Groundwater pesticide vulnerability: comparing simple assessment methods

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Residues of pesticides are found in drinking water and groundwater, in fact in all compartments of the water cycle. Should we protect our water sources better, and if so, how? Four methods have been applied to assess the vulnerability of a phreatic aquifer used for a local drinking water supply serving approximately 1000 people. Of the 24 compounds listed in the case study, a method based on labeled pesticide risk class accepted 14 (58%), a method based on data on pesticide soil-water partition values and very simplified local hydrogeological data also accepted 14 (58%), a method based on model simulation accepted 18 (75%) and a method based on tabulated values of pesticide half-life and soil adsorption values (the so called GUS-index) accepted 8 (35%) of 23 compounds. Taking probable values of high precipitation into account, the accepted compounds based on model simulation were reduced to 10 (42%). None of the 4 methods used all relevant information about risk assessment and drinking water and groundwater protection. The disagreement in the conclusions between the methods is high, ranging from 35 to 60%. Since none of them are consistent, we cannot be confident that they will work under variable conditions. Decisions supported by site specific field or local variables like pesticide half-life in soil and water, soil pesticide adsorption, climatic conditions and others, are recommended.

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Introduction

There is an increasing awareness of the possible risk of pesticide contamination of surface waters and groundwaters. Shallow groundwater to a depth of about 200 m can be characterized as having a mean retention time of 200-300 years, and thus the consequences of potential pollution have long time scales. The drinking water standards in most European countries were originally intended to have a 'no detection' level regarding pesticide residues. Several surveys and studies, however, have shown that there are residues of pesticides in drinking water and groundwater, in fact in all compartments of the water cycle, even in precipitation (Helweg 1995a). The frequency of detectable concentrations seems to be increasing, but this may also be due to improvements in analytical techniques. In Denmark, 8.6% of 1500 analyses from 825 observation wells contained pesticides, according to Helweg (1995b). A study of 284 drinking water wells revealed pesticide residues in 14% of the samples. Similarly in the UK, analysis of nearly 30,000 samples of drinking water showed residues in nearly 30%, about 7% of which were samples from groundwater (Mardsen 1992). An American study reported that in 100 wells sampled for 13 pesticides or metabolites in 1991 and 1992, the frequency of samples with detectable residues of pesticides increased from 29% to 46% when the detection limits were lowered from 0.05 μg/litre (Kolpin et al. 1995). The author did not specify the new detection limits.

In a pesticide survey programme of surface, drainage and groundwater in agricultural areas in Norway between 1994 and 1998, the herbicides mecoprop, MCPA, dichlorprop, bentazone, metribuzin, metamitron, 2,4-D, simazine, linuron, propachlor, the insecticide dimethoate and the fungicides metalaxyl, propiconazole, mancozeb and its metabolite ETU, and fenpropimorph have been detected in surface waters. The highest concentration in surface waters was 19 μg/litre of metamitron. The herbicides bentazone, dichlorprop, linuron, MCPA, mecoprop, metribuzin and metamitron, and the fungicides metalaxyl, prochloraz and propiconazole and tiabendazol were detected in the upper part of the groundwater just below the agricultural fields. The mean concentration in groundwater was 1.03 μg/litre and the highest concentration was 33 μg/litre of metribuzin of 108 positive samples. The most frequently found pesticide was bentazone. The survey also included 4 drinking water wells in which bentazone was repeatedly found. The mean concentration in the drinking water wells was 0.05 μg/litre and the highest concentration was 0.11 μg/litre of bentazone in 12 positive samples.

It seems clear from these studies that if the objective is to keep drinking water free of pesticides, the tools for protecting particularly water recharge but also discharge areas must be improved. This is especially important for groundwater resources because remediation is more difficult and takes a longer time than for surface waters. According to Norwegian standards for drinking water, the maximum concentration of individual pesticides is 0.1 μg/litre, and 0.5 μg/litre total for all pesticides. The use of pesticides within the recharge and discharge areas of drinking water sources in Norway is only allowed when approved by local health and agricultural authorities based on professional evaluation of the actual compounds and their toxicity, potential for bioaccumulation, degradation, metabolites and mobility, together with local climatic factors. These regulations are not always followed up in practice, perhaps because there are no standardised or commonly accepted methods, or the potentially useful methods are too demanding or costly for practical use. A simple monitoring programme will generally not do for groundwater supply sources because remediation will be too costly and time consuming. According to Chilton et al. (1994), both scientific investigations and routine monitoring of pesticides in groundwater present significant difficulties. A strategy needs to be developed to assist water utilities and regulatory authorities to protect groundwater from contamination by pesticides.

