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Assessment of soil contamination – a functional perspective

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Abstract

In many industrialized countries the use of land is impeded by soil pollution from a variety of sources. Decisions on clean-up, management or set-aside of contaminated land are based on various considerations, including human health risks, but ecological arguments do not have a strong position in such assessments. This paper analyses why this should be so, and what ecotoxicology and theoretical ecology can improve on the situation. It seems that soil assessment suffers from a fundamental weakness, which relates to the absence of a commonly accepted framework that may act as a reference. Soil contamination can be assessed both from a functional perspective and a structural perspective. The relationship between structure and function in ecosystems is a fundamental question of ecology which receives a lot of attention in recent literature, however, a general concept that may guide ecotoxicological assessments has not yet arisen. On the experimental side, a good deal of progress has been made in the development and standardized use of terrestrial model ecosystems (TME). In such systems, usually consisting of intact soil columns incubated in the laboratory under conditions allowing plant growth and drainage of water, a compromise is sought between field relevance and experimental manageability. A great variety of measurements can be made on such systems, including microbiological processes and activities, but also activities of the decomposer soil fauna. I propose that these TMEs can be useful instruments in ecological soil quality assessments. In addition a "bioinformatics approach" to the analysis of data obtained in TME experiments is proposed. Soil function should be considered as a multidimensional concept and the various measurements can be considered as indicators, whose combined values define the "normal operating range" of the system. Deviations from the normal operating range indicate that the system is in a condition of stress. It is hoped that more work along this line will improve the prospects for ecological arguments in soil quality assessment.

Introduction

In many industrialized countries soil contamination has become a serious problem. This is especially the case in regions with a high population density, where land is intensely used and, as a consequence, a polluted location cannot be simply set aside. Soil pollution may be due to a variety of causes (Tarradellas et al. 1997; Eijsackers 1998); some of the most important sources in countries like the Netherlands are:

- Waste dumps of various kinds; both industrial and household waste may pollute the direct surroundings by surface dispersal and groundwater leaching of potentially hazard substances.
- Former coal gas facilities have polluted the soil with tar and other materials, containing a high concentration of polycyclic aromatic hydrocarbons and cyanide.
- Petrol stations have caused pollution with aliphatic and aromatic hydrocarbons by repeated small spills of petrol during tanking and sometimes by leakage from storage tanks.
- Residues of pesticides are present in many agricultural soils, due to many years of heavy application of these compounds.
- Environmental compartments such as sediment and forest soils are polluted with substances emitted from many diffuse sources and accumulating in

those places which have the greatest binding capacity. This concerns heavy metals, organochlorine compounds, etc.

Decisions on how to deal with these problems of contamination are based on a variety of arguments, including the nature and seriousness of the pollution, the destination of the land, and the financial consequences of management decisions. In the Netherlands, soil assessment is based on the Soil Protection Act which states that the present use of the soil should not impede any future use. Various functions of the soil are recognized, such as:

- The soil as a physical base for construction works, housing and technical installations
- The soil as a filter and buffer, allowing clean groundwater
- The soil as a source of valuable materials, including ores
- The soil as a medium of sufficient fertility to allow agricultural production
- The soil as a place where ecological functions are conducted that are relevant for the biosphere in general.

It is also recognized that the ecological function of a soil is one of the most vulnerable functions and puts the most stringent requirements on the presence of chemical pollutants. That is why ecological risk assessments are a critical part of the evaluation, and are often in the firing-line if decisions are made with large financial consequences. The problem is that ecological assessment cannot yet be based on a generally accepted principle, and that there are no ecological standards against which to judge the severity of effects. It is the experience of the author that lack of agreement and instruments implies that ecological arguments often play a minor role in decisions concerning the management of polluted sites.

The aim of the present paper is to analyse (1) why the position of ecology in soil assessment is so troublesome, (2) what guidance may be derived from ecotoxicology and theoretical ecology, and (3) how a perspective can be developed that may help to improve on the situation.

Ecotoxicological approaches

Ecotoxicology is a science that deals with the ecological effects of potentially toxic substances in the environment. The origin of the discipline dates back to the 1970s; the term was first coined in 1969 by R. Truhaut (see Truhaut 1977). The first textbook (in French) was Ramade (1977). Other textbooks followed, including Moriarty (1983), Van Leeuwen & Hermens (1995) and Walker et al. (1996). So it can be said that the science of ecotoxicology is already 30 years old. The question can be raised, why ecotoxicological approaches have not led to a better position of ecology in soil assessments already?

