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Ecotoxicological Evaluation of Soil Quality Criteria

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To implement the Soil Protection Act of 1986, the Dutch Ministry of Housing, Physical Planning, and Environment has recently proposed a list of soil quality reference values. These values are, as yet, insufficiently based on ecotoxicological evidence. In this paper, a three-step procedure of risk assessment for soil contaminants is proposed. Arguing from experimental results concerning no observed effect concentrations for a set of selected soil organisms, the method aims at protecting a certain fraction of soil life, taking factors such as soil organic matter and clay content into account. When applied to cadmium, a concentration protecting 95% of soil invertebrates is estimated as 0.16 μg/g for a standard soil. The value of 0.8 μg/g, as proposed by the Dutch authorities, may, given the present variation and uncertainty of toxicity data, protect about 85% of the soil invertebrate fauna. It is concluded that even low levels of cadmium in soil may endanger the functioning of some sensitive soil animal species.

INTRODUCTION

Ecotoxicological risk analysis is the process of extrapolating, in a series of steps, acute toxicity data for test species, to estimate environmental concentrations that may have a damaging effect within ecosystems (Suter *et al.*, 1985). This extrapolation process meets many difficulties; these relate to our incomplete understanding of the functioning of ecosystems. This is particularly true for the soil environment, where ecotoxicology, as well as fundamental ecology, lags behind aquatic sciences.

Several extrapolation methods have been proposed. One approach is to draw upon a large data set of toxicity values and analyze this by means of regression techniques (Suter *et al.*, 1985; Blanck, 1984; Slooff *et al.*, 1986). For example, toxicity of a substance X to species A is predicted from its toxicity to species B; this is done by means of a regression line in a log-log plot of LC_{50} values of A and B concerning several substances already tested on both species. This approach cannot be followed in soil ecotoxicology since there is no large data base to draw upon. Furthermore, the absence of theoretical support underlying the log-log regression and the lumped treatment of various kinds of chemicals make the approach unattractive to us. We have tried to develop an extrapolation model where each step is based on pertinent ecotoxicological theory. This will strengthen the case for ecological risk estimation and the safety margins contained therein.

Kooijman (1987) has developed a model that allows for differences between species exposed to a single toxicant, and also allows for uncertainty due to the extrapolation being based on a restricted number of test species. In this paper we largely follow his model, but we have altered the protection criterion; this drastically changes the outcome of the method. Kooijman estimates a safety factor such that the most sensitive species in a community is protected from lethal effects of a chemical. In his model, it is assumed that the LC_{50}'s of species in a community follow a continuous,

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symmetric distribution on a log scale. Accordingly, the most sensitive species may have an extremely low LC_{50}, especially when the community is large. The extrapolation goes down into the utmost left tail of the distribution where no observations can be made. The method also requires the number of species in the community to be specified, which introduces an element of arbitrariness. To protect the most sensitive species, a very large safety factor is often required.

As an alternative to Kooijman's model, it is argued here that, to protect an ecosystem, it is not necessary to protect even the most sensitive species. Instead, small effects are considered acceptable in the light of the resilience and the regulatory capacity of ecosystems. Following from this principle, a safety factor can be derived that is not sensitive to the utmost tail of the frequency distribution and is not dependent on the number of species in the community to be protected.

The following was first concisely presented at a Dutch Symposium on Soil Quality in 1986 (Van Straalen, 1987). Following discussions with several people, there have been some modifications; the method is described in full here.

OUTLINE OF EXTRAPOLATION METHOD

First, we propose that the no observed effect concentration (NOEC) is the appropriate base for an ecological risk assessment. Extrapolating a no effect level from LC_{50}'s by means of a fixed application factor seems inappropriate in the light of the substance-specific and species-dependent relation between NOEC and LC_{50} (cf. Van Straalen *et al.*, 1989). We believe that NOEC values are more representative for the field situation than LC_{50} values; the same position is taken by Ma (1982).

