1 Soil quality evaluation

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Summary

As compared to the environmental compartments air and water, soil is an extremely complicated system. This is due to a huge variety of soil properties and chemical, physical and biological system conditions. The combination of these makes the development of general rules for quantitative evaluation of soil quality impossible. The large variety of values for the buffering capacity for different soils and different compounds also plays a role in this respect.

Two examples for derivation of effect-oriented soil quality standards are briefly described.

In The Netherlands, a system of reference values was derived, which refers to compound contents in soil reflecting situations of low degree of contamination. For heavy metals, these reference values consist of sliding scales in which the influence of clay content and organic matter content on the metal content is incorporated.

New developments with respect to biological indicators by means of nematodes are briefly discussed.

1.1 General consideration

Of the three environmental compartments air, water and soil, the last gives undoubtedly the most difficulties to assess its quality in a quantitative way. This is mainly due to the fact that soils show a huge variety in composition. This in turn influences the behaviour and effects of compounds in the system in chemical, physical and in biological sense as well.

Soils usually have a much larger buffering capacity than air and water. This property, as schematically indicated in Figure 1.1, can be described as the capacity to allow contents of compounds, once present at optimum level, to increase without actual occurrence of negative effects. Because a number of potentially hazardous compounds are also prerequisites for good soil functioning, these show a positive effect with increasing content at the low concentration level. The area between contents I and II in Figure 1.1 represents the buffering capacity. Actually this buffering capacity is different for all different compounds of interest, and for all different soil properties and varying system conditions occurring in practice. Moreover, these properties and conditions are usually main factors in controlling chemical speciation of a compound under consideration, which in turn plays a major role for its bioavailability, thus influencing its effect.
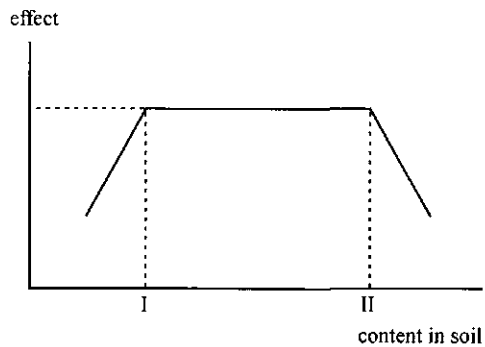
Figure 1.1. Schematic representation of the soil buffering capacity.

However, in order to develop rules and measures for soil protection, quantitative evaluation of soil quality is a prerequisite. During the C.E.C. Conference on 'Scientific basis for soil protection in the European Community' in 1987 (Barth & L'Hermite 1987) agreement was reached that such evaluation should preferably be based on effects that can be expected from the presence and behaviour of pollutants/contaminants in soil. This in turn requires a quantitative risk assessment approach to soil functioning.

Figure 1.2, taken from van Genderen (1987), presents in a schematic way how contaminants may adversely affect humans, plants and animals, including soil organisms. For soil quality evaluation, the following question must be answered: which quantity of a given compound in soil, C_s, is still acceptable in order to protect these three groups of organisms. This question is closely related to risk analysis of health and toxicological considerations. Once the exposure-effect (more strictly dose-effect) relationships are determined, the question is transformed into the assessment of the result of the exposure of the organism of interest.

Sometimes the exposure can be determined in a relatively simple way as in the case of some soil organisms. Usually the concentration (or rather the activity) in the soil solution is of major importance since the exposure dose is the product of the time of exposure and this solution concentration. This means that the relationship between solid phase content and the solution concentration is of prime concern.

Soil chemistry provides a basis to arrive at such relationships for different contaminants and various soil systems. Adsorption and desorption isotherms are the common form of expressing them, and the distribution coefficient, $K_{s,l}$, provides a useful parameter to characterize interactions between the solid and liquid phases of soil.

The concentration in the soil solution is also highly significant with respect to leaching and hence the composition of surface water and groundwater, and the uptake of compounds by plant roots. But these uptake mechanisms are generally fairly complicated (e.g. preferential or discriminating uptake) so that the content of the compound in plants, C_p, usually can not simply be derived from transpiration fluxes and solution concentration values.

Whereas the liquid phase concentration generally is the most important, there are a few examples where the exposure, or part of it, is directly governed by the content of the solid phase. These are where the exposure results from ingestion of contaminated soil or dust, as is the case with earthworms (Ma 1983), with grazing animals (Bremner

