MINIMUM SOIL QUALITY

A USE-BASED APPROACH FROM AN ECOLOGICAL PERSPECTIVE

PART 1: METALS

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1 INTRODUCTION

1.1 GENERAL FRAMEWORK

For some time, a process of policy innovation has been under way in soil sanitation under the title BEVER (TCB, 1997). The Minister of Environment and the Lower House wished for an early assessment of the sanitation arrangements in the Soil Protection Act which came into effect in 1995. The formal aim of sanitation in these arrangements was that cases of soil pollution should be cleaned up on a multi-functional basis unless there are ‘location-specific circumstances’ for not doing so (Ouboter & Kooper, 1997). Recently, it was concluded that soil sanitation is too expensive and that the soil sanitation operation is not achieving enough since too few cases are being tackled over time. Furthermore, the extent of the soil pollution problem is leading, given the available resources, to stagnation in spatial and economic processes (VROM, 1997). The difficulties which have been noted have resulted in a change of course from a multi-functional to a more **functional approach** to soil sanitation, with the initial aim being to make the soil suitable for the intended use. However, a return to multi-functionality is not excluded if this is effective in the light of result considerations. Where soil pollution has, until the present, primarily been approached on a ‘problem identification’ basis, function-based sanitation hopes to provide a solution (TCB, 1997). However, the functional approach only applies to cases of soil pollution which date from before 1987 (VROM, 1997). With new cases, the pollution should be cleaned up immediately and in its entirety.

The central issue in the functional approach is what the soil can still be used for and under what circumstances. It is therefore concerned with the minimum soil quality requirements for the use in question (engineering requirements). The Technical Soil Protection Committee has indicated that this involves a shift in the point of view taken for soil characterisation. Thinking has to be based more on use requirements and the development of sensible and sustainable soil use, rather than simply on the avoidance or acceptance of toxicological risks (TCB, 1997). A strict human toxicological approach falls short of the mark in the assessment of soil suitability. The substance criteria related to function published by the Association of Netherlands Municipalities on the basis of human exposure risks (Moet, 1995) emphasise this point. With respect to certain contaminants (including metals), numerical values are stated, for example for the ‘public parks and gardens & recreational amenities’ use category, which virtually exclude the possibility of plant growth (personal communication Ernst 1997/Van Hesteren *et al.* 1998). Moet's (1995) numerical values are no guarantee for other categories either of continued unhindered use of the soil for an intended purpose. The Technical Soil Protection Committee (1993) issued an early warning about the limitations of differentiation in soil quality requirements based on human exposure routes. The Technical Soil Protection Committee believes (1997) that it is desirable to involve relevant ecological parameters at an early stage in soil characterisation related to land use since there is otherwise a risk of losing sight of aspects of soil quality which are not related to the exposure of humans to contaminants. However, the current approach to ecotoxicological risk characterisation is not suitable for declaring sites suitable for certain types of use (ibid.).

1.2 OBJECTIVE OF THE RESEARCH

The ecological research and consultancy bureau **WEB** *NATUURONTWIKKELING* was commissioned to conduct a survey by the Technical Soil Protection Committee into the **the minimum soil quality** for various types of soil use from an ecological point of view. Forms of soil use in **urban areas** were the main focus, but the information obtained certainly has relevance in terms of the spin off for rural areas.

The research first of all involves formulating requirements which have to be set on the strength the soil being used for a certain purpose on the basis of a description of the soil use functions; what organisms should logically be present and what are the minimum ecological functions for the soil use in question. This is followed by an elaboration of the qualitative ecological parameters for the minimum soil quality for the type of use. The identification, selection and functioning of use- specific species of special concern, key species and processes and the relation to soil quality serve to underpin these ecological parameters. The research consequently does not immediately result in numerical standards but offers a first step towards incorporating ecological information in the functional approach to soil sanitation. It goes perhaps without saying that the present research approaches the matter solely from an ecological angle. Human toxicological data have not been involved in the study.

