Environmental benefits and social cost – an example of combining Bayesian networks and economic models for analysing pesticide management instruments

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Abstract There is a need for introducing interdisciplinary tools and approaches in water management for participatory integrated assessment of water protection costs and environmental benefits for different management scenarios. This is required for the Water Framework Directive. Bayesian belief networks (BN) are one example of a possible tool for participatory integrated assessment. BNs allow knowledge and data from economic, social and hydrological domains to be integrated in a transparent, coherent and equitable way. The paper reports on the construction of a BN to assess impacts of pesticide management actions on agricultural economics and groundwater and drinking water quality, with the overall aim of exploring complexity and uncertainties.

With the participatory BN learning process data, expert knowledge and modelling results were combined into a cost-efficiency and cost-benefit analysis, BN being based on a focused dialogue between participating domain experts and end-users of the research. The instruments analysed by the constructed BN were taxes on pesticides and herbicides, and pesticide-free buffer zones around field edges and water abstraction wells.
Keywords Bayesian belief networks; cost-benefit; groundwater contamination; integrated water resource management; pesticides

Introduction

The Water Framework Directive (WFD), adopted in 2000 (2000/60/EC), is one of the first European directives in the domain of water, where economics is an integrated part of the decision-making processes. Since uncertainty is inherent in economic analyses, particularly those associated with environmental benefits for which there are no existing markets, the issue for the analyst is not how to avoid uncertainty, but how to account for it and present useful conclusions to policy-makers. The Directive (article 11 and annex III) calls for approaches, tools (e.g. cost-effectiveness analysis) and economic instruments (e.g. water pricing methods) evaluating the costs and effectiveness of the proposed programme of measures to reach the environmental objectives.

Previous research projects within this field comprise more disciplinary approaches and tools in water management (Spulber and Sabbaghi 1998; Global Water Partnership 2000; Grimeaud 2001; Swyngedouw *et al.* 2002; Wurzel 2002; Jacobsen *et al.* 2004*a*; Moss 2004; Jordan *et al.* 2004). Hansen (2004) and Young (2005) provided an assessment and modelling

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of cost-effective water protection. Examples of an assessment of the benefits of water protection and the benefits of a supply of clean water has been provided by Desvousges *et al.* (1992), Russel *et al.* (2001), Hanley *et al.* (2003) and Hasler *et al.* (2005). Economic costs of land use changes aimed at protecting groundwater resources from agricultural pollution has been analysed by Schou (2003). One Danish example of cost-benefit assessment for nutrients has recently been provided by Hasler *et al.* (2005). But cross-disciplinary approaches towards integrated water resource management with emphasis on social and technological efficient water use are scarce.

Compared to the methods covered by the above references for cost-benefit and valuation, BNs allow explicit incorporation of additional non-market indicators to provide links to risks for water quality, social variables, etc., and also allow stakeholders to take part in this participatory integrated assessment. The extent to which present water management actions secure protection of the future water resources and supply of water is essential and raises complicated challenges, e.g. what are the social gains and costs of alternative water actions? A possible tool supporting some of these challenging new requirements raised by the current implementation of the Water Framework Directive could be the use of BNs on such issues for participatory integrated assessment.

Bayesian belief networks (BNs) have recently been tested in European river catchments when used in participatory processes (Henriksen *et al.* 2007*a*,*b*; Bromley 2005). BNs can be used as a tool for exploring complexity and uncertainty, and the graphical nature facilitates formal discussion of the structure of the proposed model which supports participation and a more focused cooperation of domain experts. Also, the ability of a BN to describe the uncertain relationships amongst variables is ideal. The influence of norms and values can also be incorporated in the analysis (Henriksen *et al.* 2007*b*), which is required if institutional diversity (Ostrom 2005) and institutional economy has to be fully incorporated (North 1990, 2005) in order to provide more realistic modelling of human behaviour and institutional change.

This paper reports on an interdisciplinary case with a high degree of uncertain relationships amongst variables. Construction of a BN to assess impacts of pesticide management instruments on agricultural application of pesticides, and influence on groundwater and drinking water quality by that use is described. The methodology is to analyse different scenarios by constructing a BN which combines results of macro-economic models and groundwater monitoring data.

The background for this study is the concern about the impact of modern agriculture on the environment in the past few decades which has resulted in legislation concerning the use of pesticides on Danish farms and in the wider society. Frandsen and Jacobsen (1999) showed that the cost to society of a complete or partial ban on pesticides would amount to 0.82 and 0.35% of real Gross Domestic Product (GDP), respectively (Bichel Committee 1999). However, relatively little research effort has been devoted to the cost-effectiveness of different policy instruments to regulate the use of pesticides and to evaluate effects on economics and groundwater, where the impacts of the strategies on economy and groundwater quality are assessed from a cost-benefit perspective. The objective of this study is thus to assess the impact of three different pesticide management instruments on ground and drinking water quality. The management instruments analysed are (1) taxes on pesticides, (2a) taxes on herbicides, (2b) differential taxes on herbicides and (3) pesticide-free buffer zones around field edges and water boreholes.

In the present paper we analyse which of the above four instruments is the most cost-effective by assuming that we invest the same amount of money (total cost) for the measure (0.9 billion DKR).

Methodology

The problem that we want to solve is the one of integrating hydrology and economy. We want to extend the economic analysis traditionally incorporating macroscale economic models, sector models and calculation of changes in land use and pesticide treatment index with variables from hydrology describing leaching of pesticides from the root zone, pollution of pesticides in groundwater and threats to drinking water abstraction from shallow and deep groundwater aquifers. Predicting the consequences for different measures taken at the constitutional choice level (Ostrom 2005), e.g. increased taxes on pesticides and/or areas of pesticide-free protection zones around wells and/or farming fields thereby can draw on economic models and farm statistics combined with monitoring data and expert evaluation for the groundwater system.

Bayesian networks (BNs)

A Bayesian network (BN), also called a belief network, is a decision support system based on probability theory which implements Bayes' rule of probability (Pearl 1988; Jensen 2002). This rule shows mathematically how existing beliefs can be modified with the input of new evidence. BNs organise the body of knowledge in any given area by mapping out cause-and-effect relationships among key variables and encoding them with numbers that represent the extent to which one variable is likely to affect another (see Figure 1).

BNs have been applied for many years in a variety of fields, including engineering, science and medicine (Jensen 2002; Bromley 2005, Henriksen *et al.* 2007*a*,*b*) and have gained the reputation of being powerful techniques for modelling complex problems involving uncertain knowledge and uncertain impacts of causes. The graphical nature of BNs facilitate formal discussion of the structure of the proposed model and the ability of a BN to describe the uncertain relationships amongst variables which is ideal for dialogue processes with participation of experts from different domains (e.g. economics, biologists, hydrologists, geologists, etc.) and for dialogue with stakeholders.