The paper describes and compares four simple methods for assessing groundwater vulnerability, relating the methods and the criteria set by the authorities evaluating environmental effects of pesticides and comparing how the methods separate specific pesticides into possible leaching or non-leaching compounds. The objective is to illustrate how differences appear between the methods, and not to analyse if the methods are the most up to date or perform according to ideal assumptions. The comparison is based on a case study where there are very scarce field data to do the evaluation, a situation also often encountered in practice. The methods are used to separate tolerable and non-tolerable pesticide application in the vicinity of a well producing drinking water.

Pesticide regulations in Norway

In Norway, The Norwegian Agriculture Inspection Service approves pesticides for a period of 5 years. By January 1, 1996, 190 pesticides were allowed for use in Norway, including 120 different active ingredients (Morka & Haraldsen 1996). The Royal Ministry of Agriculture gives regulations for distribution and use and the latest version was issued in 1992. These regulations state that pesticides should not have harmful effects on humans, livestock, animals and plants and the environment in general, and they must be ecologically and toxicological acceptable. It is a political objective to reduce the consumption of pesticides in the country. The procedure for approval of pesticides is based on the documentation listed in Table 1.

Methods

Normal agricultural application of pesticides is considered to be a diffuse load, although accidental spills and storage may lead to point source contamination. Pesticide data have been collected mainly from the pesticide database of the Remote Sensing and Modelling Laboratory (1997). Additional information has also been collected from: Tomlin (1994), Hartly & Kidd (1983), Kenaga & Goring (1980) and Morka (1995). Typical values for soil adsorption and pesticide half-life in soil have been chosen if given in the literature. Otherwise 'worst case' values have been chosen, i.e. maximum values for half life and minimum values for soil adsorption.

Object	Subject	Documents on
Trade Product	Toxicology	Acute oral-, dermal- and inhalation toxicity, skin- and eye irritation, allergy.
	Ecotoxicology	Effect on bees and other useful insects, earth worms, aquatic organisms.
	Physical/ Chemical data	pH, density (fluids), particle size (powders), ability to stay in suspension/ emulsion, vapour pressure, flammability.
Active ingredients	Toxicology	Acute oral-, dermal and inhalation toxicity, skin and eye irritation, skin sensibility, 90 days toxicity test (rat), 3 months toxicity test (dog), chronic feeding test (at least 2 types of mammals), carcinogenity, mutagenity (min tests), reproduction (at least 2 or 3 generation study), teratogenity (misforms), nervetoxicity, metabolism in animals (adsorption, distribution, excretion, biotransformation, assumed effect mechanisms, hematology (blood), functions of liver and kidneys, effect on enzymes), toxicity to humans.
	Ecotoxicology	Hydrolysis, photolysis, degradation and metabolism in soil, adsorption to soil particles, transport and mobility in soils (column and lysimeter studies), degradation in soils (half-life time), accumulation in soils, degradation in water (biotic, BOD/COD, half-life time, route of degradation), vaporisation, growth inhibitation of algae, acute and chronic toxicity to daphnia and fish, accumulation in fish, effect on terrestrial microorganisms and earth worms, acute and subacute toxicity to birds, reproduction toxicity in birds, field studies.
	Physical/ chemical data	Boiling point, density, vapour pressure, surface tension, water solubility, lipid solubility, pK_a, K_{ow}, solubility in organic solvents, stability to hydrolysis, thermostability, photostability, flammability, ignition.

Table 1. Information needed for evaluation of pesticides (after Morka & Haraldsen, 1996).