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1981), and with 'pica' and mouthing behaviour (direct soil uptake by children, Brune-kreef 1985). Normally, however, the exposure route is much more complicated, cf. Figure 1.2.
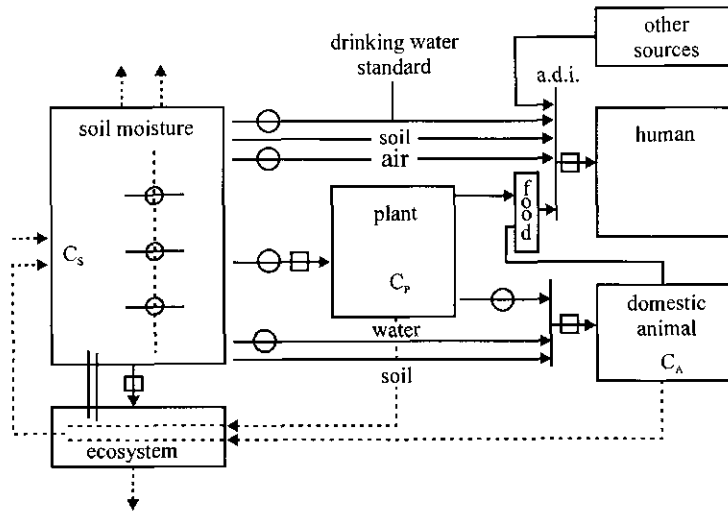

Figure 1.2. Pathways for contaminants from soil to other compartments and groups of organisms. From van Genderen 1987. For most compounds the distribution between solid phase and liquid phase (soil moisture) is of prime concern.
Circles refer to required information about transfer coefficients.
Squares to the same about dose-effect relationships.

The circles in the connecting lines of Figure 1.2 refer to the need for quantitative information about transfer factors of contaminants from the soil to other compartments and organisms indicated. The squares indicate a need for quantitative information about dose-effect relationships in order to estimate the (risk of) effects on humans, animals, plants and the soil ecosystem. The dotted lines back to the soil indicate that part of the contaminants is recycling within the system as for persistent constituents, e.g. heavy metals and slowly degradable organic compounds.

As was stated in a recent report to the C.E.C. on Soil Quality Assessment (de Haan et al. 1989):
ideally, a quantitative evaluation of soil quality requires complete information about all the transfer factors and dose-effect relationships shown in Figure 1.2:
- for all compounds that can cause malfunctioning or disfunctioning of soil;
- for all different soil types and soil properties that are found;
- and for all combinations of the different variables (e.g. pH, redox potential, accompanying compounds) that control compound behaviour in the soil system.

Because this is an impossible task at the present stage of knowledge, a number of limitations and restrictions has to be made in a first approach.

1.2 Soil quality in relation to soil functions

The quality of soil is adversely influenced by contamination (pollution) of the system. The concepts of 'contamination' and 'pollution' of soil are used here in a comparable way as they reflect only a difference in degree of damage to the soil system. Any addition to soil of contaminants, that is of those compounds that may exert adverse effects on soil functioning, can be defined as soil contamination. Because most soils do have a certain buffering capacity, as mentioned earlier, it usually takes some time before the negative effects become apparent. Once this situation occurs the soil can be considered as polluted, which for all practical purposes thus means that malfunctioning (or disfunctioning) of the soil is apparent due to an abundant presence or availability of compounds. Such malfunctioning may refer to one specific function in particular but sometimes also to a combination of different functions.

Some of the most important soil functions are:
- the bearing function, for instance as playground for children and for building of houses;
- the plant growth function; this may as well refer to natural vegetation as the production of crops for animal and human consumption; in relation to crop production not only quantity aspects like yield play a role, but also quality aspects are of concern as health of the consumers can be influenced by plant composition;
- the filtering function of water, groundwater as well as surface water;
- the ecological function of soil, with its contribution to element cycling as an important aspect.

Proper functioning of soil in the above different functions imposes a wide variety of quality criteria. This strongly hampers the introduction of a quality assessment methodology with general applicability and validity.

At first sight the bearing function would probably seem the least demanding, at least once certain physical requirements are met. However, experience of last decades in for instance Love Canal, USA, and Lekkerkerk, The Netherlands, and many other places has learned that also for building of houses some minimal requirements of chemical conditions in the soil are needed; and that a combination of use of the same site for waste disposal and later for urban expansion is not always without problems.

In the panic that arose following the discoveries of somewhat uncautious combinations of soil usages, a strong demand was felt for methods to assess health risks for people living in such polluted areas. And of course it was realized from the very beginning that this would require insight in the quantitative relationships between pollutant exposure of the organism under consideration on the one side and the resulting effects to be expected from this exposure at the other side. However, at the same time it was realized that there are many different pathways of pollutant exposure, of which in case of human beings the most important ones are:
- soil ingestion, especially for young children;
- inhalation of air, containing volatile polluting compounds;
- drinking of water and
- food in the form of plant products and animal products.

If the attention is confined for a moment to the last mentioned pathway it becomes apparent why the evaluation of soil quality in a quantitative way constitutes such a complicated problem. Assume that one can rely on the value of an acceptable daily

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intake (ADI) of a certain compound for human beings, and that there is agreement on the allotment of this ADI over different pathways of exposure; then the sequence soil-plant-animal-product-human as shown in Figure 1.2 causes the contribution to human exposure via animal products to be akin to a fourth-order derivative from the soil quality.

This complexity undoubtedly contributed to the fact that the growing general awareness of soil pollution problems caused indeed a certain boom in the development of so-called multimedia exposure models, which unfortunately so far did not surpass the level of qualitative approaches in most cases.