The gained reputation of BN being powerful for modelling complex problems involving uncertain knowledge and uncertain impacts of causes is due to two characteristics

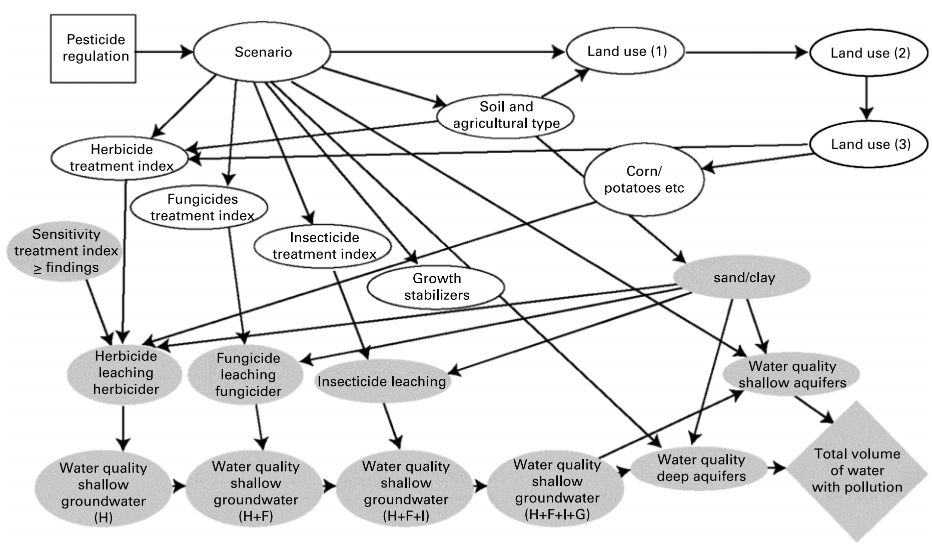

Figure 1 Constructed Bayesian network (BN) for evaluating cause-and-effect relationships between land use, pesticide treatment index and leaching of pesticides to groundwater. Variable shown with the white and grey shading refers to the agricultural and hydrological part, respectively

(Henriksen *et al.* 2007*b*). Firstly, BNs as a modelling language have a straightforward semantics, namely that of cause-and-effect. A domain expert without particular skills in probability theory can easily use them. In the present case study three domain experts from the Geological Survey of Denmark and Greenland used them in dialogue with economic experts from the Danish Economic Council. The transparent semantics supports communication between experts and a BN model can form a solid basis for discussion and shared understanding. Secondly, BNs over a universe U is a compact representation of the joint probability distribution $P(\mathrm{U})$. That is, a BN model holds all sufficient information for reasoning over $P(\mathrm{U})$. Furthermore, computer systems are available which efficiently perform the required probability calculations. In the present case we selected Hugin (www.hugin.com), which offers a graphical user interface and decision engine.

BN construction

The BN construction process followed a seven-step procedure suggested by Bromley (2005).

Define the context. First physical and socio-economic boundaries, area of interest, alternative scenarios and indicators were defined. The developed BN was to provide estimates on a national Danish scale. We decided to include existing basis conditions (Current 2002) but also to incorporate recent decisions made on the EU level (extending the EU with countries in Eastern Europe which would influence farm economics).

Identify actions (measures) and indicators (results). Important indicators in the present case comprise pesticide treatment indexes (describing the usage of pesticides) and probabilities of pesticides leaching from the root zone and associated occurrences in shallow and deep groundwater and with that of drinking water. The treatment frequency index (TFI) measures the intensity of pesticide treatment on areas potentially treated with pesticides. TFI is defined as the 'number of standard doses' divided by the 'area relevant for pesticide treatment' (TFI = number of standard doses/area relevant for pesticide treatment), as the TFI is affected by both the use of pesticides and the area potentially relevant for pesticide treatment. The standard doses depend on pesticide properties, being lowest for those pesticides with the highest risk of leaching.

Build pilot network. Important variables were identified, and directed edges (links between variables) selected and connected, e.g. links connecting land use, soil, treatment indexes, water quality, etc. Basically, two disciplines from either agricultural or hydrological domain are combined (Figure 1).

The variables related to *the agricultural domain* consist of land use, soil and agricultural type, corn/potatoes, and treatment indexes for herbicides, fungicides, insecticides and growth stabilizers (Figure 1). For the variable growth stabilizers no laboratory test method exists for analysing this content in water; therefore the influence of this product could not be linked to groundwater. In order to couple land use data deriving from different sources, e.g. the dataset for the approximately 2000 farms determined by Esmeralda, the aggregated farm-level results to national type of aggregation in Esmeralda and the limited number of different land uses applied for the coupling to groundwater leaching and treatment index data, a sequence of land use variables was introduced in the BN (see Figure 1).

The variables related to *the hydrological domain* consist of leaching (herbicides, insecticides and fungicides), water quality in shallow groundwater, and water quality in shallow and deep aquifers (Figure 1). The variable 'sensitivity treatment index → findings' captures three alternative conceptual understandings of the relationship between treatment index and leaching (Figure 1).

Collect data. Data on socio-economical behaviour, agricultural management and groundwater quality were collected from different sources as described in the data acquisition section. Primarily, a simple BN was prepared for discussion in the expert group.

After feedback BNs were adjusted and refined with additional variables and links, e.g. a 'sensitivity variable' describing three alternative assumptions about the relationship between pesticide load (treatment index) and leaching of pesticides.

Define states. For water quality, for example, there are three states: $<0.01\ \mu g/l$, $0.01-0.1\ \mu g/l$ and $>0.1\ \mu g/l$, which were implemented in BNs for all variables (see Figure 2). For the herbicide treatment index variable a large number of states were incorporated in order to provide as many details as possible.

Construct conditional probability tables (CPTs). CPTs for the hydrological variables part (variables shown with grey shading – Figure 1) were manually constructed based on the spreadsheet analysis of available monitoring data. Based on outputs of economic models (AAGE and ESMERALDA) the CPTs for the farm-economy and treatment index (variables shown with white shading – Figure 1) were calculated as a sub-BN. In order to include ontological variability, data from 2000 farms were used to define 10% fractal values for each distribution. Based on these fractals and mean values a distribution was defined and tested for existing situations. Subsequently BNs were checked for internal consistency (structure, values placed in CPTs, e.g. logical behaviour, and checks in diagnostic mode). A check for logical behaviour can be done by noting the impact of each implementation and controlling nodes to represent the current situation. A check of the network in diagnostic mode is a test where a child node is fixed, which then propagates back to describe the state of the parents needed to generate that condition (Table 1).

Collect feedback from stakeholders. In this project the BN was used as a tool for integration, e.g. for scientific focused communication between a group of experts from different fields of expertise. Thus, in step 7, opinions on the final network were collected and negotiated, but the BN CPTs were not adjusted. The stakeholders here were the experts from the economic disciplines (Danish Economic Council and Danish Research Institute on Food Economics, FOI) and a water research leader from GEUS, responsible for communicating the overall results to the Environmental Ministry before publishing results. Thus the opinion from this quality assurance responsible was discussed and used as input for a more thorough and critical discussion of results, also in relation to wider institutional perspectives and ongoing research within the pesticide research area.

Note that bridging the agricultural and hydrological domain for herbicides involves the additional parent variables 'sensitivity treatment index → findings', 'corn/potatoes, etc.' and 'sand/clay'. This means that the largest complexity has been incorporated for this branch of pesticides including soil, rotation and sensitivity into the relationship between treatment index and leaching. For fungicides and insecticides only soil is incorporated as a control variable beside treatment index impacting the child variable leaching. The design of the BN and the included links thus reflect the selection of the most important relationships included, in order to make the final BNs as simple as possible.

Data acquisition

Pesticide and agricultural economics

Required input data on pesticide and agricultural economics (e.g. pesticide treatment indexes) were based upon a combined set of economical models. The combined modelling system further described in Jacobsen *et al.* (2004*b*) comprised of: a general equilibrium model for assessing the macro-economic effects and overall impacts of different pesticide reduction strategies on the industry structure (AAGE, Agricultural Applied General Equilibrium model) and an economic agricultural sector model assessing the impact of different pesticide reduction strategies on agricultural land use, pesticide use and agricultural incomes (ESMERALDA).