Case study

The four methods have been applied to assess the vulnerability of a phreatic aquifer used as the source for a local drinking water supply serving approximately 1000 people. The well screen is 37-50 m below surface, and the mean depth to the groundwater level has been measured as 12 m from the surface. Groundwater fluctuations are not considered; the variation is assumed to be negligible compared to the depth of the unsaturated zone. The aquifer material varies from fine sand to gravel, with an assumed mean hydraulic conductivity of 8 m/day. The recharge area is about 100 hectares, of which 50% is arable land used mainly to produce strawberries. The names and descriptions of the applied pesticides are listed in

No.	Active ingredient	Type	Chlorine#	Risk class †
1	phenmedipham	Herbicide	-	C
2	glufosinate-NH_4	Herbicide	-	C
3	cycloxydim	Herbicide	no data	C
4	simazine	Herbicide	Cl	C
5	mecoprop	Herbicide	Cl	B
6	metamitron	Herbicide	-	C
7	mecoprop+MCPA	Herbicide	Cl	B
8	di- +paraquat	Herbicide	-	B
9	glyphosate	Herbicide	-	C
10	lenacil	Herbicide	-	C
11	2,4-D	Herbicide	-	B
12	MCPA	Herbicide	Cl	B
13	permethrin	Insecticide	Cl_2	C
14	clofentezine	Insecticide	Cl_2	C
15	azinphos-methyl	Insecticide	-	A
16	fenthion	Insecticide	-	B
17	demeton-s-methyl	Insecticide	-	A
18	fenvalerate	Insecticide	Cl_2	C
19	triadimefon	Fungicide	Cl	C
20	chlorothalonil	Fungicide	Cl_4	B
21	tolylfluanid	Fungicide	Cl_2	C
22	copper oxychloride	Fungicide	-	B
23	quinomethionate	Fungicide	-	C
24	vinclozolin	Fungicide	Cl_2	C

number of chlorine atoms in molecule, - = no chlorine
† Risk class: A=toxic, B=health risk, C=minor health risk

Table 2. Pesticides applied in the case study recharge area of a groundwater well (with grain crops and strawberries production)

Tables 2 & 3, showing that 12 are herbicides, 6 are insecticides and 6 fungicides. Of the 24 applied pesticides, 11 are chlorinated, 5 are indicated as anionic in acidic soil water with a probable high mobility in soil, and 3 are indicated as cationic with low mobility in soil. The annual pesticide application varies from 0.5 to 24 kg/hectare. This case study is representative for risk assessments based on relatively sparse data on local conditions, typical for many groundwater investigations. The available information about local soil and groundwater conditions is limited. It is, however, not unusual that real vulnerability assessment and control of the areal use within the recharge area of a drinking water well are based on less information than this, giving not only justification but also a need for information about the performance of such a vulnerability study based on limited information.

The methods used to evaluate the risk of pesticide leachability to groundwater were based on: (i) the labeled risk class, (ii) data on pesticide water soil-water partition, (iii) model simulation and (iv) the GUS index. The methods are described below.

Method (i), Risk class

Distribution and use of pesticides are regulated by the authorities as described above. The producers classify pesticides as follows:

X=Highly toxic
A=Toxic
B=Health risk
C=Minor health risk

Suggested acceptance criteria:

Pesticides accepted for use in recharge areas are, according to the authorities, assumed to be in class C (Statens Institutt for Folkehelse, 1987).

No.	Active ingredient	K_{oc} log	pK_a log	ionisation#	$t_{1/2}$† days	
1	phenmedipham	3.1 - 4.1	<0.1	0	20-120	
2	glufosinate-NH4	1 - 3.1	nv.	-	3-20	
3	cycloxydim	1.4	4.2	-	1-12	
4	simazine	2.0 - 2.4	12.3‡	+	70-110	
5	mecoprop	-1.4	3.8	-	7-13	
6	metamitron	-0.89	nv	nd.	<28	
7	mecoprop+MCPA					
8	di- +paraquat	2.3 - 4	nv.	nd.	>1000	
9	glyphosate	-5.1	5.7	+	2-174	
10	lenacil	0.3 - 2.8	10.3	+	82-150	
11	2,4-D	-1 - 0	2.6	-	7 -14	
12	MCPA	1 - 2	3.1	-	6-60	
13	permethrin	4 - 5	nv.	nd.	3-200	
14	clofentezine	4.6	nv.	nd.	28-85	
15	azinphos-methyl	1.7	nv.	nd.	5-15	*
16	fenthion	3 - 3.3	nv.	nd.	0-112	
17	demeton-s-methyl	2.6	nv.	nd.	8-63	*
18	fenvalerate	4.1	nv.	0	75-287	***
19	triadimefon	2.5	nv.	nd.	6-23	*
20	chlorothalonil	1.9-2.8	nv.	nd.	5-36	
21	tolylfluanid	2.4	nv.	0	2	*
22	copper oxychloride	nv.	nv.	nd.	nv.	**
23	quinomethionate	0.5	nv.	nd.	3-21	*
24	vinclozolin	2.7	nv.	nd.	3-75	