The first remark to be made is that ecotoxicology developed earlier for aquatic ecosystems than for soil. Internationally harmonized methods of testing aquatic invertebrates, fish and algae were developed in the 1970s and a large data base of toxicity data was developed much earlier than for soil organisms. Up to 1995 only one internationally accepted test method was available for soil organisms, namely the earthworm artificial soil test. A suite of new methods was developed in the years 1995–2000 and consequently the data base for soil has now grown considerably (Van Gestel et al. 1997, Løkke & Van Gestel 1998).

A major complicating factor in soil ecotoxicity experiments is that most of the polluting substances are bound to the solid phase of the soil, while the bioavailable fraction of the compound is usually much stronger related to the free concentration in the pore water than to the total concentration in the soil. Consequently, compared to aquatic systems, the soil ecotoxicologist has to deal with at least three compartments (soil, pore water and organism), rather than two (water, organism). Soil chemistry phenomena (sorption to soil, partitioning behaviour, speciation) cannot be ignored in soil ecotoxicity analysis and often the main factors determining changes in toxicity and biodegradation rates derive from changes in soil chemistry.

Another issue explaining why soil ecotoxicology is not yet in a strong position to guide ecological soil assessments is the fact that traditionally, ecotoxicology was mainly involved with the so-called inverse problem, rather than with the forward problem. This may be explained as follows (Van Straalen 1990, 2002).

In ecological risk assessment one can either argue from risk to concentration or from concentration to risk. In the first approach one defines a certain maximally accepted risk of substances in the environment. The task for ecotoxicology is then to derive a concentration level such that this risk is not exceeded. This is traditionally done using toxicity tests, in which organisms are exposed to a graded series of concentrations and effects are measured at each concentration. The concentration corresponding to the maximally accepted effect is then estimated from the results by some regression technique and this may be expressed, for example, as EC10 (10% effect concentration) or EC50 (50% effect concentration). This approach to risk assessment was called the "inverse approach". In the second approach, one argues from concentration to risk: the "forward approach". The starting point here is a site where certain concentrations of pollutants are present. The question is, how large is the risk associated with that pollution? The great difference with the inverse approach is that there is no neat series of exposure concentrations and that a host of complicating factors (other pollutants, organic matter content, clay content, pH, microbial activity, age of pollution) may modify toxicity in comparison to the controlled laboratory experiments.

The inverse approach is the classical ecotoxicological method for deriving maximum acceptable concentrations, which may take the form of quantitative quality critera or standards incorporated in environmental legislation. The problem is that such quality criteria can only give an approximate estimate of risk in specific situations, because of the many modifying factors. In soil assessments, the exceedance or non-exceedance of standards is often insufficient argument to conclude on the presence or absence of risks, because site-specific factors may modify risk to such an extent that it is not simply related to the chemically determined total concentration in the soil, as it is in laboratory experiments. Rather, to support decisions on treatment, clean-up or abstinence of action one needs a site-specific estimate of risk. In short, the forward approach is much more relevant for soil assessments than the inverse approach, however, the forward approach in ecotoxicology has not been developed to the same extent as the inverse approach.

A framework for forward risk assessment of polluted sediments was developed by Chapman et al. (1986, 1990). The term "sediment quality triad" was coined for a system in which three types of information are used to evaluate a sediment of concern: (1) chemical measurements of concentrations of pollutants, (2) results of bioassays using samples from the field site, and (3) field inventories of communities of organisms present at the site. This information can be categorized in a triangle and if the scores exceed certain thresholds, one can decide on action. This approach has been relatively successful for sediment evaluations.

A triade-type of approach was proposed for soil by Rutgers et al. (2000). These authors argued that soil evaluations should be contingent on the intended land-use of the site, that is, depending on whether the site is going to be used as a residential area, an industrial estate or an agricultural field, the endpoints to be used in ecological assessments should be different.

In the triad approach, an important role is played by bioassays; these are experiments in which organisms are exposed to samples taken from the site and their response is observed under standardized conditions. Significant advances have been made recently in developing such bioassays for soil (Van Gestel et al. 2001). The conclusion from this research is that a combination of different bioassays is necessary; risks cannot be derived from effects on one group of organisms. In addition, it is advised to conduct chronic bioassays using organisms exposed in the soil itself, rather than short-term bioassays using organisms exposed to extracts from the soil.