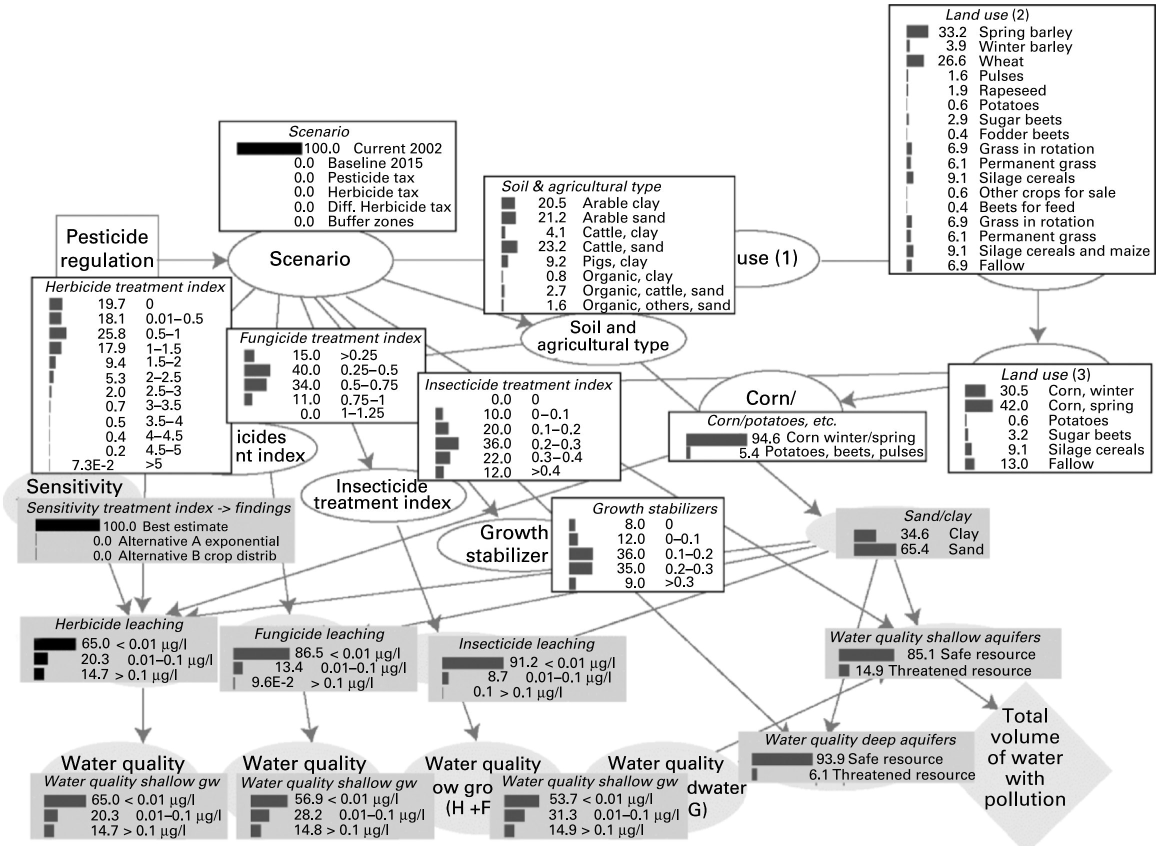

Figure 2 Bayesian belief network (BN) for basis 2002 'current state' assuming a linear relationship between treatment index and pesticide leaching (fixed variables shown with black bars) used for 'calibration'

Table 1 Conditional probability table for the variable *fungicide leaching*. For each of the two parent variables, soil and fungicide treatment index a conditional probability distributions over the three states: <0.01 μg/l; 0.01–0.1 μg/l and >0.1 μg/l have been specified

Sand/clay	**Clay**					**Sand**				
Fungicide treatment index	**>0.25**	**0.25–0.5**	**0.5–0.75**	**0.75–1**	**1–1.25**	**>0.25**	**0.25–0.5**	**0.5–0.75**	**0.75–1**	**1–1.25**
<0.01 μg/l	1	0.85	0.7	0.57	0.496	1	0.93	0.83	0.8	0.7
0.01–0.1 μg/l	0	0.149	0.298	0.427	0.5	0	0.0695	0.129	0.1985	0.298
>0.1 μg/l	0	0.001	0.002	0.003	0.004	0	0.005	0.001	0.0015	0.002

The two models were run sequentially. First, AAGE determined macroeconomic results including prices and aggregate levels of agricultural production at a national level. In the second stage, the agricultural sector model (ESMERALDA) was run, conditional on prices and agricultural output levels from the first stage, determining land allocation, pesticide intensity and agricultural income, distributed on different farm types (Jacobsen *et al.* 2004*b*).

The AAGE model covers the whole Danish economy. AAGE determines supplies and demands for commodities through optimising behaviour of five types of agents in competitive markets. Interdependencies between the individual industries, interaction between industries and consumers and between domestic and foreign agents are also accounted for in the AAGE model (Jacobsen *et al.* 2004*b*).

ESMERALDA (Jensen *et al.* 2001) describes production, input demands, land allocation, livestock density and environmentally relevant variables on representative Danish farms, and subsequently in the Danish agricultural sector at relevant levels of aggregation. A basic assumption underlying the model's behaviour description is that farmers exhibit economic optimisation behaviour, which means that farmers allocate production to the lines of production with the highest return.

The coupling of value assessment of benefits and results of AAGE, ESMERALDA and BNs for an integrated assessment are shown in Figure 3. Value assessment of biodiversity benefits and clean drinking water benefits (left side of Figure 3) were based on a quantitative questionnaire approach (Danish Economic Council 2004) and a literature review, respectively. The economic models were used in an iterative approach. The general equilibrium model AAGE, conditional on agricultural behavioural parameters from ESMERALDA, was used for simulating changes in economic prosperity whereas ESMERALDA was used, conditional on prices and agricultural output from AAGE, to simulate changes in land use and relative changes in pesticide use per hectare of the

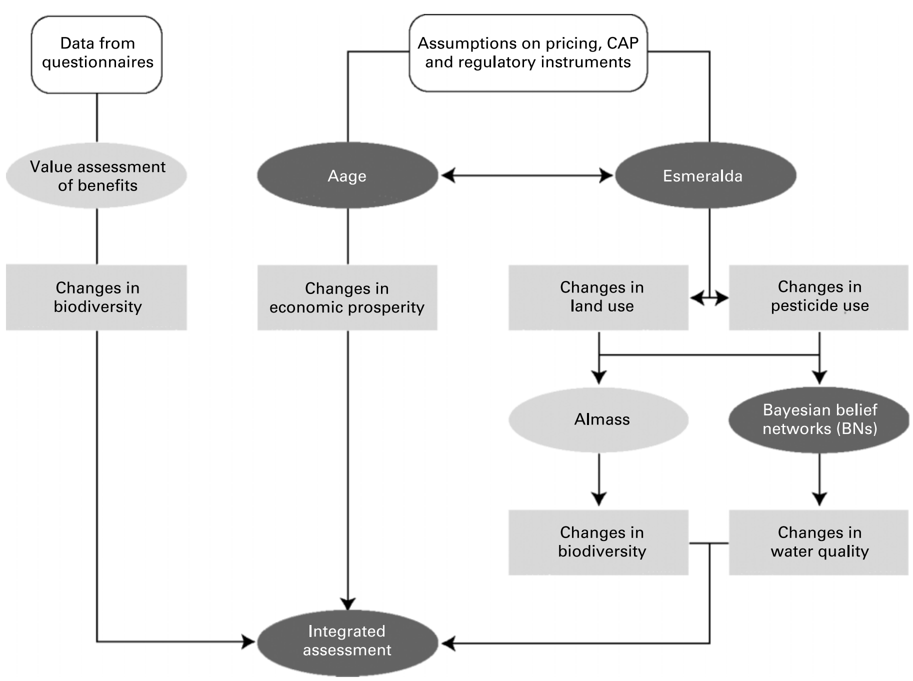

Figure 3 Dataflow diagram describing the coupling of value assessment of benefits and results of AAGE, ESMERALDA and BNs for an integrated assessment. Source: Simplified from Danish Economic Council (2004, p 240)

respective crops. A biological model, Almass, was used with inputs from ESMERALDA on land use and pesticide use change to simulate changes in biodiversity using skylarks as indicator (Topping 2005). The Bayesian belief network constructed was used for an assessment and modelling of changes in water quality using the same input simulated by ESMERALDA. Finally, based on the value assessment and the simulated changes in biodiversity, economic prosperity and water quality an integrated assessment of benefits and costs was provided.