0 = nonpolar, + = cationic, - = anionic, nv. = no value, nd. = not determined
† Half-life of compound in soil if not stated otherwise
‡ pK_b
* hydrolysed in alkaline media
** strongly adsorbed in soil
*** esther bond hydrolysed

Table 3. Case study application of pesticides in the recharge area of a groundwater well growing grain crops and strawberries, soil adsorption (K_{oc}), chemical characteristics and soil half-life ($t_{1/2}$).

Limitations:
The concept of risk classes is defined only by occupational risk, i.e. risk from direct contact with the active ingredients when handling the products, and gives no information about the risk of leaching to groundwater. It is, however, frequently used to evaluate pesticides. In a listing in Norwegian guidelines for protection of groundwater resources, 21 pesticides are mentioned as examples of compounds accepted for use, of which 10% were in class A, 43% in class B and only 29% in class C (Statens Institutt for Folkehelse, 1987).

Method (ii), Partition coefficient

Transport of pesticides through a soil column depends in part on the potential for sorption to soil particles. Assuming homogenous transport, the velocity of a pesticide relative to water can be evaluated by equation 2. The sorption is described by the partition coefficient, Kd, which is the ratio between the concentration of the substance in the soil and in the aqueous phase at the end of an adsorption test (Kuhnt & Muntau 1994). Usually only the organic carbon partition coefficient, Koc, is given for specific pesticides. The partition coefficients are related by equation 1:

$$K_d = K_{oc} * f_{oc} \qquad (1)$$

where Kd=soil partition coefficient, Koc = soil organic partition coefficient, foc= soil organic fraction (here assumed = 0.05 for local conditions).
Pesticide retardation in the soil column relative to water, R, is given by (Freeze & Cherry 1979):

$$R = 1 + r/n * K_d \qquad (2)$$

where r = bulk density, n = porosity, K_d = soil partition coefficient. If R=10 the substance in question is 10 times slower than water.

Suggested acceptance criteria:
A field location example in Norway, described in Lode et al. (1994), showed that atrazine can be found in groundwater at least 15 years after the last application. If a safety factor is added, a R-value in the order of 100 or higher can be chosen as acceptable for the case study.

Limitations:
The acceptance criteria are subjectively chosen, based on results from a location with application of atrazine, and persistence of atrazine in groundwater (Lode et al. 1994). This might have been a 'worst case' situation where the application was not according to good agricultural practice and thus not relevant for other pesticides and other locations. There is, however, an aspect of precaution. The soil adsorption data are not specified for the case study location. The method does not take into account the amount of the pesticide applied, the degradation of the pesticide and local variation in soil physical conditions and climate.

Method (iii), Model simulation

Based on the fact that the underlying data were limited a relatively simple model was selected:

Pestan
Pesticide Analytical Model 4.0
Ravi & Johnson (undated)

The model uses an analytical steady-state solution of the advective-dispersive-reactive transport equation and includes decay of pesticide in soil and water, actual precipitation, soil depth and geology, and amount of pesticides applied. The following assumptions were used:

A simulation period of 20 years, 1 day between application and start of water infiltration, annual mean precipitation and maximum precipitation expected to occur at least every 5 years.

The PESTAN model is not on the list of models specified in the Regulatory Modelling Working group FOCUS (Boesten et al. 1995), but the selection of a single model may be difficult since there is no clear scientific justification for choosing one over another.