Under the stress of the circumstances following the discoveries of the supposedly severely hazardous situations on built-on waste disposal sites, measures were developed in order to remediate the spot by cleaning up activities or by isolation measures. In this connection in The Netherlands a Soil Cleanup Act was introduced. In order to present some guidelines for estimating the severeness of the situation, so-called A-B-C values were suggested. These were actually meant as signal values with the following meaning:

A-value: background; no problems expected;
B-value: indication of need for further investigation;
C-value: indication of severe pollution and need for action.

Although it was clearly stressed that these values were of a prelimininary nature only and should be used carefully, it turned out in practice that they tended to be maintained as rigid, soundly based criteria which supposedly allowed a quantitative risk evaluation of the areas involved. This single-value interpretation of soil quality has led to much commotion (sometimes resembling light degrees of hysteria) and in many cases to damages and economic losses, the necessity of which was at least questionable.

Newly built residential quarters were sometimes completely demolished on the basis of supposed severe health risk or hazards for the environment, even without any quantitative risk assessment.

Despite the problems involved in a scientifically based judgement of situations like above, it must be realized that waste disposal sites are still restricted areas of pollution only. Thus they can more or less be considered as point sources, where controle and protection against further environmental damage can be relatively easily performed. This is contrary to the large scale diffuse contamination of soil, which now is also wide-spread. Due to the extended areas involved the possibilities for controle and clean-up are then very limited. This forces to protection measures in order to prevent such diffuse contamination and pollution. Main sources for this type of soil degradation are invariably preceeding emissions of wastes into air and water, and direct addition of compounds to soil by men.

Sometimes different soil functions may come into conflict with each other. This may for instance be the case when soil is used for crop production, whereas at the same time the groundwater underneath this soil is required for drinking water purposes. So normal nitrogen fertilization of arable land on sandy soils (e.g. 200 kg N/ha per yr) results in nitrate leaching to groundwater, under conditions of a precipitation surplus of about 300 mm on annual basis, at a concentration of more than 120 mg NO_3 per liter, thus exceeding the E.C. standard with a factor of roughly 2.5. This clearly indicates that in such cases priorities have to be set with respect to soil functions (or soil uses). A priority given to a certain function, however, obviously will include limitations to other soil uses.

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When developing standards for soil quality in relation to soil functions, it might be reasoned that protection of the most susceptible function automatically safeguards all other functions. However, knowledge is insufficient at present even in order to differentiate between degrees of susceptibility. It therefore seems somewhat more realistic to consider a specific soil function as first priority in a certain case, and then try to develop soil quality standards (conditions, requirements) needed to allow this function. Despite this limitation the complexity of the system still leaves enough problems to be solved in soil quality assessment.

A few examples have been described in literature of effect oriented standards for soil quality. Lexmond (1980; 1981) derived a toxicity index with respect to plant growth for copper in soil. The availability of copper for uptake by plant roots appears to be controled by three main factors, namely the copper content of soil in gram per kg of soil (as measured after extraction with 0.43 molar HNO_3), the organic matter content of soil (expressed as gram organic carbon per kg of soil) and the pH (as measured in a suspension of calciumchloride solution). The two latter factors are of direct importance with respect to copper binding in soil: with increasing organic matter content and increasing pH, copper ions in soil solution are increasingly inactivated. The toxicity index is presented by line 1 in Figure 1.3, together with a number of field data. Values above this line represent the situation where significant damage to soil fertility occurs, resulting in yield depressions of 10% or more. Line 2 in Figure 1.3 refers to a Cu/C ratio of 2. At this value damage to soil fertility as the result of copper availability is unlikely for the pH range which is of interest for agriculture (4.5-7.5).
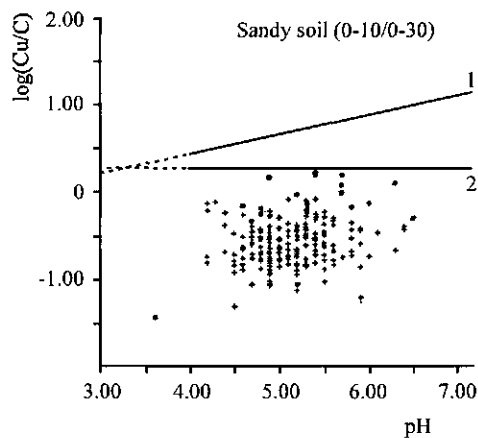

Figure 1.3. Copper toxicity index for maize, grown on sandy soil.

De Haan et al. (1987) derived acceptable Cd contents in soil with respect to leaching of cadmium to groundwater and surface water. For cadmium sorption in the practically relevant concentration range it is found that the Freundlich equation gives a satisfactory description. This equation is given by:

$$q = k.c^n \tag{1}$$

with q = amount adsorbed and c = concentration in solution.

In batch experiments the values k and n were assessed for a number of different soils. In spite of the considerable difficulties with respect to experimental accuracy a number of system parameters that were not sufficiently included in soil type differences, were varied.