This very short sketch of ecotoxicological approaches has shown that the scientific base of risk assessments of soil is not well developed, for a variety of reasons. It may be expected that the situation will improve significantly in the near future. An important bottleneck at the moment is the absence of a good reference system. Because polluted sites do not have an in-built control (a site which is identical to the polluted site except for the pollution) the results of bioassays cannot easily be placed in a proper perspective. With the further development of bioassay techniques, more data will become available on the "normal" range of outcomes of specific tests and a reference system will develop itself. The question remains, how bioassay test results fit into an overall framework, that is, how can one persuade the regulator and the public that the scores obtained in a battery of bioassays jointly are indicative of unacceptable effects? I will dwell on some holistic concepts of soil protection that may fulfill this role of a framework.

Holistic concepts of soil protection

Many authors have tried to define a general concept of soil quality. By basing the ecological arguments of soil assessment in a commonly accepted definition of soil quality, they could gain convincing power and thus improve on the troublesome condition of ecology in soil assessments referred to above.

One of the concepts that received attention in the 1990s is "soil health". This was inspired to a certain extent by the more general concept of "ecosystem health". The ecosystem health paradigm was introduced in an influential book by Costanza et al. (1992), in which ecologists collaborated with economists to develop a unifying concept of environmental management that would meet the needs felt with regulatory agencies to adopt a broader set of management goals than used until that time. In defining the concept, a parallel was sought with the medical notion of health. Just like human health is not easy to define, but is measured with various, sometimes simple, instruments (e.g., a thermometer), ecosystem health is not easy to define, but captures a general idea that everybody understands and should in principle be measurable. The consensus definition that was formulated by the participants of a workshop held in 1990, which formed the basis for the book by Costanza et al. (1992) was: "An ecological system is healthy and free from 'distress syndrome' if it is stable and sustainable – that is, if it is active and maintains its organization and autonomy over time and is resilient to stress". The normative element implied by the concept of ecosystem health made it very attractive to environmental policy makers, but it received criticism from scientists who denied the idea of nature having an intrinsic value or norm, other than the values to it assigned by man.

It was recognized from the beginning that the concept of ecosystem health would remain useless if it was not operationalized. Operational definitions have mostly come in the form of so-called indicators. The argument is that a complex concept such as ecosystem health cannot be measured as such, but that it can be approached through a series of indicators, each of which will measure a certain aspect; this is comparable to the measurement of temperature: the height of a mercury column is not temperature as such, it is a measure of temperature, and if applied to the human body, a measure of fever (illness) or its absence (health). Rapport et al. (1985), Rapport (1990) and Costanza et al. (1992) have provided a list of possible ecological indicators which could serve the use of defining ecosystem health. In various cases ecological indicator development has found its way to regulatory authorities in the form of models or decision support systems for environmental management (see, e.g., Lorenz et al. 1997).

In addition to ecosystem health and partly in analogy to it, the concept of soil health was also defined. Doran & Safley (1997) put it in this way: Soil health can be defined as "the continued capacity of soil to function as a vital living system, within ecosystem and land-use boundaries, to sustain biological productivity, promote the quality of air and water environments, and maintain plant, animal and human health". Like the definition of ecosystem health cited above, soil health includes the concept of sustainability and it explicitly places soil in a wider context of air and water quality. Also like ecosystem health, soil health needed to be operationalized in terms of indicators. A book edited by Pankhurst et al. (1997a) contains an overview of the various bioindicator approaches available to assess soil health. These vary from microbial biomass, enzyme activities, community structure of nematodes and microarthropods, abundance of earthworms, etc. In the concluding chapter of the book, Pankhurst et al. (1997b) listed five conceptual and four practical difficulties with the selection and use of biological indicators of soil health. These include the high levels of spatial and temporal heterogeneity that affect measurements of most systems and the lack of consistency in bioindicator responses across systems (e.g., different soil types). Another issue is the present lack of base line data which may act as a reference point for soils of defined health levels.

It seems that the concept of soil health is interesting and potentially valuable. Its present state is, however, insufficiently operationalized to fulfill the role of a framework of ecological soil assessments. What we may learn, however, is that indicators can act as instruments for operationalization and that soil health cannot be defined along one dimension, but should include indicators of various kinds. These lessons will return in the concluding section of this paper, but before that, I want to explore the role of ecological theory and evaluate its potential contribution to the question why the position of ecology in soil assessment is so troublesome.