Suggested acceptance criteria:
Based on previous detection of pesticides in groundwater in Norway, and on local hydrogeological conditions, the limit for acceptable infiltration depth below surface of simulated pesticides is 'arbitrarily' set to 480 cm. Thus, for the specified use described here, pesticides which do not infiltrate deeper than 480 cm in 20 years are accepted, even if the concentration exceeds the acceptable limit of 0.1 μg/l. Below 480 cm the concentration of individual pesticides must not exceed 0.1 μg/l.

Limitations:
There are no additional pesticide applications within the simulation period in this model. Normally applications are given regularly either each year or with intervals in real situations.

There are no data on pesticides for the specified location. Local variation in e.g. the geochemistry of soil water can significantly change pesticide solubility and sorption characteristics (McCarty & Jimenez 1985, Thurman 1985, Kan & Tomson 1990). There were only limited data on local soil and hydrogeology. The Pestan model, according to the manual, is insensitive to pesticide solubility, saturated hydraulic conductivity and liquid decay compared to the sensitivity in partition coefficient and precipitation. Soil bulk density, porosity and dispersion coefficient will generally not show a large variation relative to the other variables. The output concentration of pesticide is about 5 times more sensitive to soil adsorption and precipitation than other variables expressed as percentage change in concentration relative to percentage change in the variable.

Method (iv), The GUS-index

According to Gustafson (1989), the most relevant physical variables to leaching of pesticides in soils are K_{oc}, which is the soil partition coefficient KD adjusted for soil organic content, and the half-life value, which is the average time it takes (preferably measured in the field) for soil residues of the parent molecule to decline by 50%. These variables are preferred because much of the variability of leaching of pesticides is assumed to relate to the variability of the soil organic content. The GUS value is calculated according to:

$$GUS= \log_{10}(t_{0.5}^{soil}) \times (4 - \log_{10} (K_{oc})) \qquad (3)$$

where $t_{0.5}^{soil}$ is the half-life of pesticide in soil (days), and K_{oc} is the soil organic partition coefficient given in Table 3

Acceptance criteria:

According to Gustafson (1989), data show that pesticides with a GUS value above 2.8 invariably leach to groundwater, pesticides with GUS values between 1.8 and 2.8 are in a transition zone, and pesticides with GUS value below 1.8 are expected not to leach. The selected acceptance criterion is thus a GUS value equal to or lower than 1.8.

Limitations:

The Gustafson method can only be applied to uncharged pesticides. At least 8 of the case study pesticides can be classified as anionic or cationic and are thus not well evaluated by this method, as seen in Table 3. The method does not consider variability in hydrogeology or climate. Almost no regional or local field data are available for the case study. Many pesticides are ionic substances that do not relate strongly to organic content, but have more affinity to clay particles, for example glyphosate, see Table 3. Thus, the K_{oc} value might not be relevant for the actual leaching conditions.

The analysis of the methods and the case study is only theoretical but could be validated after field investigation after the groundwater well has been operated for some time and if the given pesticide application is continued.

Results and discussion

Table 4 shows that of the 24 compounds given in the case study, method (i) accepted 14 (58%), method (ii) also accepted 14, method (iii) accepted 18 (75%) and method (iv) accepted 8 of 23 (35%) compounds. The distribution of the compounds using method (iv) and the curved shape of the GUS acceptance limits are shown in Fig. 1. When taking possible episodes with high precipitation into account, the accepted compounds based on method (iii) were reduced to 10 (42%, Table 5) and eight pesticides change from non-leachers to leachers. Based on given pKa values the compounds 2, 3, 5, 10 and 11 are likely to have anionic behaviour and high mobility in groundwater. Method (iii) predicts all these as leachers.

On a subjective basis, Table 6 shows that none of the 4 methods use all relevant information about risk assessment relevant to drinking water and groundwater protection.