1.3 DESCRIPTION AND RESEARCH METHOD

Soil use categories

The terminology of local development plans distinguishes eight forms of soil use, namely: (1) dwellings with garden, (2) dwellings with vegetable gardens, (3) dwellings without gardens, (4) traffic, (5) work, (6) social cultural, (7) recreation and (8) green amenities. Moet (1995) clusters these in four types of use: (1) dwellings with vegetable gardens, (2) dwellings with gardens, (3) dwellings without gardens, traffic, social/cultural, work and (4) recreation and green amenities. Faber (1997) also uses these categories for the purposes of land-use-specific soil characterisation. The present study latches on to this in the first instance. However, in the course of the research these categories were deviated from as a result of an overly great overlap between the types of soil use in practice (see § 2.3 for the arguments). This has resulted in the following soil use categories:

- gardens and allotments;
- verges and waste ground;
- public parks and gardens and recreational amenities.

Species of special concern

For pragmatic reasons a selection has been made from the wide range of contaminants which need to be examined for the purposes of land-use-specific soil characterisation. The functional approach to soil sanitation assumes a distinction between mobile and non-mobile pollutants. This does not, however, mean a simplified subdivision into mobile and non-mobile substances (see also § 3.1), but an assessment of the mobility of the contaminants on the basis of specific soil properties by site. Mobility therefore would seem to be unsuitable as a selection criterion for demarcating the substances/species for special concern for the present study. On the basis of the availability of literature data on phytotoxicity and soil biological processes, the ecological relevance, the commonness of the type of soil pollution and the relative immobility, the choice was for metals in general and arsenic, cadmium, chromium, copper, lead, nickel and zinc in particular. Arsenic is actually a metalloid but, to keep things simple, has been counted as a metal in this study.

Research method

Two types of information were used for the inventory: (1) literature data and (2) specific knowledge among experts. Various automated literature search systems were used in a specific search for recent information (articles, manuals and basic reports) relating to soil use, reuse of polluted sites, surface layers, metal toxicity and tolerance among plants, soil fauna and micro-organisms, biological availability, life support functions and redundancy (the degree to which different species can take over each other's function in soil biological processes). Experts were consulted (in the form of interviews and brief telephone conversations), because of the relatively new angle of land-use-specific soil quality and to support choices that have been made in the research process or fill gaps in knowledge in the research literature through expert judgement.

1.4 REPORT STRUCTURE

The next chapter presents qualitative ecological parameters for each soil use category. Chapter 3 then goes on to explore the question of how and whether a relationship can be made from these parameters with minimum soil quality (metals). Chapter 4 presents alternative angles of determining or achieving minimum soil quality. Chapter 5 lastly presents the conclusions of the study.

2 FORMULATING REQUIREMENTS BASED ON SOIL CATEGORIES

2.1 MULTIFUNCTIONALITY VERSUS FUNCTION-SPECIFIC REQUIREMENT

As already indicated in § 1.1, policy on soil sanitation recently abandoned the restoration of multifunctionality as preferred alternative. This means that when a site is being cleaned up its intended use is taken into consideration in the decision-making process and initially it only has to be made functional again for a specific type of soil use (Lokhorst, 1997). However, more extensive sanitation may be considered from the point of view of returns for the user or the environment. From an ecological perspective the minimum sanitation alternative related to the function could jeopardise the achievement of biodiversity objectives for certain types of soil use outside of protected areas, e.g. public parks and gardens, allotments and verges. This is diametrically opposed to current practice in various local authorities, where these forms of soil use frequently constitute part of green structures (urban ecological infrastructures) and/or are approached from a perspective of importance to nature (De Bruin *et al.,* 1995; Vissers *et al.,* 1995; Van Hesteren *et al.,* 1996; Van der Weijden and Schippers, 1996; Tenner *et al.,* 1997; see also § 4.3).