Second, we propose that the NOEC should be determined on the basis of ecologically relevant criteria for a number of test species. Test organisms should preferably be selected on the basis of their "representativeness" to the community to be protected. Suggestions for criteria to be used to construct a relevant set of test organisms are:

Ecological function: the set should include primary producers, consumers, and saprotrophs.

Anatomical design: the set should include species from different taxonomic groups since sensitivity is often correlated with physiologically determined mechanisms differing between taxa (LeBlanc, 1984).

Exposure route: the set should include species exposed to chemicals in different ways (from the soil solution, from ingested soil material, from soil air).

When NOEC values have been determined, the following steps are necessary to implement a risk analysis:

1. Where different types of soil were used in toxicity experiments, data should be normalized.
2. As species differ greatly in their sensitivity to chemicals, a safety factor should be applied to protect most of the community.
3. To account for modification of toxicity under field conditions, a laboratory-field extrapolation factor should be estimated.

Each of these points is elaborated below. A numerical example for cadmium is given.

In applying the procedure we have, thus far, restricted ourselves to soil invertebrates (earthworms, springtails, mites, woodlice, etc.); another main component of soil life is the bacteria and fungi, these are usually judged on the basis of the processes they catalyze (carbon dioxide evolution, nitrification, etc.). Although it is possible to incorporate microflora in the procedure, there are some details specific to microorganisms that may make a separate treatment of this group preferable (cf. Domsch *et al.,* 1983).

INFLUENCE OF SOIL FACTORS

The first step in the procedure is to normalize the toxicity data. This is necessary since the toxicity of chemicals in soil is modified by a number of physicochemical soil factors. Organic chemicals tend to bind to dead organic matter; the adsorption equilibrium being largely determined by the polarity of the molecule. Heavy metals may form complexes with humic substances in the soil and also bind to clay minerals; the concentration of the free metal ion is further influenced by the soil's acidity. For example, Van Gestel and Ma (1988), in a study of the toxicity of chlorophenols in earthworms, compared two soils. It appeared that earthworm toxicity was significantly higher in the soil with a low level of soil organic matter; this difference disappeared when LC_{50} values were expressed in terms of the concentration in the soil solution. It is clear that any statement on the toxicity of a chemical in soil is useless without a specification of the soil's properties.

The Dutch Ministry of Housing, Physical Planning, and Environment has provided a list of reference values for contaminants in a clean soil, each value being a function of soil organic matter and clay content. The list was based on, among other things, a study of "background" levels of contaminants in Dutch soils from rural areas and nature reserves (Edelman, 1983). It was assumed that these soils still met the requirements of multifunctionality, in the sense of the Dutch Soil Protection Act. Following an analysis of the relationship between these background levels and various soil properties (Lexmond *et al.,* 1986), for each contaminant on the list a linear equation was derived, giving the relation between the reference value and the soil's organic matter and clay content. (For organic chemicals, only soil organic matter was considered.) For example, for cadmium the following relationship is given (MPV, 1987):

$$R_{\mathrm{Cd}} = 0.4 + 0.007(L + 3H), \tag{1}$$

where R_{Cd} = Cd − reference value in μg/g; L = lutum content (particles < 2 μm), in % (w/w); H = organic matter content in % (w/w).

From the formula, it can be seen that in soils with high clay content and high organic matter content, a higher level of cadmium is considered acceptable than in soils that are low in these factors; organic matter content is weighted heavier than clay content. A soil with $H = 10$ and $L = 25$ is defined as a "standard soil"; its reference value for Cd is 0.8 μg/g.

We propose that the relationships in the Dutch list of reference values can be used for provisionally extrapolating between toxicity data from various sources. Each NOEC can be corrected for differences in soil factors by applying the following normalization:

$$\widehat{\text{NOEC}} = \text{NOEC}(L, H)\frac{R(25, 10)}{R(L, H)}, \tag{2}$$

where $\widehat{\text{NOEC}}$ = NOEC valid for standard soil; NOEC (L, H) = NOEC at experimental L, H; R (25, 10) = reference value for standard soil ($L = 25$, $H = 10$); R (L, H) = reference value at experimental L, H, calculated by means of a substance-specific equation, such as Eq. (1).