The impact of pesticide usage reduction on skylarks was in Danish Economic Council (2004) evaluated using a landscape-scale individual-based model (Topping 2005). The biodiversity results indicated that the general reductions in pesticides would have only minor positive impact on skylarks. The greatest benefit for biodiversity (skylarks) was found for the scenario with the use of unsprayed field margins. For simplicity reasons this paper focus only on the water quality aspects as an example (and not the full complex story).

Groundwater quality

Required input data for risk of groundwater contamination for different land uses and rotations were assessed from the Danish Pesticide Leaching Assessment Programme PLAP (Kjær *et al.* 2004) and the groundwater monitoring program GRUMO (Stockmarr 2004), as well as water quality data from a survey of small waterworks (Brüsch *et al.* 2004).

In order to couple the pesticide treatment index with the groundwater system, the PLAP data describing the observed root zone leaching (below the root zone in 1 m depth) was utilised as input for shallow groundwater quality, Table 4. The leaching risk summarised in Table 4 thus derives from the PLAP programme measuring the leaching of the application of 74 pesticides occurring over a five year period on six different experimental fields (Kjær *et al.* 2001). In addition to the treatment index, soil type and crop may also influence the leaching risk. Difference in time of application as well as properties of the applied pesticides makes some crops more prone to pesticide leaching than others. Moreover those pesticides prone to macropore transport may be more leachable on loamy soil than on sandy soils. In order to account for these factors two extra CPTs accounting for soil type (sand/clay CPT) and crop type (corn/potatoes CPT) were included in the BN. The differentiation within crop types was, however, only done for herbicides as the limited data available for fungicide and insecticides did not allow for further differentiation.

The existing PLAP data represents leaching risk associated with a treatment index in the interval 1.5–2 and CPTs for other treatment index 'states' were estimated based on different assumptions. A linear relationship between pesticide dosage (here expressed as treatment index) and leaching is valid for those pesticides being sorped linearly, but not for those pesticides sorping nonlinearly that require a nonlinear relationship. Moreover the relationship was found to be crop-dependent (Henriksen *et al.* 2004). In order to account for these uncertainties three different relationships were applied in the scenarios, these being:

- linear relationship between treatment index and leaching (best estimate),
- a linear but crop-specific relationship between treatment index and leaching (type A), and
- a non-linear crop-specific relationship between treatment index and leaching (type B).

The relationship between the contamination risk of deep and shallow groundwater was assessed from an in-depth analysis of groundwater monitoring data and accounted for in the 'Water quality aquifers' CPT. As compared to a shallow aquifer the contamination risks of the deep sandy and loamy aquifers were assumed to be 50% and 34% lower, respectively (Henriksen *et al.* 2004).

The estimates of the probable polluted amounts of groundwater and drinking water is calculated based on the probability of findings of pesticides having the state: >0.1 μg/l, e.g.

for the Current 2002 scenario, and assuming a linear relationship (Figure 2) this probability is 14.9% for the variable 'water quality for shallow aquifers' and 6.1% for water quality for deep aquifers. The polluted amount of water is then based on the estimates of groundwater recharge to shallow groundwater (= 9 million m^3/yr) and the estimates of available drinking water resources (= 1 billion m3/yr) based on the National Water Resource Model (Henriksen *et al.* 2003; Henriksen and Sonnenborg 2003).

Pesticide scenarios

First a basis 2002 scenario (current conditions for the year 2002) and a baseline scenario (future conditions for the year 2015) were constructed to introduce all ongoing policy developments and known shocks to the economy, taking departure in current trends in economic growth, productivity, the EU enlargement from May 2004, etc. (Jacobsen *et al.* 2004*b*). With regards to the agricultural sector, three policy initiatives, which had been decided but not yet fully implemented, were accounted for in the baseline scenario with the focus on the macro-economic forecast of the Danish economy up to 2015. The assumed changes in productivity led to an increase in effective labour units and consequently a total increase in GDP, leading to an increase in the capital stock. The baseline scenario implies some changes in the agricultural sector, due to changes in foreign and domestic demands as well as changes in the supply conditions caused by, e.g., environmental regulations and reforms of the Common Agricultural Policy. The cultivated area is reduced by 180 000 ha in the baseline projection, due to increased demand for land for other purposes (e.g. urban growth and afforestation).

The baseline scenario represent the likely projection of the coming 10 years' development of Danish economy and agriculture. The assumptions of the baseline scenario are:

- Public consumption shock with actual development from 1995 to 2003, thereafter an annual increase of 1% per annum is assumed.
- Prices in foreign trade from GTAP model simulations; this also introduces effects of the enlargement of the European Union.
- Labour productivity, annual growth, assumed between 2.5–6% in agriculture, 2.2% in manufacturing and 2.1% in service.
- The 2003 reform of the CAP including intervention price cuts for butter and skimmed milk powder, compensatory payments to the dairy quota, increase in the dairy quota, full decoupling of hectare premium, partly decoupling of animal premium and modulation of direct support.
- Action Plan for the Aquatic Environment III with phosphorus taxes, buffer zones with compensation payments to land, late crops requirements tightened and manure utilisation requirements tightened.
- Pesticide taxes in the 1995–2003 period.

The four instruments analysed for pesticide reduction introduced the following assumptions:

- Scenario 1 (pesticide taxes): A general levy on all pesticides with the aim of reducing the total use of pesticides by 25%.
- Scenario 2a (herbicide taxes): A levy on herbicides leading to the same welfare loss as scenario 1 (induces a reduction in the use of herbicides by 40%).
- Scenario 2b (differentiated herbicide taxes): A soil-type differentiated levy on herbicides, obtaining 80% of the herbicide-use reduction on clayey soil and 20% on sandy soil.

- Scenario 3 (buffer zones): Pesticide-free buffer zones around all fields and abstraction wells leading to the same welfare loss as scenario 1 (the resulting buffer zone area amounts to 14% of the total arable area).

The effects of different instruments intended to reduce the use of pesticides are calculated by comparing output from the three scenarios with that of the baseline scenario for the year 2015. The calculations were scaled to levels so that the social costs of the regulation were similar in all scenarios, slightly below DKR 1 billion per year (which corresponds to about 1200 000 Euro).

Cost–benefit analysis

The benefit estimate by Hasler (2005) was not available when Danish Economic Council (2004) made the cost-benefit assessment. Instead a literature survey was carried out in order to compare the social costs with the economic value of the environmental benefits for groundwater and drinking water, which is necessary to obtain monetary estimates of the value of the environmental improvements.

A number of international studies were used by Danish Economic Council (DØR) to estimate a rough valuation of safe drinking water, with a resulting estimate for how much consumers will pay in additional payments to their water bill for 'safe drinking water supply' of 900 DKR/year/household. Finally, this estimate was used to assess the overall Cost Benefit Analyses (CBA).

Results

Treatment index

A general tax on all pesticides is the most cost-effective means to reduce the total use of pesticides, the estimated reduction being 26% (Table 2). On the other hand, if the objective is to mainly reduce the use of herbicides, a general tax on herbicides is the most cost-effective measure estimating an obtained reduction of 33% (Table 3). As the most known groundwater contaminants are related to herbicides (Stockmarr 2004), taxes focusing on herbicides should be considered further (Table 4).

The relative cost-effectiveness depends on the aim of the regulation. The overall cost-effectiveness of buffer zones is thus relatively low, if the aim is to reduce the total use of pesticides. However, if the aim is to improve the conditions for wildlife, etc., pesticide-free buffer zones may be a more cost-effective regulation as such buffer zones address habitats close to the field borders (Topping 2005). However, influence on biodiversity is not included in this paper, so in the following the scope is narrowed to the influence on water quality.