The role of ecological theory

If we look at soil communities from an ecological perspective, a distinction can be made between the structure of a community and its functional attributes. The structure includes all quantities that can still be observed on a snapshot of the community, or when time would stand still. This includes things like species richness, biomass, dominance structure, feeding groups, etc. The functional aspects include the rate of organic matter processing, degradation of substrates, respiration, nitrification, etc. While soil protection is mostly interested in functions, it might achieve this aim through protecting the structure. Guaranteeing

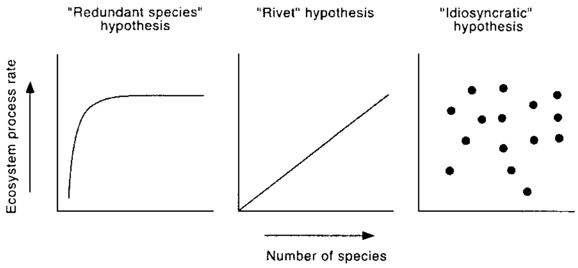

Figure 1. Three hypotheses about how ecosystem processes may depend on the biodiversity of that ecosystem, according to Lawton (1993). Reproduced by permission.

that all species are present, and that biomass distributions, dominance structures, etc. remain unaltered, could be one way of protecting the ecological functions in soil. This approach would assume that there is some relationship between ecological functioning and biodiversity.

The role of biodiversity in maintaining ecological functions has been subject to intense theoretical and experimental analysis in the last years. In general it is assumed that there is an asymmetric relationship between structure and function, that is, protection of functions does not require protection of all structures, while on the other hand protection of structures will always guarantee protection of functions. However, it is still a matter of debate what kind of form this asymmetric relationship should take. Lawton (1994) has proposed three alternative hypotheses that differ from each other in the extent to which a decrease in the number of species endangers an important function of the system (Figure 1). The hypotheses are discussed in terms of a graph in which some ecosystem process is plotted as a function of the number of species in the ecosystem. The argument is, what happens with the ecosystem function (plotted on the vertical axis), when biodiversity (plotted on the horizontal axis) decreases or increases. The three hypotheses are:

- The *redundant species hypothesis*: with a decrease of biodiversity, ecosystem functions are unaffected up to the point where only a small number of key species remains; if one of these species is removed, the system collapses. The idea is that many species in the ecosystem are "redundant" in the sense that their contribution to the ecosystem process can be taken over by other, similar species.
- The rivet *hypothesis*: with a decrease of biodiversity, ecosystem functions decrease proportionally. The idea is that every species makes a (smaller or larger) contribution to the process, so if it is removed, that contribution is subtracted from the process.
- The *idiosyncratic hypothesis*: there is no universal relationship between structure and function, rather, the relationship is ecosystem-specific. In one case there may be a strong reduction of function with a loss of biodiversity, in another case there may hardly be an effect.

The general feeling among soil ecologists is that functional redundancy indeed plays a role in soil communities. The argument is supported by the common observation that soil respiration is considerably less sensitive to the effects of toxic substances than nitrification, which is attributed to the fact that all heterotrophic organisms contribute to respiration, while only a few bacterial genera are responsible for nitrification (Domsch 1984). On the other hand, it has been shown recently that a species-poor community of nitrifiers may nevertheless be responsible for a large and stable activity of nitrification (Laverman et al. 2001). In studies of heavy metal contamination, it has been demonstrated that there was a considerable loss of species of fungi in a gradient of pollution around a metal smelting works, however, respiration of the soil was affected only at very high levels close to the metal source (Nordgren et al. 1983).

Under the influence of the biodiversity debate triggered by the Rio convention, a great ecological research effort was launched in the 1990s to develop a theory about structure and function (Tilman & Downing 1994; Naeem et al. 1994; Naeem & Li 1997; Van der Heijden et al. 1998). Despite these efforts, there is very little empirical evidence at the moment that might substantiate or disprove one of Lawton's hypotheses, although the general notion that biodiversity is important for ecosystem functioning is subscribed by most ecologists. It seems that the question must be phrased differently in order to arrive at meaningful answers. For example, Naeem & Li (1997) have coined the term "ecosystem reliability" to indicate the degree to which functions of an ecosystem are maintained over time and argued that biodiversity plays a role as an "ecological insurance" in the sense that it is one of the factors that contributes to the maintenance of ecological functions. There is also an increasing awareness that not biodiversity as such is important, but biodiversity in relation to the properties of the species (Walker et al. 1999). That is, to evaluate the effects of diminishing species richness on ecosystem processes, we must look at the biodiversity of species attributes in a community, not only at species numbers. This approach is also taken in food-web ecology, in which species are classified in so-called functional groups, before their effect on an ecosystem process is evaluated (De Ruiter et al. 1995; Berg et al. 2001).