These parameters were ionic strength (I_0), solid:solution ratio, electrolyte composition (including the competing heavy metals zinc, lead and copper), pH, etc. As an example of the large differences encountered for one specific soil at varied (Cl^-), I_0, and (Ca^{2+}), the experimental and fitted isotherms are given in Figure 1.4.
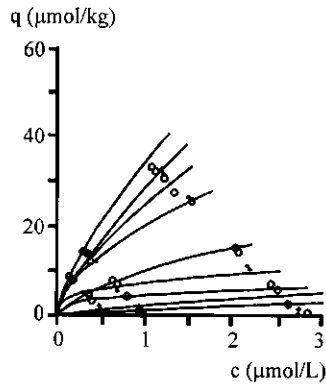

Figure 1.4. Cadmium adsorption (q) as a function of the concentration of cadmium in solution (C), for different values of ionic strength, chloride and calcium concentrations in solution.

The huge influence of these three system parameters on cadmium sorption is clearly demonstrated with the differences in sorption in this figure. This indicates the great care required to assess the adsorption parameters for a particular case as realistic values of the system parameters may differ between, for example agricultural soil water and waste percolation water. Taking into account the major factors controlling cadmium sorption an empirical relation was derived from equation (1):

$$q = k_1 \ (H^+)^a (Ca^{2+})^b c^n \qquad (2)$$

where a $\approx$ -0.48 and b $\approx$ -0.41 were fairly constant for a wide range of soil types. In order to take into account the effects of ionic strength (I_0) and cadmium chloride complex formation the concentration in equation (2) should be replaced by the Cd^{2+} activity (Chardon 1984). The value of k_1 varied significantly for different conditions even for one soil type. The following relation can be derived to relate sorption to the predominant sorbing phase (organic carbon, oc), and pH:

$$q = k^* \ oc \ (H^+)^a c^n \qquad (3)$$

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with oc as % (w/w) (100 x mass of organic carbon/mass of dry soil) and (H^+) in mol l^{-1}.

In equation (3), a $\approx$ -0.48, and oc varies from 0.8 to 4.0% (w/w). The value found for k^* is on an average 2.37 x 10^{-5} if q is given in mol kg^{-1} and c in mol m^{-3}. From equations (2) and (3) it is seen that as system parameters may vary in space, this is also the case for concentrations in equilibrium with a designated total cadmium content ρT on mass basis) given as:

$$\rho T = \rho q + \theta c \tag{4}$$

The definition of acceptable total contents in soil of contaminants should preferably be based on the relation between total content (T) and soil quality. Thus, with equation (4) in combination with equation (1) acceptable total cadmium content in soil may be calculated for different commonly accepted standards for cadmium concentration in ground water and surface water.

In illustration, the following standards are considered (in the units commonly used in this context):
- the Dutch standard for surface water to be used for preparation of drinking water: 1,5 μg l^{-1};
- the Dutch advisory value for surface water: 2.5 μg l^{-1};
- the EC standard for drinking water: 5,0 μg l^{-1};
- the FAO and WHO standards for drinking water: 10.0 μg l^{-1}.

In Table 1.1 the total contents in equilibrium with these concentrations are given for the different k values. A pronounced variation in acceptable total cadmium contents results. The contents given in Table 1.1 may be evaluated with reference to the common total cadmium content found for Dutch agricultural topsoils, which is T < 0.3 mg kg^{-1}. A soil with a total content exceeding 1 mg kg^{-1} is considered as heavily contaminated.

In both cases it took a number of years of intensive research work in order to collect the data for deriving these standards. It became clear, as should be expected, that the single value approach is useless in case of soil quality evaluation, but that instead sliding scales are required due to varying soil properties and system conditions.

Table 1.1. Acceptable cadmium contents of soil, in mg/kg, for different criteria for the cadmium concentration in soil solution, in μg/l, for varying values of k (μg^{1-n} l^n g^{-1}) and n = 0.8.

k	C			
	1.5 μg/1	2.5 μg/1	5 μg/1	10 μg/1
0.001	0.001	0.002	0.004	0.006
0.01	0.01	0.02	0.04	0.06
0.1	0.1	0.2	0.4	0.4
0.3	0.4	0.6	1.1	1.9
0.5	0.7	1.0	1.8	3.2
0.9	1.2	1.9	3.3	5.7
6	8.3	12.5	21.7	37.9

Because of the unacceptable long time required for standard development for the many compounds of interest, a new approach was followed, namely the development of reference values for soils that can be considered as 'non-polluted' or 'slightly contaminated'. These values have been indicated as 'new Dutch reference values'.

1.3 Reference values for unpolluted soil; heavy metal contents as an example

During the last decade considerable effort has been spent in The Netherlands in order to obtain insight in what levels of heavy metals could be related to good soil quality. Therefore an intensive analysis was made of areas designated as nature reserves, that have been managed as such for a long time (Edelman 1984). 'Good' soil quality should then be interpreted as non-heavily polluted, or almost unpolluted by human activities. It was, of course, realized that a certain degree of contamination by means of deposition from the air is unavoidable for most areas in Europe, and even for probably all areas of the world. This makes it impossible to obtain from present sampling an indication for natural background values, at least for some elements. Cadmium and lead are examples of heavy metals that have been wide-spread by aerial deposition as the result of human activities.

