 0 and no costs were assigned to SO_2 reduction. The costs for the CRP alternative is thus a combination of CRP as estimated from the RAINS model for NO_x and NAP for NH_3. The second alternative is the Maximum Feasible Reductions (MFR) scenario. This scenario simulates the full implementation of all available technologies for emission control and allows for quantification of the possible progress toward full achievement of critical loads, as stipulated by the Council of European Commission [5].

4. Overview of decision analysis

In this section, a short background to monetary decision analysis is given, primarily based on the review paper by Freeze et al. [6]. Decision analysis in a monetary context is performed in order to choose the economically most favourable alternative among a given number of alternatives. In a decision problem, the benefits, costs, and risks of each alternative are taken into account by defining an 'objective function', ϕ_i, for each alternative $i = 1, \ldots, n$. The objective function should reflect the specific problem *and* the preferences of the decision-maker, and thus, varies according to the key variables involved. The objective function has the general form:

$$\Phi_i = \sum_{t=0}^{T} \frac{1}{(1+r)^t} \left[B_i(t) - C_i(t) - R_i(t) \right] \tag{1}$$

where B_i are the benefits of alternative i in year t [ECU], C_i are the costs of alternative i in year t [ECU], R_i are the risks, or probabilistic costs, of alternative i in year t [ECU], r is the discount rate [decimal fraction], and T is the time horizon [years]. The objective function represents the net present value of the alternative i. To make the most economically most favourable decision, the alternative i that maximizes the objective function is chosen. The risk term is defined as:

$$R = P_f(t) C_f(t) \tag{2}$$

where P_f is the probability of failure [decimal fraction] in year t and C_f is the economical consequences associated with failure in year t [ECU]. Failure is defined with respect to some compliance level that has to be met, e.g., MAC for consumption water. The C_f term includes the costs that arise *if* the groundwater becomes contaminated, e.g., water treatment costs or the extractive value of the resource [7]. A simpler type of the objective function, a 'risk–cost minimization model', may be appropriate if the benefits do not depend on the costs and risks. The risk–cost objective function can be expressed as:

$$\Phi_i = \sum_{t=0}^{T} \frac{1}{(1+r)^t} \left[C_i(t) + R_i(t) \right]. \tag{3}$$

The costs term includes all costs associated with efforts made to reduce the probability of failure, e.g., to reduce emissions. To make the economically most favourable decision, the purpose is to minimize the risk–cost objective function, i.e., to find the optimal risk (Fig. 1). In this context, actions that are more costly than the risk-reduction they give cannot be justified.

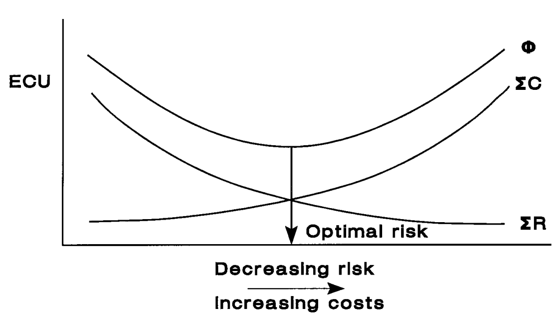

Fig. 1. Risk–cost minimization. The concept of optimal risk (after Ref. [6]).

In societal decision making [8], social costs are used. Social costs are the sum of private costs, which include production and distribution costs, and external costs, which are costs paid for by others than the producer/consumer. It was noted by James et al. [9] that risk-based decision analysis is, in general, more valuable on the large scale than for detailed decisions. This is because the big-picture decisions are the most important economically and may also require much less effort and mathematical complexity than detailed problems. As also pointed out by James et al. [9], effective communication is particularly needed when decision-making involves many and diverse stakeholders, including those who may not have the background or time to understand all the technical issues of the problem.