One of the important factors that may interfere with the structure-function debate is the nature of the limiting factors. This issue has been very important in analysing nutrient loadings into aquatic ecosystems, but it is also very relevant for soil communities. Despite the fact that organic matter on the average appears to be abundant, microorganisms in soil cannot always grow because they are limited by environmental heterogeneity of nutrients and physicochemical factors. Levine (1989) discussed this issue by making a distinction between capacity-limited processes and substrate-limited processes (Figure 2). Capacity limited processes are not sensitive to changes in the substrate supply rate. Because the functional machinery is already saturated, the introduction of more substrate cannot increase the throughput of material through the community, it will only cause an increase in the amount of substrate that goes into environmental sinks. However, a capacity limited process is very sensitive to environmental toxicants, that will affect the number of functional units. Because the capacity of each functional unit (species or species group) is maximally deployed, a decrease of species richness will directly affect the overall rate of processing (Figure 2 left). On the other hand, in a substrate limited process, the capacities of the functional units are not fully deployed and if such a system is affected by toxicants, the overall throughput may remain unchanged because each functional unit can increase its share in the process (Figure 2 right). The theory of Levine (1989) can thus be seen as a qualification of the biodiversity-function debate mentioned above.

That substrate availability is of utmost importance to evaluate the effects of toxicants in soil was also demonstrated by Van Beelen & Doelman (1997). These authors argued that toxicity tests for microbial activity are wrongly conducted under conditions of high nutrient supply. The fact that soil respiration is an insensitive endpoint should not be taken to imply that the microorganisms conducting soil respiration are not sensitive. When compensatory growth is prevented, the sensitivity of soil microorganisms to toxicants is comparable to that of soil animals. Very little research has been done on the influence of nutrient supply on the sensitivity of microorganisms to toxicants.

This analysis of ecological theory has demonstrated that little guidance can be given at the moment as to the question how to proceed with reinforcing the ecological basis of soil protection. Some general concepts have emerged, but the theory is not yet developed to a stage sufficient for providing strong practical guidelines for soil evaluation. In fact, some authors have proposed that an analysis of toxicant-disturbed systems might actually help to shed more light on this fundamental ecological question (Eijsackers 1994). Rather than trying to argue from ecological theory, I propose that arguing from practical and feasible approaches may be a better way of bringing the issue of ecological soil evaluations further. One such an approach is based on an analysis of responses in terrestrial model ecosystems.

Terrestrial model ecosystems

Traditionally, ecotoxicological effects of soil pollutants are assessed using single species toxicity tests, in which endpoints such as 50% effect concentrations are estimated from the performance of a test species exposed to a graded series of exposure concentrations of the pollutant of concern. The results of these experiments have significantly improved our understanding of the hazards of potentially toxic compounds and in several cases a good correspondence between effects predicted from laboratory experiments and population responses in the field has been observed (Van Gestel 1992; Heimbach 1992; Van Straalen & Van Rijn 1998). At the same time it is recognized, however, that single species laboratory tests cannot predict effects that critically depend on the interaction between two or more species. For example, effects of soil toxicants on microarthropods and earthworms are often mediated by effects on soil microorganisms or on gut microflora (Pokarzhevskii et al. 2000). That is, the primary effects of the toxicant may be on the food or on the symbiontic microflora of an animal, rather than on the animal itself. In such cases, effects in communities cannot be understood unless the interactions between animals and microorganisms are taken into account (Van Wensem et al. 1997).

Many ecotoxicologists have studied the effects of toxicants under conditions where interactions between organisms are involved. This is often done in experimental systems that allow enough complexity for such interactions to develop, while at the same time retaining experimental manageability and replication; these systems are usually designated as “microcosms” or “model ecosystems” (Morgan & Knacker 1994; Ed-