This should not be taken to mean that the approach is necessarily the best one. For example, no attention is paid to soil pH, which may greatly influence the toxicity of heavy metals, organic bases, and organic acids. Further research could add more qualifying factors.

VARIATION IN SENSITIVITY

The second step in the procedure is to allow for differences between species; these often constitute the greatest source of variation in toxicity data (Slooff *et al.,* 1983). In this section we will develop a model by which a safety factor can be derived allowing for differences between species. The model is largely based on Kooijman (1987), but differs in one crucial aspect: while Kooijman estimates the HCS, i.e., a hazardous concentration for the most sensitive species in a community, we estimate the HCp, i.e., a hazardous concentration for p% of the species in a community. Kooijman's criterion is much more restrictive and leads to greater safety factors than in our method.

Assume that, with respect to one specific substance, the distribution of NOEC values of species within a large community can be described by a loglogistic function. Then, when a species is selected at random, the probability that its ln (NOEC) falls between x_1 and x_2 is equal to

$$\int_{x_1}^{x_2} f(x)dx,$$

where

$$f(x) = \frac{\exp\left(\frac{\mu - x}{\beta}\right)}{\beta\left[1 + \exp\left(\frac{\mu - x}{\beta}\right)\right]^2}. \tag{3}$$

In this formula, μ and β are parameters, where μ corresponds to the mean and β is a measure of the width of the distribution.

The logistic distribution is very similar to the normal distribution (see Fig. 1). However, unlike the normal distribution, the logistic distribution can be integrated analytically, which considerably simplifies the formulas to be derived below.

We now define a hazardous concentration for p% of the species (HCp) as a value of x, such that the probability of selecting a species with an NOEC smaller than HCp is equal to δ_1, where δ_1 is an arbitrary small number equal to $p/100$, such as 0.05. In other words:

$$\int_{-\infty}^{\ln \text{HC}p} f(x)dx = \delta_1,$$

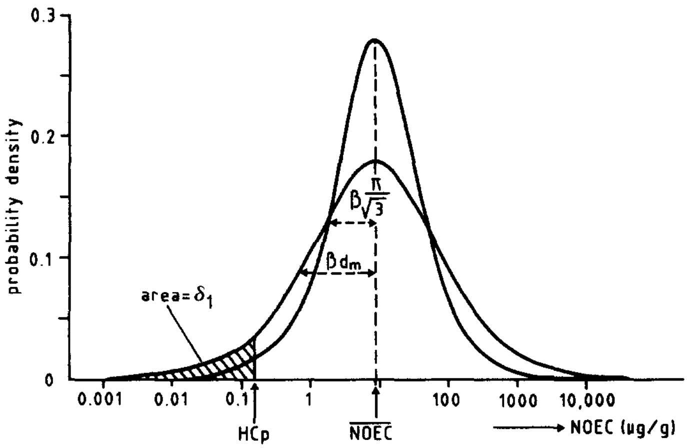

FIG. 1. Loglogistic distribution of NÔEC values of soil organisms, with respect to Cd. Numerical values for parameters were taken from Table 3. The peaked distribution is the one estimated directly from the mean and standard deviation of NÔEC values of tested species. The broadened distribution is the one allowing for uncertainty in the parameter estimation, due to only seven species being tested. For explanation of symbols, see Table 3.

where $f(x)$ is given by Eq. (3). After integration, this can be written as

$$\frac{1}{1+\exp\left(\dfrac{\mu-\ln \mathrm{HC}p}{\beta}\right)}=\delta_1,$$

and after some manipulation as

$$\mathrm{HC}p=\exp\left[\mu-\beta\ln\left\{\frac{1-\delta_1}{\delta_1}\right\}\right]. \tag{4}$$