Triangulation with groundwater monitoring data

The BN estimates (Table 5) describing the current situation (basic 2002) were generally consistent with observed data indicating a reasonable description of groundwater

Table 2 Simulated pesticide treatment indexes for current 2002, Baseline 2015 and the four scenarios. Source: Jacobsen *et al.* (2004*a*,*b*)

	Current 2002	Baseline 2015	Pesticide tax (1)	Herbicide tax (2a)	Diff. Herbicide taxes (2b)	Buffer zones (3)
Herbicides	0.96	0.93	0.64	0.54	0.76	0.75
Fungicides	0.45	0.59	0.47	0.54	0.56	0.48
Insecticides	0.26	0.24	0.16	0.25	0.24	0.21
Growth regulators	0.20	0.18	0.16	0.16	0.17	0.14
Total	1.87	1.93	1.42	1.48	1.73	1.57

Table 3 Simulated average herbicide treatment indexes. Source: Jacobsen *et al.* (2004 *a*, *b*)

	Current 2002	Baseline 2015	Pesticide tax (1)	Herbicide tax (2a)	Diff. herbicide taxes (2b)	Buffer zones (3)
Spring barley	0.75	0.75	0.53	0.43	0.62	0.63
Winter barley	1.20	1.27	0.74	0.60	0.95	1.06
Sugar beets	2.14	2.04	1.61	1.49	1.65	1.70
Pulses	2.82	2.41	1.80	1.62	2.13	1.94

Table 4 Crop rotations with pesticide applications with findings of one or more pesticides or degradation products above the drinking limit value (MAC $> 0.01\ \mu$g/l). The numbers in brackets are findings in percent of the total number of rotations (Source: GEUS)

	Clay			Sand		
	Fungicide	Herbicide	Insecticide	Fungicide	Herbicide	Insecticide
Number of rotations (tests) in total	7	16	11	5	9	2
Number of rotations with:						
-no leaching of pesticides 1	4 (57%)	3 (19%)	6 (55%)	4 (80%)	5 (56%)	2 (100%)
-leaching below limit value 2	3 (43%)	9 (56%)	5 (45%)	1 (20%)	2 (22%)	0 (0%)
-leaching above limit value 3	0 (0%)	4 (25%)	0 (0%)	0 (0%)	2 (22%)	0 (0%)

contamination risks. Contaminated ground water (containing more than 0.1 μg/l pesticides) was found in 10–20% and 5% of the monitoring wells situated in the shallow and deep aquifers (<40 m below ground surface), respectively (Figure 4). This was in good agreement with the estimation from the BN, being 14% and 6% for shallow and deep aquifers, respectively.

We do not regard this comparison as a validation of the approach and method because the lag time of the groundwater system makes it extremely complicated to evaluate monitoring data from different domains, e.g. leaching from rootzone having an age of 1–2 yr, water in shallow aquifers being something between 1 and 4 decades, and groundwater from deep aquifers which can have a lag time from 4 decades to 100–1000 decades. Therefore we have selected the term 'triangulation' to emphasise the more soft evaluation of the BN we have attempted.

Overall results of instruments and pesticides scenarios

The overall result of the BN analysis is shown in Table 5. The best estimate (linear relationships) in fact shows the 'worst total estimate of contaminated groundwater', compared to alternatives A and B assuming a crop- and soil-dependent relationship between treatment index and leaching.

A tax on herbicides (i.e. weed killers) is the most effective instrument with respect to the protection of the total groundwater resource (Table 5 and Figure 5 – upper), including groundwater reserves not currently utilised for drinking water. When the environmental effects measured in physical units are combined with the estimated valuation of the reduced contamination of the drinking water (Figure 5 – lower), it appears that the value of the environmental gains exceeds the social costs, i.e. that there is a net benefit associated with pesticide-free buffer zones (Table 6).

The most effective scenario for reducing total pesticide loads is introducing taxes on pesticides. However, taxes on pesticides were not the most cost-effective solution for

Table 5 Overall result of Bayesian network. Probability of groundwater pollution (in %) for current 2002 situation

Instrument	Probability of herbicides in shallow groundwater (>0.1 g/l)	Probability of pesticides (herbicides + fungicides + insecticides) in shallow groundwater (>0.1 g/l)	Probability of pesticides in abstracted raw water from deep groundwater aquifer (>0.1 g/l)	Probability of pesticides in abstracted raw water from shallow groundwater (>0.1 g/l)
Volume (%)			75% of total abstraction	25% of total abstraction
Linear relationship (treatment index - > pesticide leaching)				
Current 2002	14.8	14.9	6.1	14.9
Baseline 2015	14.4	14.5	5.9	14.5
Pesticide tax (1)	11.5	11.5	4.8	11.5
Herbicide tax (2a)	10.2	10.2	4.2	10.2
Diff. herbicide tax (2b)	12.7	12.7	5.2	12.7
Buffer zones (3)	13.2	13.2	5.0	0.9
Alternative A				
S-curve relationship				
Current 2002	12.1	12.3	5.2	12.3
Baseline 2015	12.0	12.2	5.1	12.2
Pesticide tax (1)	8.1	8.3	3.5	8.3
Herbicide tax (2a)	7.2	7.3	3.1	7.3
Diff. herbicide tax (2b)	9.7	9.8	4.1	9.8
Buffer zones (3)	10.4	10.5	4.0	0.8
Alternative B				
Crop specific relationship				
Current 2002	10.4	10.5	4.8	10.5
Baseline 2015	9.0	9.1	4.3	9.1
Pesticide tax (1)	7.3	7.5	3.5	7.5
Herbicide tax (2a)	6.1	6.2	2.9	6.2
Diff. herbicide tax (2b)	7.7	7.8	3.7	7.8
Buffer zones (3)	8.0	8.1	3.6	0.7

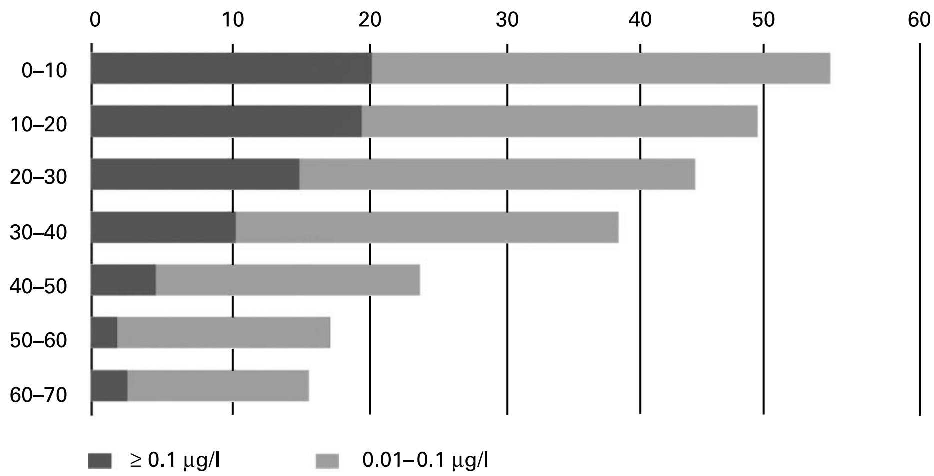

Figure 4 Results of groundwater monitoring. Depth intervals: 0−10, 10−20, . . . ,70−80 m below surface. Bars show percent of findings of pesticides in groundwater monitoring programme above drinking water limit value (>0.1 µg/l − shaded), and percent findings above detection limit but below limit value (light shaded)