			Method †			
No.	Pesticide	Application ‡	(i)	(ii)	(iii)	(iv)
1	phenmedipham	6	A#	A	A	NA#
2	glufosinate-NH4	10	A	NA	NA	A
3	cycloxydim	6	A	A	NA	NA
4	simazine	3	A	A	NA	NA
5	mecoprop	6	NA	A	NA	NA
6	metamitron	8	A	NA	A	NA
7	mecoprop+MCPA	3	NA	NA	A	NA
8	di- +Paraquat	24	NA	NA	NA	A
9	glyphosate	8	A	NA	A	A
10	lenacil	2.5	A	A	A	A
11	2,4-D	1.5	NA	A	A	NA
12	MCPA	1.5	NA	NA	A	NA
13	permethrin	0.35	A	A	A	A
14	clorfentezine	0.6	A	A	A	NA
15	azinphos-methyl	2.3	NA	A	A	NA
16	fenthion	3	NA	A	A	NA
17	demeton-s-methyl	0.5	NA	NA	A	NA
18	fenvalerate	0.75	A	A	A	A
19	triadimefon	4	A	A	A	NA
20	chlorothalonil	2	-	A	A	A
21	tolylfluanid	10	A	NA	NA	NA
22	copper oxychloride	7	NA	NA	A	no data
23	quinomethionate	0.5	A	A	A	A
24	vinclozolin	1.2	A	NA	A	NA

Table 4. Risk assessment of selected pesticides and groundwater pollution.

† Method i=Risk class, ii=partition coefficient, iii=pestan model, iv=GUS

‡ Amount of pesticides applied [kg/hectare]

A= the pesticide is considered as Acceptable with respect to the risk for leaching to groundwater, NA = Not Acceptable

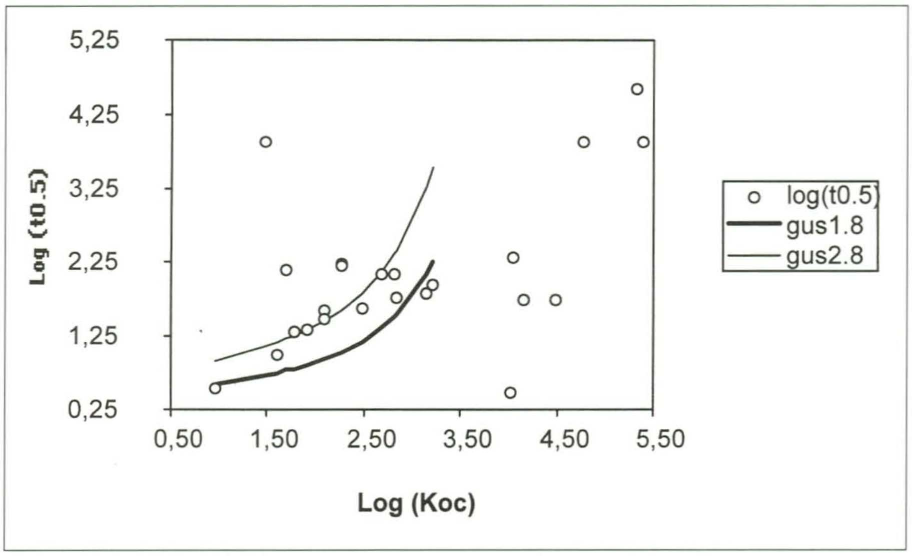

Fig. 1. The GUS-index of the 24 pesticides in the case study relative to GUS-value 1.8, the upper limit for non-leachers, and GUS-value 2.8, above which all the pesticides are considred to be leaching. The GUS-index depends on the soil adsorption value, K_{oc} (-), and the pesticide soil half-life, $t_{0.5}$ (in days).

Although here I try to apply the same assumptions for the methods used, the disagreement between the results from each method is probably due to different assumptions and characteristics of the methods. The disagreement between the methods is high, ranging from about 35 to 60% (Table 7). Since none of them is consistent, we cannot be confident that they will work under variable conditions.

No.	Pesticide	Method iii & mean precipitation	Method iii & high precipitation	Max. conc. †
7	mecoprop+MCPA	A ‡	NA	2.7
9	glyphosate	A	NA	0.5
11	2,4-D	A	NA	13
12	MCPA	A	NA	15
15	azinphos-methyl	A	NA	10
17	demeton-s-methyl	A	NA	20
19	triadimefon	A	NA	47
20	chlorothalonil	A	NA	35

† Estimated maximum concentration of pesticide in groundwater (μg/l)
‡ A= acceptable risk for groundwater pollution, NA = unacceptable risk

Table 5. Model output by Pestan when precipitation is changed from the mean to the maximum value.