The parameters μ and β are estimated from a series of NOEC values, derived from chronic toxicity experiments. Assume that m species have been tested and that they are a random sample from the community. The mean of their ln(NOEC) values is denoted by x_m, and the standard deviation of these values by s_m. Then μ and β can be estimated from x_m and s_m by

$$\hat{\mu}=x_m \quad \text{and} \quad \hat{\beta}=\frac{s_m\sqrt{3}}{\pi}.$$

We therefore obtain an estimate of HCp as

$$\mathrm{H\hat{C}}p=\exp\left[x_m-\frac{s_m\sqrt{3}}{\pi}\ln\left\{\frac{1-\delta_1}{\delta_1}\right\}\right]. \tag{5}$$

Since H$\hat{\mathrm{C}}p$ is an estimate, it is subject to an error of estimation. This is an undesirable situation: the probability that the real HCp is still smaller than H$\hat{\mathrm{C}}p$ is 50%. There is a fair chance that more than p% of the species in a community are affected, when HCp is estimated by Eq. (5).

One source of error in $\hat{HC}p$ is the number of tested species being limited to m. This may cause an error both in $\hat{\mu}$ and $\hat{\beta}$, but Kooijman (1987) showed that only the error in $\hat{\beta}$ is relevant to the estimation of HCp. To avoid an overestimation of HCp due to this error, a second logistic distribution is introduced, with the same mean, but with a greater standard deviation (Fig. 1). Kooijman (1987) demonstrated that, if the standard deviation of this distribution is taken to be βd_m, then the probability of overestimating HCp can be set equal to a small number δ_2. The factor d_m was approximated by computer simulations, assuming that $\hat{\mu}$ and $\hat{\beta}$ are random variables following a logistic distribution. d_m depends on m and δ_2 and is given in tabular form in Kooijman (1987) (Table 1). d_m decreases with increasing m; for $m \rightarrow \infty$, $d_m \rightarrow \pi/\sqrt{3}$ (see also Fig. 1).

The shape parameter of the second distribution, β', is now estimated by

$$\hat{\beta}' = \frac{s'_m\sqrt{3}}{\pi} = \frac{\hat{\beta} d_m\sqrt{3}}{\pi} = \frac{3s_m d_m}{\pi^2}.$$

Equation (4) now becomes:

$$\hat{HC}p = \exp\left[x_m - \frac{3s_m d_m}{\pi^2}\ln\left\{\frac{1-\delta_1}{\delta_1}\right\}\right]. \tag{6}$$

This expression may also be written as

$$\hat{HC}p = \frac{\exp(x_m)}{T} = \frac{\overline{\text{NOEC}}}{T}, \tag{7}$$

where T denotes a safety factor given by:

$$T = \exp\left[\frac{3s_m d_m}{\pi^2}\ln\left\{\frac{1-\delta_1}{\delta_1}\right\}\right]. \tag{8}$$

To summarize, using the theory developed by Kooijman, it has been shown that a safety factor protecting (100-p)% of the species in a community is given by Eq. (8); it must be applied to the geometric mean of the NOEC values (Eq. (7), and it depends on the standard deviation of the NOECs (s_m), the fraction of species not protected (δ_1), the number of tested species (m), and the probability of overestimating HCp (δ_2).

The model may also be used inversely, to estimate the risk associated with a certain concentration c. Equating the left hand side of Eq. (6) to c and rewriting the equation, it can be seen that

$$q = 100\left(1 - \left[1 + \exp\left\{\frac{\pi^2(x_m - \ln c)}{3s_m d_m}\right\}\right]^{-1}\right), \tag{9}$$

where q is the percentage of species protected when the environmental concentration equals c ($q = 100\,(1 - \delta_1)$).