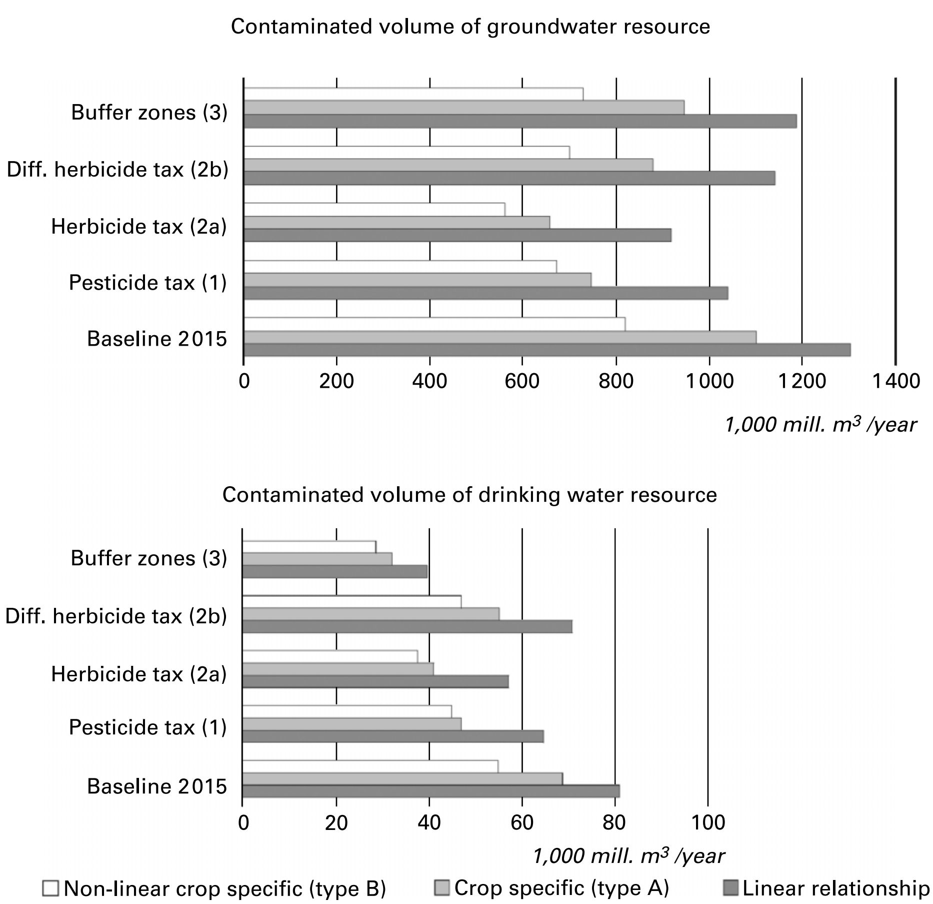

Figure 5 Contaminated volume of groundwater (upper) and drinking water (lower) or different scenarios and assumption about treatment index → leaching. The available resources of groundwater and drinking water is 9000 mm^3 and 1000 mm^3, respectively (Henriksen and Sonnenborg 2003)

groundwater. Instead herbicide taxes were more cost-efficient. For the reduction of pesticides in drinking water, buffer zones were the most cost-efficient tool.

Cost–benefit analysis

When we focus first on the environmental effects of the alternative pesticide regulation methods, without including the monetary values of these effects, it appears from the analyses that pesticide-free buffer zones would have the greatest positive impact on biodiversity as well as on pesticide residuals in drinking water (Danish Economic Council 2004).

As evaluated by DØR there are methodological problems associated with the applied hypothetical valuation method. The overall monetary benefits of the different types of pesticide regulation should, for that reason, be interpreted cautiously. However, since the pesticide-free buffer zones yield a net gain even if the monetary value of the environmental benefits is considerably lower than that found in the valuation studies, and since it is likely that there are additional environmental benefits and health benefits not taken into account in the study, the overall result of the economic reasons for implementing protection zones seems to be quite reasonable and credible. Recent studies (Hasler *et al.* 2005) have shown a valuation of the same level or the double based on quantitative surveys which support this evaluation by DØR.

A tax on herbicides (i.e. weed killers) is the most effective instrument with respect to the protection of the total groundwater resource, including groundwater reserves not currently utilised for drinking water. However, the tax on herbicides did not have as great a positive effect on biodiversity as was found with pesticide-free buffer zones (Danish Economic Council 2004).

It should be noted that there are problems concerning implementation and control associated with pesticide-free buffer zones that do not arise when using a pesticide tax, as it is costly to oversee whether farmers act in conformity with the pesticide-free buffer zone regulations. In comparison, it is relatively cheap to increase the level of pesticide taxes, though increased tax levels may lead to an increase in cross-border trade (Danish Economic Council 2004).

In Denmark, groundwater is the main source for the public water supply and for watering crops (99% of the water supply is based on groundwater). The extraction of groundwater in parts of Zealand exceeds the sustainable level (Henriksen and Sonnenborg 2003), causing environmental consequences, e.g. water levels in lakes and rivers are lowered, and the quality of the future groundwater has a high risk of further decline. The current level of exploitation is not easily reduced and sustainability may therefore not be addressed by economic regulation but since the groundwater at Zealand is at the alert risk, safe groundwater protection and monitoring of the decline in groundwater quality is a necessary way forward.

The pesticide-free buffer zones yield a net gain even if the monetary value of the environmental benefits is considerably lower than that found in the valuation studies (900 DKR per family per year).

Table 6 Cost-benefit assessment (Dors 2004). Modified only to include drinking water

	Pesticide tax (Billion DKK per year)	Buffer zones (Billion DKK per year)
Social cost	−0.9	−0.9
Environmental benefits drinking water	0.5	1.3
Total	−0.4	0.4

Discussion of results

Reducing uncertainties in a complex world

A general characteristic of human history has been the systematic reduction in the perceived uncertainty associated with the physical environment and therefore a reduction in those sources of uncertainty to be explained by beliefs embodied in witchcraft, magic and religion (North 2005). A Bayesian belief network (BN) is about exploring the domain of scientific knowledge, perceiving and sharing mental models at, and sometimes beyond, the boundaries of current knowledge disciplines about the physical and human environment. The BN exercise hereby is an attempt to bridge different research disciplines, e.g. in our case 'farm economy and social costs' with 'environmental benefits of clean groundwater and drinking water'.

Humans have altered the environment in order to make it more predictable (North 2005), with different degrees of reducing uncertainty:

- Uncertainty that can be reduced by increasing information given the existing stock of knowledge.
- Uncertainty that can be reduced by increasing the stock of knowledge within the existing institutional framework.
- Uncertainty that can be reduced by altering the institutional framework.
- Uncertainty in the face of novel situations that entails restructuring beliefs.
- Residual uncertainty that provides the foundation for 'non-rational' beliefs.

The development of more information about the fate of pesticides in the environment and the Danish pesticide ban approach, where certain high risk pesticides are eliminated from the market, is the current way of developing an informed and regulated use of pesticides. Focus on treatment index is the prevailing tool. With the constructed BN the information is extended beyond the treatment index to capture water quality in groundwater and as such provides useful information about the risk of groundwater contamination by including the existing stock of knowledge from the Danish monitoring programme.

However, there is still a lot of room for improving the predictive capability of the constructed BN. For instance, collection of more monitoring data and tests would reduce the variability (ontological uncertainty) and improve the possibility to construct proper conceptual models (epistemic uncertainty) relating leaching risks with treatment index, soil, rotation, etc. For the constructed BN it was only possible to explore these relationships to some degree.

Using BNs as a participatory tool for exploring the complexity and the uncertainties, we discovered variables that we didn't know enough about, e.g. the fate of growth stabilizers in groundwater. This problem could not be further analysed due to a lack of laboratory test methods for analysing the content of growth stabilizers in groundwater.

Another example is climate change (a context uncertainty) which is not included in the BN although it may influence leaching of pesticides and farm economics. In that way we were ignorant about this complex impact on the problem variables. On the other hand, it was possible to include context uncertainty by including the Baseline 2015 scenario assuming the extension of EU with new member states which happened in 2005.

Integration of disciplines

Inter-disciplinary research is necessary for WFD and for Integrated Water Resource Management (IWRM), as shown in this paper, for describing the results of cause-effect relationships between variables dealing with pesticide instrument scenarios, land use, economics, soil, resulting pesticide treatment indexes, pesticide leaching to shallow and deep groundwater and drinking water quality. However, it is important not only to focus on sub-systems, but also to evaluate how the system 'as a whole' interacts, in order to assess how groundwater quality is influenced by buffer zones around wells and fields, taxes on herbicides, etc. Integration means that the impact of management decisions must not be restricted to the water resource itself, but also embraces the wide range of other factors that play a role in the life of a river basin. To make a balanced and fair judgement we need to be able to evaluate the impacts of decisions on an extensive array of factors. An integrated policy requires all these benefits, drawbacks and costs to be considered and evaluated (Bromley 2005), where some will be physical (e.g. groundwater quality), others economic (e.g. agricultural output) or social (e.g. recreational facilities) or of some other type.