The natural background value depends on the nature of the parent material from which the soil has developed. For The Netherlands this parent material has been brought down from abroad by wind, water or through transport with ice. During transport and deposition a sorting took place according to particle sizes, which meant at the same time a sorting with respect to mineralogical composition. This is of utmost importance for the occurrence of heavy metals. Sand and silt consist of primary minerals which are resistent against weathering, whereas the clay fraction ($< 2 \mu m$) consists of secundary minerals formed after weathering of less resistent primary minerals. Quartz usually is the most important component of the sand and silt fraction, whereas clay minerals and hydrous oxides constitute the bulk of the clay fraction. The heavy metal content varies between the different minerals. In quartz this content is extremely low. The result is that the heavy metal content of sediments is related to their texture (grain size distribution). These values for natural background contents are fairly well known for different sediments. They may be considered as the natural background because the sediments have been formed and deposited at the time that human influence on heavy metal distribution was still absent. Of course, the original background values have been changed since that time because of soil forming processes and later as the result of human activities, especially when the industrial society became under development.

Therefore, the information gathered by soil sampling of nature reserves, as mentioned before, should preferably be characterized as 'current background values', indicating that they refer to the original values corrected for soil forming processes and air deposition. To give these values the indication 'reference values for good soil quality', as is at present usually done in The Netherlands for policy reasons, is (as a matter of principle) essentially not correct because such an indication would suggest information about desirable metal content and soil functioning. As stated before such information is still lacking. Moreover, it might well be that the contents are below optimum levels for specific soil functions, e.g. the crop production function in relation to required levels of those heavy metals which are trace elements for plant growth. Therefore it is preferable to characterize these contents as 'present background values for not abnormally contaminated soils'. Lexmond & Edelman (1987) developed a model which

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enables a quantitative description of the influence of clay content and organic matter content of the soil on the occurrence of heavy metals, under consideration of the absence and presence of human influence on the metal contents. These results are summarized in Figure 1.5. For detailed information about the symbols given in Figure 1.5, the reader is referred to the original publication in which the model was derived and discussed.
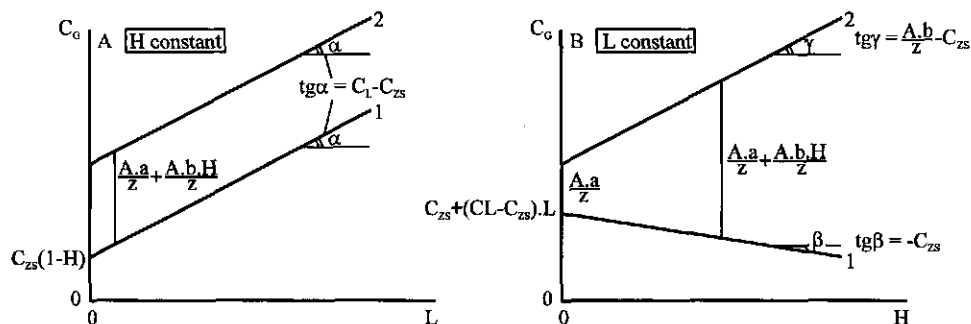

Figure 1.5. The relationship between metal content (C_G) and clay content (L) for constant value of humus content (H) (upper figure), and between metal content and humus content for constant value of clay content (lower part). Lines indicated with 1 and 2 refer to absence and presence, respectively of human influence on metal content. From Lexmond & Edelman 1987.

The symbols in Figure 1.5 have the following meaning:

C_G: total metal content in soil (mg.kg^{-1});

C_{zs}:   metal content of sand and silt fraction (mg.kg^{-1});

C_1:   metal content of clay fraction (mg.kg^{-1});

L: mass clay fraction (kg.kg^{-1});

H: mass humus fraction (kg.kg^{-1});

A: metal load through human activity (mg.m^{-2});

z: depth (m);

a,b: constants for the relationship between bulk density and humus content.

The value for soil bulk density is decreasing with increasing organic matter content. This means that the same metal deposition leads to a higher increase of content on weight basis for soils with increasing organic matter content.

This model has been used for a profound analysis of the data originally collected by Edelman (1984) in the topsoil (0-10 cm) of nature reserve areas (Lexmond & Edelman 1986; 1987). The results of this analysis are presented here for a few heavy metals in a summarizing manner by means of some figures (cf. Figure 1.6). The figures represent the relationship between the content of the metal indicated (vertical axis, mg.kg^{-1}) and the clay content of soil (horizontal axis, g.kg^{-1}).