LABORATORY-FIELD EXTRAPOLATION

Toxicity data obtained from laboratory experiments may not be valid under field conditions. The "representativeness" of experimental results varies greatly, depending on the experimental conditions. For example, the artificial soil test using *Eisenia fetida* is likely to provide data more relevant to the field situation than the filter paper contact test. In soil ecotoxicology, the situation may be more hopeful than in aquatic

TABLE 1

ARGUMENTS USED IN ESTABLISHING A LABORATORY-FIELD EXTRAPOLATION FACTOR

+ 1. In the laboratory, organisms are tested under optimal conditions.
− 2. In the field, biological availability of chemicals is lower than in laboratory tests.
+ 3. In the field, organisms are exposed to mixtures of many chemicals.
− 4. In the field, ecological compensation and regulation mechanisms are operating.
− 5. In the field, adaptation to chemical stress may occur.
+ 6. Adaptation often entails costs in ecological performance.

Note. A plus indicates a positive argument to maintain an extrapolation factor greater than one, a minus sign indicates a negative argument.

toxicology since the soil acts both as a medium and as a buffer determining sorption equilibria, while in aquatic toxicology, many tests are done in the absence of complexing agents that reduce toxicity in the field.

In choosing a laboratory-field extrapolation factor, the main argument is whether it should be greater or smaller than one. Table 1 provides some often heard arguments in this discussion. It can be concluded that no decisive answer can be given. It seems to us that, as yet, the laboratory-field extrapolation factor should not be taken to differ from one.

The laboratory-field extrapolation problem may well require a species-dependent treatment. The population consequences of toxic action are not the same for each species. Comparing soil invertebrates with diverging life histories, Van Straalen *et al.* (1989) concluded that the population growth rate of the oribatid mite *Platynothrus peltifer* was sensitive to cadmium stress, while population growth of the collembolan *Orchesella cincta* was less sensitive. Development of ecotoxicological theory, incorporating ecological population theory, may contribute to identifying conditions where the field situation will differ significantly from predictions based on laboratory tests.

Since ecological factors (life history, regulating agents, competitors) are essential for estimating toxicity under field conditions, it is essential that the ecology of test species be known. In addition to arguments given above, this is a further issue of concern when constructing a set of test species.

NUMERICAL EXAMPLE

We have applied the above methodology to data for effects of heavy metals and pesticides on soil invertebrate fauna. Results for cadmium will be discussed here. Table 2 lists a number of NOEC values from the open literature concerning various soil invertebrate species. In most cases the NOECs were not given as such by the author(s), but were read from figures or tables. When other experimental conditions were varied in conjunction with cadmium, the NOEC determined with other conditions being optimal was taken.

Only NOECs referring to reproduction were used in the calculations. This is supported by the observation that population development of soil invertebrates is greatly influenced by reproduction changes; for this type of animals, maintenance of reproduction is of great ecological importance (Van Straalen *et al.*, 1989). It seems unwise to mix up different criteria as the assumption for the loglogistic distribution of sensitivities may then be invalid. Note that reproduction may not always be the most

TABLE 2

NOEC VALUES FOR TOXICITY OF CADMIUM TO REPRODUCTIVE PARAMETERS OF VARIOUS SOIL ANIMALS

Species	NOEC (μg/g)	L (%)	H (%)	N$\widehat{\text{OE}}$C[a] standard soil (μg/g)	Refs.
Dendrobaena rubida	100	0	5.7	154	Bengtsson *et al.* (1986)
Lumbricus rubellus	10	17[b]	3.4[b]	13.5	Ma (1982)
Eisenia foetida	25	0	50[c]	13.8	Malecki *et al.* (1982)
Helix aspersa	10	0	86[d]	3.63	Russell *et al.* (1981)
Porcellio scaber	10	0	95	3.33	Van Capelleveen (1987)
Platynothrus peltifer	2.9	0	95	0.97	Van Straalen *et al.* (1989)
Orchesella cincta	56	0	95	18.7	Van Straalen *et al.* (1989)

Note. L = lutum content of substrate; H = organic matter content of substrate.