General Quality of Nature

Following a series of discussions in the magazine *Landschap* concerning the General Quality of Nature concept, Udo de Haes *et al.* (1997) establish a relationship between soil use types and the profusion of species. In their view there is a fundamental responsibility to contribute towards maintaining and developing the diversity of species in the Netherlands for each type of soil use. Dutch policy on **biodiversity** is elaborated in a Strategic Plan of Attack (SPA), in which *one* action item focuses on putting into practice **biodiversity** objectives outside the national network of important ecosystems for the purposes of environmental policy (in Schouten *et al.,* 1997). A study of this was carried out by Jansen (1996). With regard to the functional significance of the general quality of nature he draws a distinction between maintaining biodiversity for the sake of economic functions (e.g. earnings from recreation owing to biodiversity) and maintaining biodiversity for the sake of life support functions (see also § 2.2). In this context the concept of life support is narrowed down to 'support for human life', that is to say that value is attached to species that participate in processes presently known to be important to humans or that perform functions important to the survival of mankind (ibid.). When putting into practice the functional significance of the general quality of nature it is important to acknowledge that the ecological sciences are not yet in a position to name all the processes of importance to man, let alone all the species that may play an important part in them (ibid.). A pragmatic choice of species or taxa and processes which, based on current knowledge, constitute clear indicators therefore holds out the most prospects (Alkemade & Schouten, 1995). A similar approach is elaborated in this present study as well (see § 2.2 and § 3.3).

Eusynanthropic flora and fauna

With regard to a use-specific ecological characterisation of soil quality it is first necessary to indicate the extent to which ecological parameters must be met if soil use is to be brought to an acceptable level at least (Faber, 1997). Besides an anthropocentric angle of approach to the functional significance of the general quality of nature, as indicated above, it is possible, of course, to take into account the intrinsic significance of special qualities of nature with regard to engineering requirements for the various categories of soil use. To this end Faber (1997) adds a new element to the use-specific ecological characterisation of the soil, namely the importance of maintaining **eusynanthropic species** (flora and fauna that occur solely in the built environment) as an ecological parameter for four forms of use (see table 2.1).

Approach within the study

In this present study it was decided to elaborate the human-based angle first of all in chapters 2 and 3. This stems from the consideration that function-based sanitation - certainly where a minimum quality is concerned - is a policy choice which allocates less importance to general, non-use-related ecological parameters. Section 4.3 then looks at the question of whether an angle of minimum soil quality, in which justice is done to the supplementary natural functions of soil use categories, is desirable.

Table 2.1 Qualitative ecological parameters for different forms of soil use; the degree to which these should be met may vary for each form of use (after Faber, 1997).

Form of use	Ecological parameters
nature	all species, interactions and processes
agricultural business	most sensitive crop plants and livestock, self-remediating ability of the soil
recreation, green amenities	non-sensitive plant species, nutrients cycles, self-remediating ability of the soil, pets, eusynanthropic flora and fauna
dwellings with vegetable gardens, allotments	most sensitive crop plants, nutrients cycles, self-remediating ability of the soil, pets, eusynanthropic flora and fauna
dwellings with gardens	plant growth (ornamental plants), nutrients cycles, self-remediating ability of the soil, pets, eusynanthropic flora and fauna
dwellings without gardens, traffic, work, social / cultural	plants in green amenities and verge vegetation, self-remediating ability of the soil, eusynanthropic flora and fauna

2.2 REQUIREMENTS BASED ON PLANT GROWTH AND SOIL ECOLOGY: KEY PROCESSES, KEY SPECIES AND SPECIES OF SPECIAL CONCERN

A subdivision according to subecosystems

In order to arrive at function-specific engineering requirements and the associated minimum soil requirements it is first of all necessary to work out the ecological parameters for each category of soil use. Following the example of Lokhorst (1997) this was based on a - simplified - division into three subecosystems, namely: (1) general and microbial processes, (2) soil fauna and (3) flora. The first two subecosystems are closely related to the sustainable functioning of the soil. This study involves selecting key species and processes within these subecosystems which are relevant to the various categories of soil use, focusing on the concept of the **life support system**. Although this concept has much in common with striving for **multifunctionality** of the soil (Schouten *et al.,* 1997), it is obvious that certain soil processes are required in other soils than just multifunctional soils. Maintaining a certain degree of **soil fertility** (degradation / availability of nutrients) applies, for example, as a minimum functional requirement for all categories of soil use distinguished in § 1.3 (see *inter alia* § 3.3). This general requirement largely corresponds with the **nutrients cycles** parameter of table 2.1, as applied by Faber (1997). The relationship between the type of soil use and the subsystem of flora (or desired crop) is more direct: the very presence of certain plants is already a fleshing out of the soil use category. One example is the presence of trees in public green zones.