Points refer to samples with an organic matter content, H, of < 0.25 kg.kg^{-1}, whereas open symbols refer to H values > 0.25 kg.kg^{-1}. Full-drawn lines represent the relationship meant, whereas the broken lines refer to the upper boundary of the range of element contents normally found in mineral top soils of nature reserve areas.
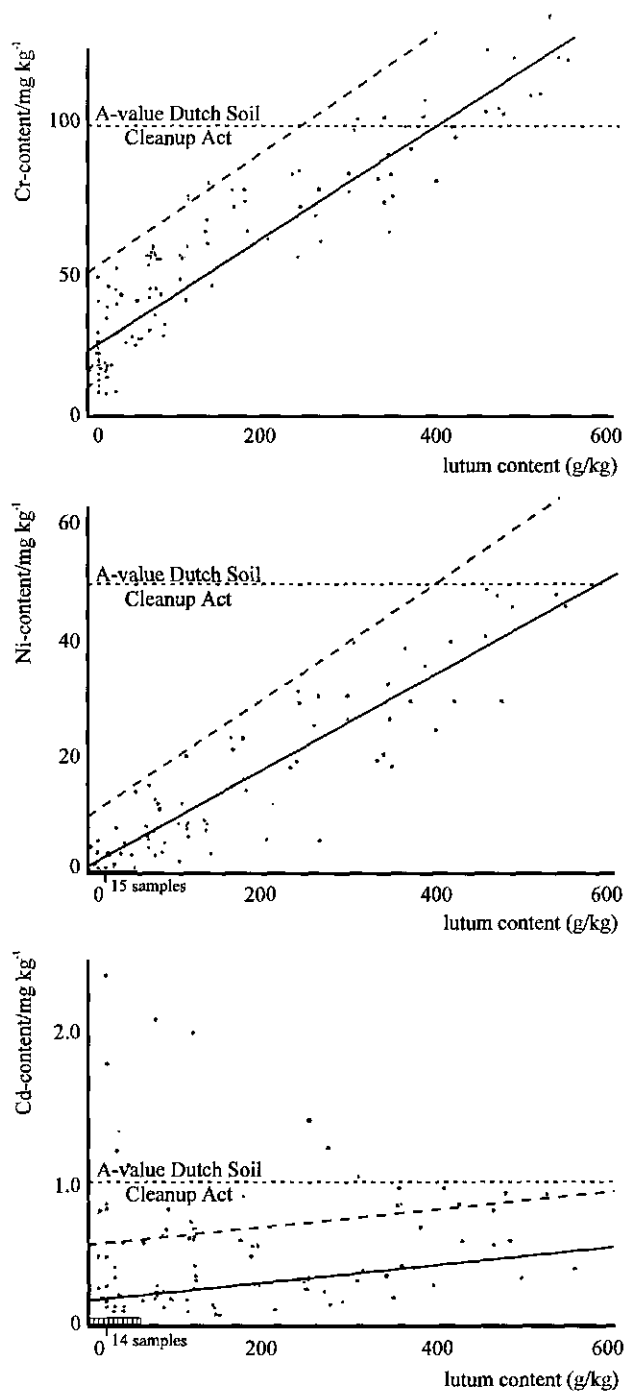
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Figure 1.6. Element content in nature reserve soils as related to clay content.

Examples are given for the heavy metals chromium (Cr), nickel (Ni) and cadmium (Cd). In these figures also the original A-values as mentioned in the Dutch Soil Clean-up Act are represented, indicating that these were, for all three elements presented, chosen at the high side.

The Institute of Soil Fertility at Groningen, The Netherlands, has made a comprehensive inventory of heavy metal contents in Dutch arable soils (van Driel & Smilde 1982; Wiersma 1985). These values for topsoil samples (0-20 cm), again as function of the clay content of the soil samples, can now be compared with the data for the nature reserve areas. This comparison is made in Figure 1.7, again for the metals Cr, Ni and Cd. The lines shown in Figure 1.7 refer to the earlier presented upper boundary of the range of element contents found in mineral soils of nature reserve areas.

It is thus shown that for Cr and Ni the situation in arable soils is in fair comparison with that in nature reserve soils. For Cd the arable soils show in a number of cases higher values which undoubtedly must be attributed to the use of phosphate fertilizers. A same comparison for the metals Cu, Hg and Zn would show many more situations where arable soils are higher in content than the nature reserves.

Table 1.2. Reference values for heavy metals, arsenic and fluoride.

compound	SOIL (mg.kg.$^{-1}$ dry soil) relationship	standard soil (H=10; L-25)	GROUNDWATER (μ g.l^{-1})
Cr	50 + 2L	100	1
Ni	10 + L	35	15
Cu	15 + 0.6 (L+H)	36	15
Zn	50 + 1.5 (2L+H)	140	150
As	15 + 0.4 (L+H)	29	10
Cd	0.4 + 0.007 (L+3H)	0.8	1.5
Hg	0.2 + 0.0017 (2L+H)	0.3	0.05
Pb	50 + L + H	85	15
F	175 + 13L	500	-

The new reference values for heavy metals, arsenic and fluoride in their relationship with organic matter content and clay content of soil is given in Table 1.2. The formulae for calculating the reference values as presented in Table 1.2 have been derived from the inventory data as mentioned before. As an example the values for a so-called standard soil with 25% clay and 10% humus are also presented.