The present paper uses a practical method for integration between two sub-systems: 'pesticide and agricultural economics' and 'groundwater quality' based on Bayesian networks. Since the aim of integrative research is to use a range of different worldviews as the basis for a better understanding of human-environment systems where integration means the process of constructing new worldviews, Bayesian networks are especially powerful in order to provide a focused dialogue in order to construct shared, qualitative and quantitative worldviews or collaborative 'conceptual blending' of domain knowledge from which completely new ways of seeing the world and innovative solutions to problems can emerge (Newell *et al.* 2005).

When governing, sustainable resources like groundwater design principles for instruments include a number of concerns (Ostrom 2005), e.g. clearly defined boundaries, proportional equivalence between benefits and costs, collective choice-arrangement, monitoring, graduated sanctions, conflict-resolution mechanisms, recognitions of rights to organize and nestled enterprises. When individuals affected by buffer zones are not authorized to participate in making and modifying their rules, the rules may not be accepted as fair by participants (farmers). This could be a problem not considered properly in the present analysis of especially the buffer zone instrument. Therefore rules, e.g. buffer zones, made on the constitutional choice level and not on the collective choice level, may be less efficient in the long run.

According to Jakeman and Letcher (2003), evaluating several tools for participatory integrated assessment (e.g. system dynamics, Bayesian networks (BNs), metamodels, risk assessment approaches, coupled component models, agent-based models and expert systems), the requirements of these tools are the following key points:

- being problem-focused, using an iterative, adaptive approach that links research to policy;
- possessing an interactive, transparent framework that enhances communication;
- being enriched by stakeholder involvement and dedicated to adoption;
- attempting to recognise missing essential knowledge; and
- connecting complexities between the natural and human environment, recognising spatial dependencies, feedbacks and impediments.

The first four requirements are fully supported by the constructed BN. The ability of BNs to facilitate a focused dialogue between different domain experts, to explicitly represent uncertainties, using the probabilistic approach of relevance for risk assessment and cost-benefit evaluations, and the powerful exploring of complexity, e.g. as demonstrated for the sensitive relationships between pesticide application and leaching, thus qualify BNs as a tool for analysing complex and uncertain decision problems. However, parts of the latter requirement, being the representation of spatial dependences and feedback, is not fully accounted for in the constructed BN. There are techniques which allow such functionality, e.g. which allow coupling to GIS and which allow dynamic BNs and feedback. However, these features were not available in the HUGIN software when used for BN analysis.

Critical feedback to the framing of the case study (deconstruction of constructed BNs)

The process of constructing a BN should help the researchers to understand the nature of the complex human–physical world. A BN should help researchers to explore the complexities and uncertainties of a problem field, not reduce it to a simplistic or easier solution space. It should encourage the researchers to identify all relevant information and to analyse it more in depth. Furthermore, it should be recognized that it is impossible to be certain about the consequences of any environmental management decision, and that the uncertainty has to be explicit in the responses from the BN. Furthermore, without stakeholder consultation, it is unlikely that an environmental management decision can be implemented.

This was also the case in the present study where two members of the leadership group in GEUS gave important feedback to the constructed BN before the reporting of the results. The raised critical comments were related to three main issues: (1) geological uncertainty, (2) point sources and (3) treatment index as an indicator. In a way, for all three issues, the comment was related to a lack of trust in taxes or treatment index as the way to reduce uncertainties further.

Uncertainties related to geological variability are not fully considered. On sandy soils approximately 10–15% of the agricultural area is more vulnerable than the field sites (PLAP sites) investigated in this study (Nyegard 2004). This implies that there is an uncertainty in relation to vulnerability not considered. Nyegard (2004) has shown that, in fact, some sandy soils are significantly more vulnerable compared to others and buffer zones could be delineated using this more complex knowledge giving a more efficient groundwater protection. For clay this fraction is not known (GEUS and DJF 2004).

Moreover the monitoring data describing the leaching risk (PLAP data) represent pesticides used today, not necessarily pesticides permitted for 2015. Any effect of regulations or bans against pesticides leaching to groundwater according to the registration procedure is not included in the analysis. Moreover, point sources most often related to 'bad management practice' or accidental spills are not accounted for in this analysis although they can contribute significantly to observed contamination (Kreuger and Nilsson 2004).

In this scenario, the treatment index was the only indicator for pesticide leaching, while other factors such as pesticide properties or time of application were not accounted for. The same relationship between treatment index and pesticide leaching was applied for all pesticides, although leaching risk, and thus the impact a reduced dose will have on leaching, may differ considerably within the different pesticides.

If we compare the results of the constructed BN based on the assumptions in Danish Economic Council (2004) of a willingness to pay estimated to 900 DKR per family per year with the willingness to pay estimates of Hasler *et al.* (2005) based on Choice Experiment (CE) and Contingent Valuation (CV) methods, Hasler *et al.* estimate a willingness to pay for these two methods to amount to, respectively, 1900 DKR and 700 DKR per year per household. As pointed out by Hasler *et al.* (2005) the used benefit of 900 DKR per family per year can be considered to represent an underestimate compared to the results by Hasler *et al.*'s CE method. The environmental benefits for drinking water for 'pesticide taxes' and 'buffer zones' using the valuation metod (CE) would be approximately double.

Conclusion

Integrated management is the key to the sustainable development of Europe's water resources. This means that decisions need to be made in the light of not only environmental considerations, but also their economic, social and political impact. BNs are a practical approach for allowing a range of different factors to be linked together in order to come up with a cost-benefit assessment based on probabilistic dependencies, and at the same time provide a framework within which the contribution of models, monitoring data and domain expert knowledge can be analysed.

By the graphical presentation of all variables, links and probability distribution for states, BNs provide transparency supporting the overall Cost-Efficiency and CBA results, which is important for further dialogue with policy-makers, end users and stakeholders. BNs are therefore a possible tool for participatory integrated assessment and social learning, integrating different disciplines and bridging the gap between researchers and policy-makers.

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References

Bichel Committee (1999). *Report from the Sub-committee on Production. Economics and Employment*. The Danish Environmental Protection Agency, København. Available at: http://www.mst.dk

Bromley, J. (2005). *MERIT Guidelines*. Available at: http://www.merit-eu.net/guidelines

Brüsch, W., Stockmarr, J., Kelstrup, N., von Platen-Hallermund, F. and Rosenberg, P. (2004). *Pesticidforurenet vand i små vandforsyningsanlæg* (in Danish). GEUS Report 2005/9. GEUS, Copenhagen. Available at: http://www.geus.dk

Danish Economic Council (2004). *Danish Economy, Autumn 2004* (in Danish, with English summary). Available at: http://www.dors.dk

Desvousges, W.H., Naughton, M.C. and Parsons, G.R. (1992). Benefits transfer: conceptual problem in estimating water quality benefits using existing studies. *Wat. Re. Res.*, **28**(3), 675–683.

Frandsen, S.E. and Jacobsen, L.B. (1999). *Analyser af de sektor- og samfundsøkonomiske konsekvenser af en reduktion i forbruget af pesticider i dansk landbrug.* SJFI Report No. 104. Danish Research Institute of Food Economics, København.