Key processes: general and microbial processes

With regard to the content of the life support system, maintaining essential processes in the soil (life support functions), such as the degradation of organic matter and the recycling of nutrients is crucial. In most cases these processes are the result of the activities of soil organisms (Schouten *et al.*, 1997). Sections 2.3. to 2.5. describe the broad requirements for maintaining the life support functions per soil use category. In chapter 3, these requirements are fleshed out in more detail. Although accepting a certain degree of metal contamination results by definition in a soil that cannot function entirely independently and/or optimally (personal communication Doelman, 1997), it is possible to characterise a number of subprocesses that are also crucial to the various soil use categories (**key**

processes). Based on a number of selection criteria (among which differences in sensitivity of general and microbial processes to metals), a key process has been selected in § 3.3. The relationship of this process to specific metal toxicity data is likewise discussed in the relevant section.

Key species: soil fauna

A number of species or taxa are known to perform important subprocesses for life support functions. By identifying **key species** and taxa of soil fauna it is possible to substantiate specific requirements for the minimum soil quality. This is discussed in § 3.3. Of importance here is not only what specific function a given species or taxon performs, but also the degree to which **functional redundancy** can be expected. It is assumed that in general many species can perform the same function in soil ecosystems, in other words there is redundancy in certain subprocesses of life support functions, so that if one or a number of species disappear the subprocess will not stagnate. For some subprocesses in the process of degradation this is obvious (for example, soil fauna feeding on fungi), whereas for a function like nitrogen fixation by *Rhizobium* in root nodules the disappearance of *one* species alone can result in a loss of function (Giller *et al.*, 1997). Finally, criteria such as information availability (on the relationship with metal contamination) and usability relating to differentiation according to soil use category play a part in the choice of key species and/or taxa within the soil fauna.

Species of special concern: flora

The possibility for plant growth is in general an engineering requirement that naturally applies to all three different categories of soil use. However, it is possible to differentiate within a general requirement like plant growth via types of plant species. In that case, however, it is not a question of species or taxa that are indispensable from the point of view of the functioning of the soil as an ecosystem, but of **species of special concern** whose presence should within reason be possible in order to vindicate the functioning of a given category of soil use. An example is lettuce *(Lactuca sativa)* in an allotment. In some cases the distinction between key species and species of special concern is vague. In the case of white clover (*Trifolium repens*) in meadows and verges, for example, the simple presence of this species does not say enough. The point is whether the symbiotic interaction of white clover and *Rhizobium* can function or not. In sections 2.3. to 2.5. species or taxa of special concern have been selected per category of soil use based on the current – accepted **-** interpretation of the type of soil use.

2.3 CATEGORY OF USE A: GARDENS AND ALLOTMENTS

Defining soil use category

For a number of reasons it was decided in this study to opt to further elaborate the combined category of soil use **gardens and allotments**. During the study it became clear that the distinction drawn between the categories of soil use ‘dwellings with gardens’ and ‘dwellings with vegetable gardens / allotments’ is only relevant from the human toxicological point of view. From an ecological or ecotoxicological angle it is plant growth in particular that is of importance. In practice there are no differences between the two categories from the point of view of the subecosystem flora (plant species found). Doelman (personal communication, 1997) states that there are no arguments from the soil fertility angle (availability of nutrients for plant growth) either for drawing a distinction between the categories.

For assessing the exposure risk to humans in the category ‘dwellings with gardens’ the Association of Municipalities in the Netherlands (VNG) assumed that 10% of home-grown vegetables were consumed, for the category ‘dwellings with vegetable gardens’ that 50% of potatoes and 100% of vegetables were consumed (Moet, 1995). The different percentages are irrelevant to the present study, but the fact that the above already suggests that in practice different types of **crop plants** are grown in gardens and in vegetable gardens and allotments is.