[a] Calculated from NOEC, L and H according to Eqs. (1) and (2).

[b] Values obtained by personal communication.

[c] Assuming horse manure is 95% organic matter, mixed with sand in ratio 10:9.

[d] Assuming food is 95% organic matter, mixed with lime in ratio 9:1.

sensitive measure: for the collembolan *O. cincta* and the woodlouse *Porcellio scaber* growth is more sensitive to cadmium than reproduction.

Each NOEC value was normalized for standard soil by means of Eq. (2). For animals exposed through their food, it was assumed that food was equivalent to a soil with an organic matter content of 95%, except when inorganic material had been mixed through it. The normalization increases some values and decreases others; the geometric mean NOEC of the normalized values is lower and the standard deviation is higher. This is mainly due to some of the arthropod species being very sensitive to Cd when exposed through their food, while some earthworms are less sensitive.

In Table 3 a summary is given of calculations underlying the estimation of HCp. A protection level of 95% has been arbitrarily choosen ($\delta_1 = 0.05$); the corresponding

TABLE 3

SUMMARY OF SYMBOLS AND CALCULATIONS

Parameter	Symbol	Value	Calculation
Number of species tested	m	7	Table 2
Mean of ln(N$\widehat{\text{OE}}$C)	x_m	2.236	Table 2
Geometric mean N$\widehat{\text{OE}}$C (μg/g)	$\overline{\text{NOEC}}$	9.4	exp(x_m)
Standard deviation of ln(N$\widehat{\text{OE}}$C)	s_m	1.618	Table 2
Fraction of soil life not protected by HCp	δ_1	0.05	Arbitrary choice
Probability of estimating HCp too high	δ_2	0.05	Arbitrary choice
Factor dependent on m and δ_2	d_m	2.82	Obtained from Kooijman (1987), Table 1
Safety factor applicable to mean N$\widehat{\text{OE}}$C	T	59	Eq. (8)
Hazardous concentration for 5% of the species (μg/g)	HC5	0.16	Eq. (7)
Percentage of species protected at proposed reference value (0.8 μg/g)	q	85%	Eq. (9)

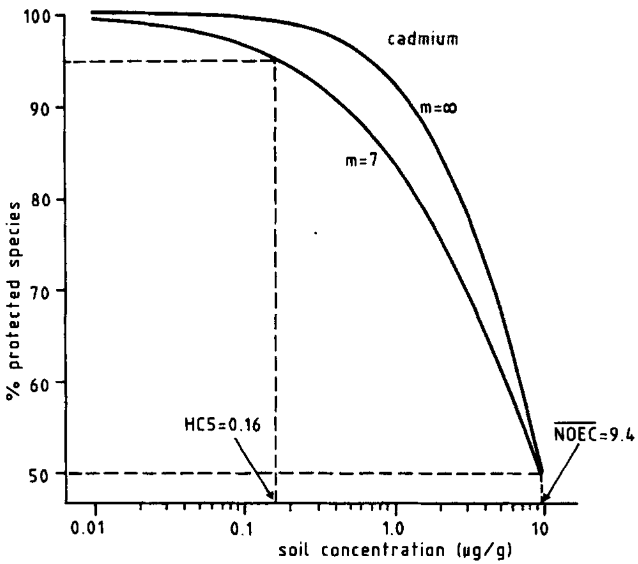

FIG. 2. Estimated percentage of species in a soil community protected from adverse effects of cadmium in soil, calculated from Eq. (9), with parameter values as given in Table 3. The line $m = 7$ is obtained using presently available toxicity data (Table 2). The line $m = \infty$ indicates the protection level when all species in the community would have been tested, other things being equal. For explanation of symbols, see Table 3.

HCp will be denoted by HC5. In accordance with Kooijman (1987), we see no reason for choosing δ_2 different from δ_1, so $\delta_2 = \delta_1 = 0.05$. The estimated HC5 is 0.16 μg/g. Laboratory-field extrapolation is ignored (see above).