5. Probability of failure estimations

SMART2, as used in this project, gives predictions of nitrate and aluminium contamination for 5×5-km grid cells. Each grid cell was divided into 25 1×1-km subgrid cells (Fig. 2a). The model gives a concentration value for each subgrid cell, resulting in a distribution of concentrations for each large grid cell. Given the specific MAC value and the concentration distribution provided by SMART2, the area fraction of each large grid cell that exceeds the compliance levels can be calculated. This is referred to as the area of impact, A:

$$A = \frac{f}{k} \tag{4}$$

where f is the number of subgrid cells exceeding MAC and k is the total number of subgrid cells in the large grid cell. A conceptual description of the concentration distribution, MAC, and A for each large grid cell is shown in Fig. 2b. In order to analyze how the uncertainties of the input parameters to the models propagate to the

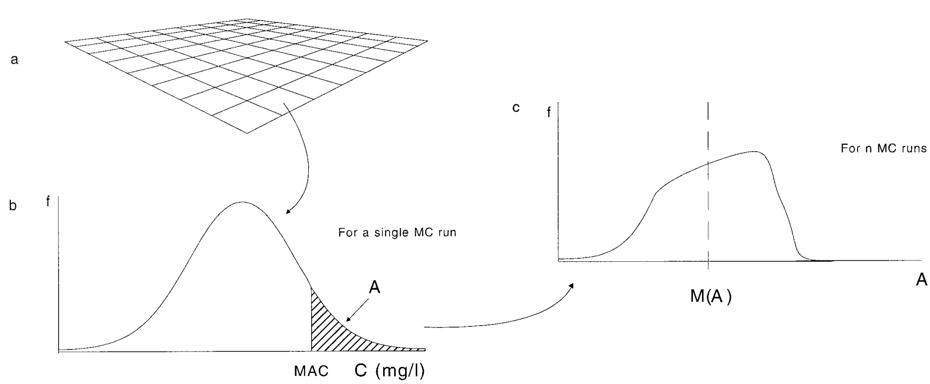

Fig. 2. Schematic outline of the results provided from the stochastic modelling of nitrate and aluminium: (a) grid map, (b) the concentration distribution for a single Monte Carlo run, including a realization of A, and (c) the probability distribution of A based on the entire set of Monte Carlo runs and with the position for the sample median value, $M(A)$, indicated.

output, Monte Carlo analysis is used. This yields a probability distribution of A for each large grid cell, as described in Fig. 2c.

The probability distribution of A is not likely to be normal because (1) A is restricted to have a value between 0 and 1 which in most cases leads to a skewed distribution, (2) the distribution for A may differ between different investigation areas, due to differences in land use, soil types, and so on, and (3) the number of Monte Carlo runs may be limited due to the complexity of the models, which results in a small sample. For these reasons, it was therefore decided to use a non-parametric approach to obtain a robust statistical handling.

6. Cost of failure estimations

The decision analysis was made with respect to non-compliance with drinking water standards as the primary criterion (Directive 80/778/EEC). If contamination above MAC occurs, it is assumed that the water has lost some of its value, since it will not be possible to use directly for drinking purposes. The C_f was, in this study, based on the following assumptions: (1) the value of the water is equal to the water price and (2) about 10% of the recharge is used for drinking water production in private or public wells. The percentage of drinking water in relation to the recharge was based on the assumptions that the natural land provides groundwater to the population in relation to its areal proportion of the country and that each person uses 200 l/day. These are rough assumptions that need to be refined for a more precise decision analysis. On the other hand, for decision analysis on the European scale, the information available will, in many cases, be of a general character and based on rough assumptions.

7. Results and discussion

Calculations of the risk were made for three time steps; year 1995, year 2000, and year 2010. Interpolation between the time steps yielded an objective function for the entire 15-year period. The calculations of the objective function were made adopting an approach of equality between generations and a discount rate of 0 was therefore used. Fig. 3 shows the prediction intervals of the objective functions for the Netherlands with respect to the CRP and MFR scenarios, using the 10% and 90% percentiles. The prediction intervals of the objective function for aluminium are 2.7 and 3.4 billion ECUs for the CRP and MFR scenarios, respectively, over the 15-year period. For nitrate, the corresponding figures are 4.8 to 5.1 billion. Thus, the uncertainty is larger for nitrate than for aluminium. For both aluminium and nitrate, the CRP alternative appears to be more favourable, given the assumptions made on the economical factors in the decision analysis.