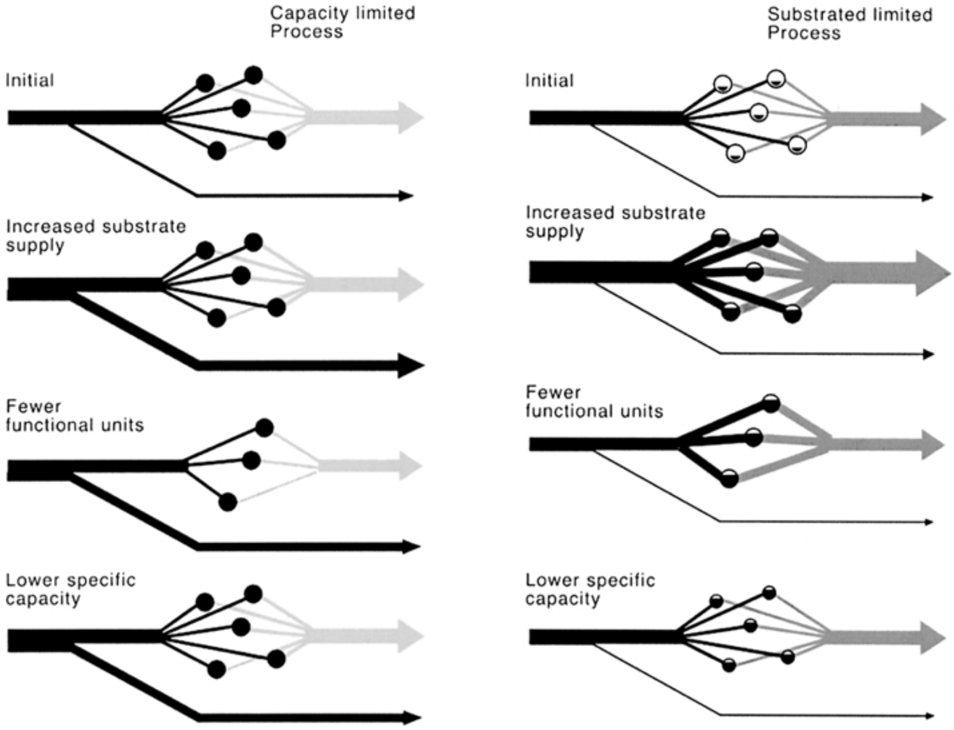

Figure 2. Illustrating the different ways in which toxicants may affect the rate of throughput of a substrate through an ecosystem, depending on whether the system is limited by the capacity of its functional units (left) or by substrate availability (right). Reproduced from Levine (1989), with permission.

wards et al. 1996; Verhoef 1996; Sheppard 1997). An example is given in Figure 3. An excellent review of the design of microcosms and their use in ecotoxicological testing is given in Sheppard (1997). The scientific literature shows an increased interest in these systems in recent years (Komulainen & Mikola 1995; Salminen et al. 1995; Salminen & Haimi 1996; Edwards et al. 1996; Bogomolov et al. 1996; Parmelee et al. 1997; Martikainen et al. 1998; Vink & Van Straalen 1999). In international co-operation programmes, a.o. sponsored by the European Union, significant progress has been made in the standardization and field validation of a type of soil microcosm, called terrestrial model ecosystem (TME). Characteristics of this approach are:

- Use of undisturbed soil columns taken from the field (rather than artificially reconstituting a soil column from separate materials).
- Inclusion of living vegetation growing on the soil (rather than using only the soil itself or only the litter layer). This will allow for interactions between soil-living organisms and plant roots.
- Taking a column with a content of several liters (rather than the small systems of a few centimeters used as microcosms). This will allow for larger animals such as earthworms to develop more or less normally and it also takes away part of the microscale variability.

TMEs are incubated in the laboratory under artificial daylight and constant ambient temperature and incubated for a certain period (e.g., 4 months), while measurements are made during incubation and upon termination of the experiment. The columns are usually equipped with funnels that will allow soil leachates to be collected in flasks. Replicated systems are to be taken, for example in a site suspected of pollution and in a reference site, or along a gradient. Differences in the performance of systems taken at different sites may be evidence of altered ecological functioning of the soil.

A great variety of variables can be measured in TMEs: nutrient concentrations in the leachate (e.g., ammonium, nitrate, sulphate, phosphate), evolution of gases (e.g., carbon dioxide and nitrogen oxide), decomposition of organic matter (using, e.g., bait