It can not enough be emphasized that the above approach, although it may provide a valuable tool for a first rough evaluation of the degree of soil contamination, is by no means an effect-oriented soil quality assessment. Such assessment should not only consider merely compound presence but also compound behaviour, especially with respect to bioavailability. Although this subject will be given specific attention during this course, a few remarks may be made here.
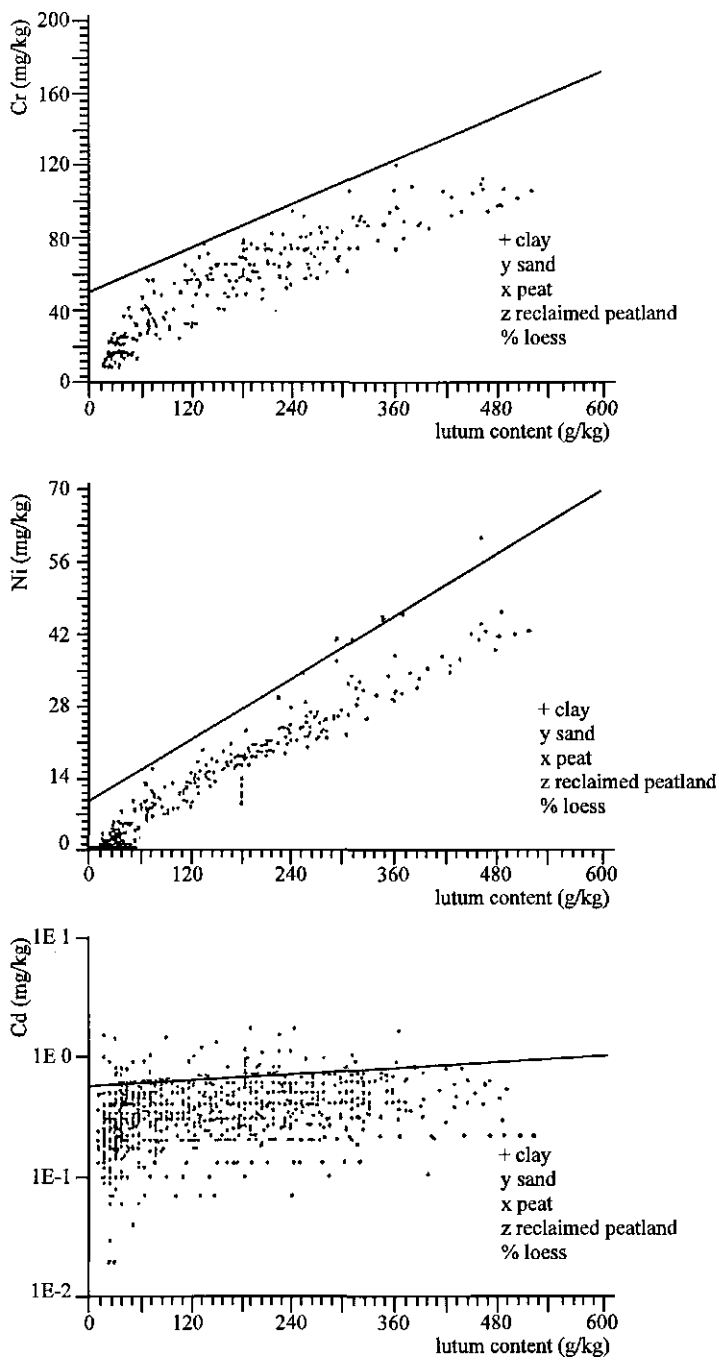
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Figure 1.7. Element content in topsoil of arable soils as related to clay content, lines refer to upper boundary in nature reserve soils.

1.4 Some remarks on bioavailability of compounds in soils

The concept of bioavailability is very important with respect to soil quality since it links in principle compound behaviour and the effect of these compounds on organisms. At present these links are not very well established and the word bioavailability sometimes is used more as a buzz word. A precise scientific definition is not simple to derive. Recently, soil chemical aspects of bioavailability have been discussed by van Riemsdijk (1990); some of these aspects are mentioned here.