It should be noted that the results are preliminary and that the economical estimations of the C and C_f terms are not final. However, the preliminary results indicate that the uncertainty of the hydrogeological model outputs leads to a variation in the order of several billion ECUs of the expected monetary value of each of the simulated alterna-

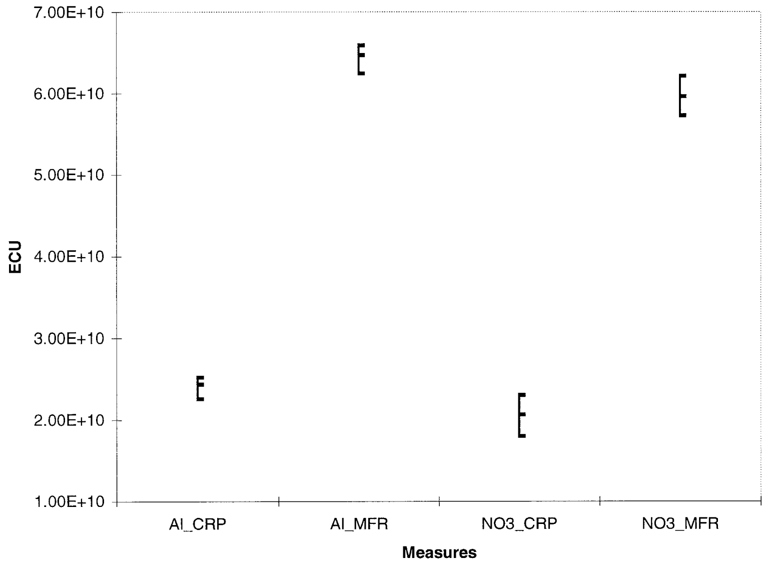

Fig. 3. Objective function intervals for the Netherlands, indicated by 10%, 50% and 90% percentiles for nitrate (NO_3) and aluminium (Al) with respect to CRP and MFR reduction scenarios.

tives. It should also be noted that the uncertainties displayed in Fig. 3 only reflects the uncertainty of hydrological modelling, and that other factors, e.g., the economical valuing of the groundwater resource and the costs for emission reductions, represent additional sources of uncertainty not included in this study.

Acknowledgements

This work was funded by the European Union within the Environment and Climate Programme (Project ENV4-CT95-0070, UNCERSDSS). We would like to thank Hans Kros at the Winand-Staring Centre for providing the SMART2 results, Tommy Norberg at the Department of Mathematical Statistics at Chalmers for statistical discussions, and the UNCERSDSS project group for providing useful comments to the decision-analysis.

References

[1] J. Kros, G.J. Reinds, W. De Vries, J.B. Latour, M. Mollen, Modelling of soil acidity and nitrogen availability in natural ecosystems in response to changes in acid deposition and hydrology, Report 95, DLO Winand Staring Centre, Wageningen, 1995.

[2] A. Vatn, L.R. Bakken, M. Azzaroli Bleken, P. Botterweg, H. Lundeby, E. Romstad, P.K. Rørstad, A. Vold, Policies for reduced nutrient losses and erosion from Norwegian agriculture—integrating economics and ecology, Norwegian J. Agric. Sci. Suppl. 23 (1996) 1.

[3] U. von Brömssen, U. Bertills, Effekter på grundvatten, in: Y.-W. Brodin (Ed.), Effekter Av Svavel—Och Kvävebelastning På Skogsmark, Yt—Och Grundvatten, Swedish Environmental Protection Agency Report 3762, 1990, pp. 105–126.

[4] J. Alcamo, R. Shaw, L. Hordijk (Eds.), The RAINS Model of Acidification, Science and Strategies in Europe, Kluwer Academic Publishers, Dordrecht, the Netherlands, 1990.
[5] M. Amann, I. Bertok, J. Cofala, F. Gyarfas, C. Heyes, Z. Klimont, M. Makowski, S. Shibayev, W. Schöpp, Cost-effective Control of Acidification and Ground-Level Ozone, Third Interim Report to the European Commission, DG-XI, IIASA, 1997.
[6] R.A. Freeze, J. Massmann, J.L. Smith, T. Sperling, B.R. James, Hydrogeological decision analysis: 1. A framework, Ground Water 28 (5) (1990) 738–766.
[7] Committee on Valuing Ground Water, National Research Council (U.S.), Valuing groundwater: economic concepts and approaches, National Academy Press, Washington, DC, 1997.
[8] B.C. Field, Environmental Economics. An Introduction, McGraw-Hill International Editions, 1994.
[9] B.R. James, D.D. Huff, J.R. Trabalka, R.H. Ketelle, C.T. Rightmire, Allocation of environmental remediation funds using economic risk–cost–benefit analysis: a case study, Ground Water Monitoring and Remediation, Fall 1996, pp. 95–105.