Faber (1997) does see possibilities for differentiating between the two soil use categories (see table 2.1.) on the basis of ecological parameters. For vegetable gardens and allotments he opts as parameter for a soil that imposes no restrictions on the most sensitive crop plants. This means that in principle it should be possible to grow any crop on the soil. For gardens Faber (ibid.) opts for the general criterion of plant growth, adding **ornamental plants** instead of the most sensitive crop plants. A distinction of this kind does not tally with the practice, in which three types of allotment are found: vegetable gardens, ornamental gardens and combined gardens (Vissers *et al.*, 1995). The two extremes (ornamental gardens and vegetable gardens) are less usual than combined gardens, in which both types of plants (crop and ornamental) are grown (De Zoeten *et al.*, 1998).

Ecological parameters

The soil use category 'gardens and allotments' requires it to be possible for a broad range of plant species and structure types to be found. In the case of ornamental plants this includes woody species such as the butterfly bush (*Buddleia davidii*), tulip tree (*Magnolia x soulangeana*) and the conifer arbor vitae (*Thuja occidentalis*), tuberous and bulbous plants like the yellow crocus (*Crocus flavus*) and daffodil (*Narcissus pseudonarcissus ssp. major*) and herbaceous species. Of this latter group alone the seed list of supplier Cruydt-hoeck (1993) describes 897 species for gardens. Besides this, lawns are fairly general in gardens and allotments. By far the most important (often even the only desired) component of these are **species of grass** (Neuteboom, 1989). In addition to ornamental plants and species of grass for lawns, **native flora** are also of importance to the soil use category, for example in the case of private botanical gardens. In these gardens the emphasis is placed on pursuing a wide floristic diversity in native species through fairly intensive maintenance and the creation of abiotic variation (Vissers *et al.*, 1995).

Finally, the soil use category 'gardens and vegetable gardens' requires it to be possible to grow a wide variety of crop plants: virtually all species of vegetable, potato and many varieties of native fruit (e.g. strawberry, apple and pear). Fast-growing leafy vegetables like lettuce, endive and spinach are frequently found in vegetable gardens (De Zouten *et al.*, 1988). From the human exposure routes it is known that these plants accumulate manifestly more cadmium than other plants (Wegener Sleeswijk and Kleijn, 1993). This study, however, concerns phytotoxicity. In view of the fact that both ornamental plants, crop plants and grasses as well as native flora constitute a functional part of the soil use category, a soil quality in which there is no nascent phytotoxicity would appear to be a minimum requirement. For this reason no specific species of special concern have been distinguished for this category.

According to the Technical Soil Protection Committee (1993) a use-specific cleanup of agricultural land or dwellings with gardens or vegetable gardens approximates very closely to a complete restoration of multifunctionality. In our view, however, there is a fundamental difference between the 'accepted agriculture' form of use and the soil use category 'gardens and allotments'. In agriculture, dependence on biological processes in the soil is circumvented by the use of fertiliser (Schouten *et al.*, 1997). Although it is obvious that fertiliser is used in certain cases in gardens and allotments, it cannot in practice be held to be a necessary measure. In the first place the use of fertiliser is not permitted in certain allotment complexes (De Zouten *et al.*, 1988). Secondly, users of gardens and allotments cannot be deprived of the possibility of opting for an 'ecological garden' (personal communication Doelman, 1997). The consequence is that in addition to far-reaching ecological parameters based on plant growth, life support functions such as the **degradation of organic matter, recycling of nutrients and the availability of nutrients for plants** must be guaranteed to a sufficient degree. In addition, an important role is reserved for **mycorrhiza** and **symbiotic nitrogen fixation** for non-stimulated systems (personal communication Doelman, 1997). It is therefore necessary to formulate specific requirements based on a number of relevant soil biological parameters (see § 3.3 and 3.4).