GEUS and DJF (2004). *Særligt pesticidfølsomme sandområder: Forudsætninger og metoder for zonering. Koncept for Udpegning af Pesticidfølsomme Arealer, KUPA* (Nyegard, E. (ed.)). Danmark og Grønlands Geologiske Undersøgelse (GEUS) og Danmarks JordbrugsForskning (DJF), Copenhagen. Available at: http://www.geus.dk

Global Water Partnership (GWP) (2000). *Integrated Water Resources Management*. TAC Background Papers, no 4. GWP, Stockholm. Available at: http://www.gwpforum.org/gwp/library/Tacno4.pdf

Grimeaud, D. (2001). Reforming EU water law: towards sustainability. *Europ. Environ. Law Rev*, **10**(2), 41–51.

Hanley, N., Bell, D. and Farizo, B.A. (2003). Valuing the benefits of coastal water quality improvements using contingent and real behaviour. *Environ. Res. Econ.*, **24**, 273–285.

Hansen, N.G. (2004). Nitrogen fertilizer demand from Danish crop farms. *Can. J. Agric. Econ.*, **52**(3), 313–331.

Hasler, B (2005). *Costs and Benefits of Nutrient Reduction to Danish Water 2007–2010*. Funded project under the programme Agriculture, The Neighbours and the Environment. DFFE, København.

Hasler, B., Lundhede, T., Martinsen, L., Neye, S. and Schou, J.S. (2005). *Valuation of Groundwater Protection versus Water Treatment in Denmark by Choice Experiments and Contingent Valuation*. NERI Technical Report No. 543. National Environmental Research Institute, Roskilde. Available at: http://www2.dmu.dk/1_viden/2_Publikationer/3_fagrapporter/rapporter/FR543.PDF

Henriksen, H.J., Kjær, J. and Brüsch, W. (2004). *DØR pesticidinstrumenter og grundvandspåvirkning*. Available at: http://www.dors.dk

Henriksen, H.J. and Sonnenborg, A. (2003). *Ferskvandets kredsløb*. GEUS 2003. Available at: http://www.vandmodel.dk. GEUS, Copenhagen.

Henriksen, H.J., Rasmussen, P., Brandt, G., von Bülow, D. and Jensen, F.V. (2007*a*). Engaging stakeholders in construction and validation of Bayesian belief network for groundwater protection. In Castelletti, A. and Soncini-Sessa, R. (Eds.), *Topics on System Analysis and Integrated Water Resource Management*, Elsevier, Amsterdam, ch 3, 49–72.

Henriksen, H.J., Rasmussen, P., Brandt, G., von Bülow, D. and Jensen, F.V. (2007*b*). Public participation modelling using Bayesian networks in management of groundwater contamination. *Environ. Modell. Soft.*, **22**, 1101–1113.

Henriksen, H.J., Troldborg, L., Nyegaard, P., Sonnenborg, T.O., Refsgaard, J.C. and Madsen, B. (2003). Methodology of construction, calibration and validation of a national hydrological model for Denmark. *J. Hydrol.*, **280**(1–4), 52–71.

Jacobsen, B.H., Abildtrup, J., Andersen, M., Christensen, T., Hasler, B., Hussain, Z.B., Huusom, H., Jensen, J.D., Schou, J.S. and Ørum, J.E. (2004*a*). *Costs of Reducing Nutrient Losses from Agriculture - Analyses Prior to the Danish Aquatic Programme III*. Report 167. Fødevareøkonomisk Institut, Frederiksberg.

Jacobsen, L.B., Andersen, M. and Jensen, J.D. (2004*b*). *Reducing the Use of Pesticides in Danish Agriculture*. FØI Working Paper No. 11/2004. Den Kgl. Veterinær- og Landbohøjskole, København. Available at: http://www.dors.dk

Jakeman, A.J. and Letcher, R.A. (2003). Integrated assessment and modelling: features, principles and examples for catchment management. *Environ, Modell. Soft*, **18**, 491–501.

Jensen, F.V. (2002). *Bayesian Networks and Decision Graphs*, Springer Verlag, New York.

Jensen, J.D., Andersen, M. and Kristensen, K. (2001). *A Regional Econometric Sector Model for Danish Agriculture*. Report No. 129. Fødevareøkonomisk Institut, Frederiksberg. Available at: http://www.foi.dk

Jordan, A., Liefferink, D. and Fairbrass, J. (2004). The Europeanization of environmental policy: a comparative analysis. In Barry, J., Baxter, B. and Dunphy, R. (Eds.), *Europe, Globalisation and Sustainable Development*, Routledge, London, pp. 130–151.

Kjær, J., Olsen, P., Barlebo, H.C., Juhler, R.K., Plauborg, F. Grant, R., Gudmunsson, L. and Brüsch, W. (2001). *The Danish Pesticide Leaching Assessment Programme (Varslingssystem for udvaskning af pesticider til grundvand –VAP)*. GEUS, DJF and DMU, Copenhagen. Available at: http://www.varslingssystem.dk

Kreuger, J. and Nilsson, E. (2004). *Sådan blev pesticidindholdet i overfladevand reduceret i Skåne* (in Danish). DJF Report No. 98, Markbrug, Plantebeskyttelse 1, 129–133. 1. Danske Plantekongress, 13–14 januar, Herning.

Moss, T. (2004). The governance of land use in river basins: prospects for overcoming problems of institutional interplay with the EU Water Framework Directive. *Land Use Policy*, **21**, 85–94.

Newell, B., Crumley, C.L., Hassan, N., Lambin, E.F., Pahl-Wostl, C., Underdal, A. and Wasson, R. (2005). A conceptual template for integrative human-environment research. *Global Environ. Change*, **15**, 299–307.

North, D.C. (1990). *Institutions, Institutional Change and Economic Performance*, Cambridge University Press, Cambridge.

North, D.C. (2005). *Understanding the Process of Economic Change*. Princeton University Press, Princeton, NJ.

Nyegard, E. (Ed.), (2004). *Særligt pesticidfølsomme sandområder: Forudsætninger og metoder for zonering. Koncept for udpegning af pesticidfølsomme arealer, KUPA (in Danish).* GEUS, Copenhagen.

Ostrom, E. (2005). *Understanding Institutional Diversity*. Princeton University Press, Princeton, NJ.

Pearl, J. (1988). *Probabilistic reasoning in intelligent systems*, Morgan Kaufmann, San Mateo, California.

Russel, C.S., Vaughan, W.J., Clark, C.D., Rodriquez, D.J. and Darling, A. (2001). *Investing in Water Quality: Measuring Benefits, Costs and Risk*. Inter-American Development Bank, Washington, DC.

Schou, J.S. (2003). *Miljøøkonomisk analyse af skovrejsning og braklægning som strategier til drikkevandsbeskyttelse* (in Danish). Faglig rapport fra DMU, no. 443. National Environmental Research Institute, Roskilde. Available at: http://www2.dmu.dk/1_viden/2_Publikationer/3_fagrapporter/rapporter/FR443.pdf

Spulber, N. and Sabbaghi, A. (1998). *Economics of Water Resources: From Regulation to Privatization*. Kluwer, Amsterdam.

Stockmarr, J. (2004). Groundwater quality monitoring in Denmark. Review of Survey Activities 2004. *GEUS Bull.* 7, 33–36. Available at: http://www.geus.dk

Swyngedouw, E., Page, B. and Kaika, M. (2002). Sustainability and policy innovation in multi-level context: cross cutting issues in the water sector. In Henialt, H., Getimis, P., Kafkalis, G., Smith, R. and Swyngedouw, E. (Eds.), *Participatory Governance in Multi-Level Context: Concepts and Experience*. Leske and Budritch, Opladen, pp. 107–131.

Topping, C.J. (2005). *The Impact on Skylark Numbers of Reductions in Pesticide Usage in Denmark*. NERI Technical Report No. 527. National Environmental Research Institute, Roskilde. Available at: http://www.dmu.dk

Wurzel, R.K. (2002). *Environmental Policy-making in Britain, Germany, and the European Union: The Europeanisation of Air and Water Pollution Control*. Manchester University Press, Manchester.

Young, R. (2005). *Determining the Economic Value of Water: Concepts and Methods, Resources for the Future*. Future Press, Washington, DC.