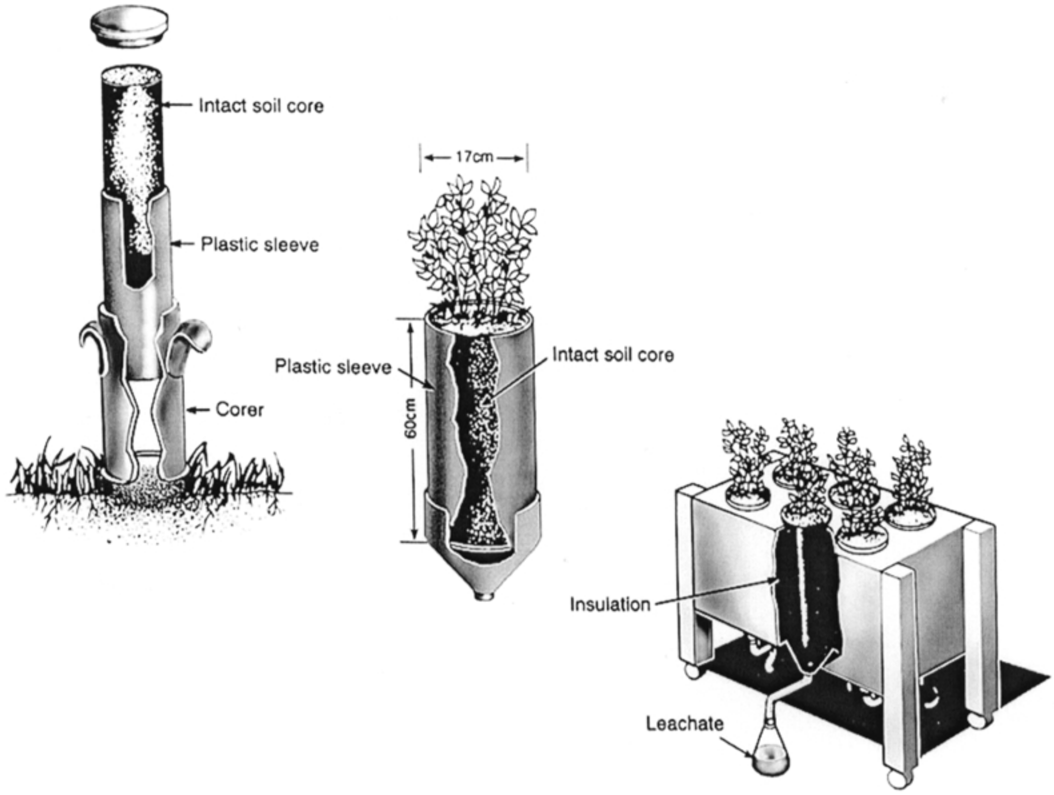

Figure 3. General design of a terrestrial model ecosystem, after Sheppard (1997).

laminas, cotton strips or litterbags), microbial biomass (e.g., by substrate-induced respiration), microbial community structure (e.g., using DGGE or PLFA profiles), microbial physiological diversity (e.g., with BIOLOG), enzyme activities (dehydrogenase, phosphatase, etc), invertebrate populations (following extraction with Tullgren funnels), etc. Jointly these variables constitute a mixture of structural and functional parameters. Some measurement endpoints combine a structural with a functional aspect; for example, bait laminas (sticks that carry a specified substrate in holes, which are introduced in the soil and assessed for the number of empty holes after incubation) are often considered as a typical functional measure (organic matter breakdown, see Kula & Römbke 1998), however, if there are earthworms in the column these will feed on the bait, and so bait laminas can also be considered as an indirect measure for earthworm biomass, a typical structural property.

In the light of the previous discussion in this paper, TMEs are one way of implementing the forward approach of ecotoxicological risk assessment. Since TMEs are taken from the field, their responses learn us something about site-specific conditions. The standardized way of incubation makes sure that transient factors such as temperature and humidity will not have an overriding influence on the variables to be measured. For the same reason, TME experiments cannot last too long, because then the responses will depend on the incubation conditions rather than on the original conditions in the field.

A bioinformatics approach to ecological soil assessment

The variables measured in model ecosystem experiments, as mentioned above, can all be recorded on a single soil column. This means that each column can be considered as a case that is characterized by a multidimensional state. By considering all the measurements simultaneously a much better characterization of the state of the system is obtained, than each variable can provide on its own. The state can be considered as a hypervolume in a space of n dimensions, where n is the number of variables measured on a single system. I call this a "bioinformatics approach" to soil assessment, because, like in the analysis of DNA sequences, one needs to deal with a large amount of information at the same time. The analytical apparatus fit to treat data like this is multivariate statistics.

One of the first authors who has proposed a multivariate approach to (aquatic) microcosm analysis was Kersting (1984). He has introduced the concepts "normal operating range" and "normalized ecosystem strain" as parameters that summarize the information content of the data and express it in a simple score (Figure 4). Suppose we have a system characterized by two state variables. When the system is not perturbed, all combinations of the variables fall in the "normal operating range" (NOR) of the system. Mathematically, this may be defined as the 95% confidence area of unperturbed states. When more than two state variables are under consideration, such an area can still be defined as an n-dimensional volume. Subsequently, Kersting (1984) defined a quantity called "normalized ecosystem strain" (NES) as the distance between a certain state and the centre of the 95% confidence area, divided by the width of the confidence area at the same place (Figure 4). This definition assures that NES is greater than unity when the system is out of its NOR, and smaller than unity when it is within the NOR. Thus NES is a simple score that summarizes the state of a system, taking into account the multivariate nature of the data.