Bioavailability relates to the availability of a chemical with respect to a specific organism. Bioavailability can only be measured in terms of an effect of a chemical on an organism. This effect is in general a complicated function of the concentration of the specific chemical as present in a certain complex matrix. This matrix may be for instance a certain soil or sediment. The effect that is being measured can be growth, uptake, respiration, degradation, root development, enzymatic activity, etc. When the chemical is of inorganic nature, it is suggested to define bioavailability in terms of the concentration of the inorganic element (Cu, N, P, Al, etc.) whereas for organic pollutants a specific chemical species (e.g. HCH) should be used. Bioavailability can thus be described as follows: Bioavailability of an element or organic species as present in a (complex) matrix is related to the possibility to cause an effect (judged positively or negatively) on a specific organism. The bioavailability of an element present in a specific form depends on the type of organism. For blue algae and leguminous plants $N_2(g)$ may be fully available whereas for most plants the bioavailability of nitrogen in the form of $N_2(g)$ is zero. Nitrogen in the form of nitrate may be fully bioavailable. The bioavailability of an element is thus dependent on the form in which it is present. In other words, different chemical species containing the same element may have a completely different bioavailability. The bioavailability is also dependent on the composition of the matrix in which it is embedded. These matrix effects can for instance be caused by competitive effects of other species present in different amounts in various soils, difference in pH, reactive surface etc. The formation of soluble complexes between a metal and organic ligands like humic and fulvic acids may either increase or decrease the bioavailability of this metal. An increase in bioavailability may occur although the metal-organic complex itself it non bioavailable. Enhancement of bioavailability may take place in case the transport of the metal to the organism is limiting the bioavailability. Complexation with dissolved ligands leads to an increased total metal concentration in solution facilitating the transport to the surface of the organism, where it may exert its effect after dissociation into the free metal ion. The organic-metal complex in this case is merely a carrier for transport. A decrease in bioavailability due to metal-organic complex formation may occur when the main effect of the complexation is a decrease in the (available) free metal concentration in solution.

The pH or proton concentration may also strongly affect the bioavailability of an element. The effect of the pH on the bioavailability may strongly depend on the composition of the matrix. In a culture solution, the availability or toxicity as measured by (reduced) growth increases with increasing pH, i.e. with decreasing proton concentration (Lexmond 1980). This observation may be explained by the decrease in competition between protons and metal ions at higher pH for surface sites present on the organism (root, cell wall) resulting in a higher affinity of the biosurface for copper leading to increased bioavailability. However, in soils copper toxicity or availability for plants generally leads to a decrease in bioavailability with increasing pH. In the plant-soil system, the copper partitions over the solution phase, the surface of the biota and

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the surface of soil colloids. With an increase in pH the preference of copper to bind to the surfaces (both living and dead) will increase because of decreased competition with protons. If the preference for the soil surfaces is increased more strongly than for the surface of the biota, a decrease in bioavailability will be expected as a result of an increase in pH.

Another interesting and complicating aspect of bioavailability is that the organisms may actively influence the bioavailability by changing the chemical composition of their immediate environment which may lead to an increased or decreased bioavailability. An example of a decrease in bioavailability is the excretion of organic ligands by earthworms. The excreted organic ligands bind for instance lead and cadmium rather effectively (Kiewit 1989) thus reducing the bioavailability as measured by the metal uptake of the earthworms from aqueous medium. The above given examples show that for an interpretation of the effect of environmental factors on the bioavailability a rather detailed knowledge is required on the chemical interaction processes. What is needed are generalizable concepts of bioavailability that can be applied to any soil, sediment or aqueous system in order to be able to interpret, predict and model the effect of environmental factors on the bioavailability. The examples given indicate that reaching this goal is a non trivial problem.

1.5 Recent developments in biological soil quality assessment

In addition to physical chemical methods as indication for soil quality, also the biological approach obtained increased attention during recent years. These different approaches may have their own advantages and limitations, but may also be complementary to each other. Ecotoxicological aspects of soil quality is given extensive attention in different parts of this course.

For biological soil quality indication, soil nematodes seem to be promising organisms. There is a large variety of nematodes, which have been very well described and classified. As described by Bongers (1988; 1990) nematodes are almost ideal organisms for laboratory experiments. They can easily be isolated from the soil, there are methods available for automatic counting of the organisms, whereas procedures for automatic identification are well under development now.

At the Department of Nematology of the Wageningen Agricultural University recently a new index has been developed, which can be applied for determination of disturbances within the soil nematodes communities. In cooperation with the British Museum (Natural History) and the University of Milan it is now under study whether this approach may be used Europe-wide for a biological soil qualification system and an ecological soil typology.

The principle of the approach may briefly be described as follows. The composition of the nematodes community is more or less a reflection of soil conditions, with respect to chemical aspects (e.g. pollution or contamination) as well as with respect to physical aspects (e.g. drought). This may also be indicated as a relationship between nematode community composition and stress conditions in soils.

In this respect the families of nematodes are classified in five different categories, of which the extremes are indicated as Colonists and Ascetics, respectively. The Colonists are considered as more or less opportunistic organisms, which are characterized by a short life cycle, high reproduction and other properties which are related to the ability of filling up socalled ecological vacancies induced by changing soil conditions. The Colonists are the least sensitive for stress conditions. However, after a certain time the

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Colonists are replaced by the Ascetics which are very efficient in feeding, have a large competition power and are very sensitive towards disturbances of the system.

The relative presence of Colonists in the nematode community is expressed in a so-called Maturity Index, which can be used as a measure for the degree of disturbance of the soil ecosystem functioning.

This new approach apparently has very promising perspectives and thus deserves further stimulation. Especially the combination of such biological indicators with quantitative description and interpretation of physical-chemical soil conditions may allow an important step forwards into the direction of soil quality assessment.

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