Until now the use of pesticides in the vicinity of water sources in Norway has been evaluated by simple methods, and it is reasonable to assume that this will be the case also in the near future. Therefore, this study has been restricted to an evaluation of simple methods and it seems reasonable to suggest that groundwater vulnerability should not be evaluated by an arbitrarily chosen method. It is probably important to use the results from an active surveillance programme on pesticide leaching to groundwater from agriculture, industry, transportation infrastructure and urban garden areas, if they are available. Ramsay (1995) investigated the use of chemical indicators for groundwater vulnerability from pesticides, and concluded that a groundwater age indicator and adsorption indicator could be used. The study was based on supply wells in Denmark containing groundwater with pesticide concentrations exceeding the regulatory limit of regulatory limit of 0.1 mg/l. The age indicator was the ratio between the sum of Ca2++Mg2+ and HCO3-, with values less than 1 indicating good protection. The adsorption indicator was the ratio between Na and Cl, with values greater than 1.5 indicating low vulnerability. Other indicators like nitrate, sulphate, electrical conductivity can also be used.

Object	Subject	Method i Risk class	Method ii Part.coeff.	Method iii Pestan	Method iv GUS
Trade Product	Toxicology	High	No	Low	Low
	Ecotoxicology	No	No	Low	No
	Physical/ Chemical data	No	Low	High	Low
Active ingredients	Toxicology	Medium	No	Low	Low
	Exotoxicology	No	Low	Low	Low
	Physical/ chemical data	Low	Low	High	Low
Drinking water standards		No	No	Yes	No

Table 6. Subjective estimation of the capacity (no-low-medium-high- complete) to assess risk based on use of different kinds of background data.

Method		**Method ‡**		
	(i)	**(ii)**	**(III a)**	**(IIIb)**
Risk class (i)	0			
P. coeff. (ii)	50	0		
Pest.low (iiia)	50	33	0	
Pest. high (iiib)	35	42	33	0
GUS (iv)	41	52	57	39

† Disagreement = (number of pairs of assessment with different outcome)/(total number of pairs of assessments)

‡ Method i=Risk class, ii=partition coefficient, iii a=pestan model with normal precipitation, iv=GUS index

Table 7. Disagreement † (in %) between risk assessment with different methods.

Questions remain to determine how robust relevant assessment techniques are against variability in soil water solubility of pesticides due to variable water quality because of, e.g., fertiliser use or other environmental factors such as soil water organic content, humidity, pH and temperature. It is important to characterise local but representative field variables such as pesticide half-life in soil and soil water, soil pesticide adsorption and/or others. Other important factors are soil textural and structural features, which allow preferential and even flow of larger, non-colloidal particles to great depths. It is further important to incorporate all the aspects listed in Table 1 when assessing risk from pesticides of drinking water sources. Work is needed to formalise the link between available or wanted background information and practicable tools for risk assessment of pesticide usage.

Conclusions

Of the 24 compounds and their specific applications given in the case study, method (i) based on risk class accepted 14, or 58%, method (ii) based on data on pesticide soil-water partition values also accepted 14, method (iii) based on model simulation accepted 18, or 75%, and method (iv) based on the so called GUS-index accepted 8 out of 23, or 35% of the compounds. Taking possible episodes with high precipitation into account, method (iii) accepted 10 compounds (42%). The disagreement between the methods is high, ranging from 35 to 60%. Since none of them is consistent, we cannot be confident that they will work under variable conditions. In view of the fact that some of the estimation methods, particularly methods (i) and (ii), are inadequate, this is not surprising.

A risk analysis can be made on many levels from the simple to the complicated. This study shows that choosing a too simple approach can give unacceptable results compared to more sophisticated methods. Care should be taken that the input values chosen are representative for the local problem. This is especially important when taking mean values over space and time. Generally, it is not desirable to use complicated models without a detailed knowledge of the critical variables in the system, i.e. their distribution and uncertainty. Future risk assessments will probably be based on combinations of simple assessments from the compound's physical, chemical and toxicological qualities and more sophisticated and field validated model simulations based on the models listed in the Regulatory Modelling Working group FOCUS (Boesten et al. 1995).

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