One of the advantages of a bioinformatics approach is that the discussion about which variables to choose for soil assessments and whether structural or functional parameters should be included, can be avoided; we do not have to wait until ecological theory has given an answer to the questions about structure and function, we just take as many variables as is practically possible. After statistical analysis, the data themselves will show which of the variables contribute most to the definition of NOR and which variables only produce noise. The great advantage of using TMEs to define the normal operating range of soils is that these systems represent reproducible coherent units on which a great number of mutually connected measurements can be made.

The approach advocated here is related to the statistical framework developed by Van den Brink & Ter Braak (1999). These authors proposed a constrained form of principal component analysis (PCA), called principal response curves (PRC) analysis to summarize the time-dependent effects of toxicants on communities. The analysis produces a graph showing the multivariate effect parameter (canonical coefficient) as a function of time, where the control treatment is represented by a straight line and effects are represented by negative values for the canonical coefficient. Another graph shows the loadings of the separate variables onto the canonical coefficient; from this graph it can be seen which variables contribute most to the canonical coefficient and in what direction (positive or negative). A PRC analysis was applied by Koolhaas et al. (2002) on the communities of mites and Collembola in TME experiments, aiming to establish the effects of the fungicide carbendazim. It appeared that a dose of 0.13 mg kg^{-1} could be considered as a no-effect level and that higher doses introduced changes in the community that did not recover within the duration of the experiments (16 weeks).

The bioinformatics approach to soil quality assessment includes the idea of indicators, as discussed above in the context of soil health. It acknowledges that the question about which indicators will be *a priori* more suitable than others is not very fruitful. Multivariate statistical analysis will show which indicators contribute most to the operating range of the system and which indicators are of little value. A multidimensional approach to environmental quality assessments was also advocated by Karr (1992), Moore & De Ruiter (1993), Attril & Depledge (1997) and Elliott (1997). Further work along this line is necessary to develop a data base that will allow the definition of soil-type specific normal operating ranges. This will hopefully reinforce the position of ecological arguments in soil quality assessments.

Finally, one may speculate on the possibility of using the tremendous information content of molecular data to characterize and assess soils. If it were feasible to develop DNA microarrays for defined types of soils, it would be possible to define the NOP of a soil not in terms of ten to hundred variables (each requiring significant analytical effort), but in terms of thousands of relatively easily measurable responses. Such a bioinformatics approach could revolutionize soil quality assessments.

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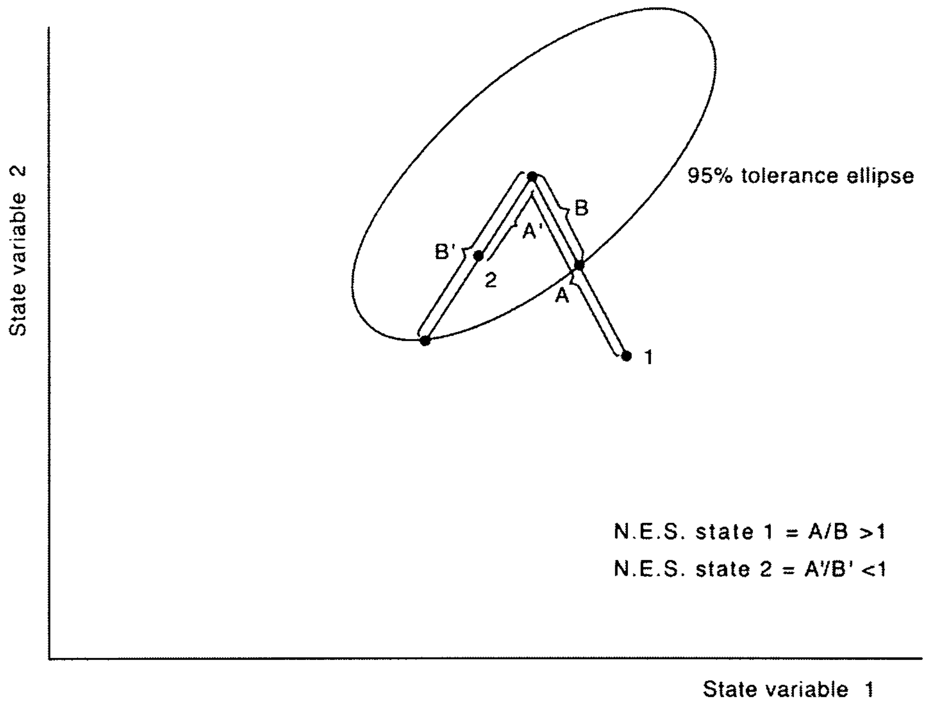

Figure 4. Illustrating the concepts of "normal operating range" and "normalized ecosystem strain", as introduced by Kersting (1984).

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