No absolute value should be attached to the HC5. Various sources of error may influence its value. One of the most important issues is the representativeness of test species. The model assumes that the set of tested species constitutes a random sample from the community to be protected. In practice, species have been selected for quite other reasons (abundance, experimental manageability, interest of researcher). Another source of error is experimental inaccuracy. Canton and Adema (1978) investigated the reproducibility of well-standardized *Daphnia* tests and concluded that LC$_{50}$ values determined at different places or at different times could differ by a factor of 2. In the present data, tests were not standardized; moreover, NOEC values may be subject to even greater errors of estimation than LC$_{50}$'s. It seems that, for the present data, experimental uncertainty may influence the HC5 by (at most) a factor of 5; the smallest estimate of HC5 would then be 0.03 μg/g.

The model was also used to estimate the percentage of species protected by the proposed standard soil concentration of 0.8 μg/g (MPV, 1987). At this concentration, 85% is protected, given a probability of 5% of over-estimating HCp due to the restricted sample of test species ($\delta_2 = 0.05$). In Fig. 2 the percentage of species protected is given as a continuous function of the soil concentration, using Eq. (9). The two lines in this figure ($m = 7$ and $m = \infty$) illustrate that, with increasing number of test species and "other things being equal," the uncertainty on the representativeness of the sample will disappear ($d_m \rightarrow \pi/\sqrt{3}$). The difference between the lines can be seen as the research effort needed to increase information and decrease the necessary safety factor.

DISCUSSION

The outcome of the risk analysis method outlined above greatly depends on the level of protection one wishes to achieve. This is not a purely scientific problem. In our opinion, a 95% protection level is a reasonable requirement. In the U.S. EPA "Guidelines for Deriving Water Quality Criteria," the same criterion is used for calculating the "Final Acute Value" (Stephan *et al.*, 1985). The method of Kooijman (1987), if applied to the cadmium data, would yield an HCS of 0.000011 μg/g (assuming the protected community to consist of 1000 species). This value is far below the natural background concentration of cadmium in soil. This is due to the assumption that the distribution of log-sensitivities extends its tail to minus infinity. In contrast with Kooijman's model, our method is not so sensitive to the tails of the distribution since it assumes a threshold concentration (HCp) beyond which possible effects are considered acceptable.

Although the criteria applied in this paper are certainly not overconservative or extremely ecocentric, most HC5 values derived are less than the soil quality reference values given by the Dutch authorities. Vegter *et al.* (1988) proposed that the current reference values can be seen as a provisional scientific null hypothesis. Following the iterative process between policy and science, as advocated by them, it would seem that the present values require adjustment. The cadmium case shows that some sensitive soil organisms may suffer from concentrations that are only slightly above background levels. Cadmium appears to be an element that exerts a naturally selective action on the soil community.

The problem of choosing a level of protection for advisory values for environmental quality can be circumvented by presenting the results of risk analysis in the form of Fig. 2. The choice of an environmental standard is then left over to the decision maker. However, the figure demonstrates that a scarcity of scientific information will inevitably lead to a low standard. Kooijman (1987) discusses the implication of research costs in relation to what may be unnecessarily low (costly) standard values.

In comparison to aquatic toxicology, ecotoxicology of soil organisms is only just beginning. For many problematic substances in the soil, there is a great lack of even simple effect data. To fill this gap and to carry on the iterative interplay between science and policy, a major stimulus of soil ecotoxicology is required.

CONCLUSIONS

Derivation of criteria for contaminant concentrations in soil can be based on interspecific variation in sensitivity, as displayed by the various species in a soil community. By modifying a model developed by Kooijman (1987), a contaminant concentration (HC5) can be estimated such that at most 5% of the species in a community is exposed above its NOEC. The numerical example using cadmium data demonstrates that the procedure leads to realistic values. The robustness of the model to changes in the assumed frequency distribution and to various patterns in sensitivity among related and unrelated species has to be further investigated.

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