2.4 CATEGORY OF USE B: VERGES AND WASTE GROUND

Defining soil use category

The soil use category ‘verges and waste ground’ corresponds in principle to the form of use ‘dwellings without gardens, traffic, work, social / cultural’ as distinguished in Moet (1995) and used by Faber (1997) as well. **Verges** are the part of infrastructural facilities covered with vegetation (Niemeijer & Verburg, 1995a). Although verges can also constitute part of wet infrastructural facilities (hydraulic works), e.g. banks and dikes, this study confines itself to the functions of verges with respect to road construction and traffic. They include both side verges and central reservations. **Waste ground** is in fact a verge as well, but then in a broader sense of the word: waste ground is - overgrown – remnants of land in the urban environment or on industrial estates. The ecological function of verges and waste ground is discussed in § 4.3.

Ecological parameters

A verge serves in the first place as a porous retainment of the actual road. In connection with traffic safety a verge also needs to have sufficient load-bearing capacity. This can be improved by making the **top soil sufficiently nutrient-poor**, so that fewer tyre tracks are created (Van der Weijden & Schippers, 1996; Niemeijer & Verburg, 1995a). Consequently, the fertilisation of verges is not normally desirable. The vegetation in the verge is not intended for production. Besides, more than enough nitrogen fertilisation occurs through air pollution and the leaching in and blowing over of fertilisers from agricultural activities (Niemeijer & Verburg, 1995b). Nutrient-poor verges harbour fewer **worms**; as a result they are not as loose as nutrient-rich verges and present less danger of skidding. Owing to these points nutrient-poor verges are deemed to be more functional than nutrient-rich, verges (Van der Weijden & Schippers, 1996). This angle of approach holds out prospects for the aspect of soil fertility, whereby the requirements for verges in regard to the **availability of nutrients for plants** is clearly lower than those for gardens and allotments.

As already stated, a verge is as a rule covered with vegetation. The disadvantage of paved verges is that they cannot accommodate cables and pipelines that must remain accessible (Niemeijer & Verburg, 1995a). Faber (1997, see table 2.1) provides two ecological parameters from the perspective of plant growth: **green zone plants and verge vegetation**. These parameters exhibit much overlap if one looks at the structural diversity of the concept of verge vegetation. This can be divided into three types: (1) grassy vegetations (grasses and herbs), (2) dwarf shrubs (heather) and (3) trees. By stating minimum requirements and species of special concern relating to this structural diversity, the parameters for waste ground can likewise be expected to be guaranteed.

With regard to the prevention of erosion an important goal is to obtain sods that are strong and compact enough. An accepted method of creating a grassy, erosion-resistant vegetation is to sow the verge with **grasses**. It is not customary to sow herbs because of the possibility of flora adulteration (Niemeijer & Verburg, 1995a). On the basis of ecological departure points, applying verge mowings from the area is one possibility of creating herbaceous vegetations fast. Various species of grass are suitable for sowing. It is therefore possible for the verge maintenance authority to choose its own species and variety. However, there is also a standard mix for verges. This is the B3 seed mixture contained in the list of varieties for agricultural crops (Ebskamp & Bonthuis, 1997). In our view the use of this mixture is a minimum engineering requirement for grassy verge vegetations. Based on the composition of the standard mixture, the species ***Festuca rubra, Festuca ovina*** and ***Agrostis capillaris*** have been selected as **species of special concern** for the soil use category ‘verges and waste ground’. An additional ecological parameter is that the symbiotic nitrogen fixation of white clover (***Trifolium repens***) be able to function. This is needed for starter fertilisation in order to obtain erosion-resistant sods quickly (Melman & Verkaar, 1990; Niemeijer & Verburg, 1995a). Without additional green manuring this need not necessarily be at the expense of the wish expressed earlier for a nutrient-poor top layer.

On relatively acid soils heather is sometimes also sown, as heather varieties grow on low-nutrient soils. Depending on the moisture content of the soil the variety may be cross-leaved heath (***Erica***

tetralix; wet to moist soil) or heather (*Calluna vulgaris*; dry to moist soil). Both count as **species of special concern** for the soil use category 'verges and waste ground'.

With regard to the third type of structure of verge vegetation, trees, the following. The desire to achieve a richly varied verge planting within a short space of time resulted in the seventies in plantings in mixtures of up to 10 varieties (in Van der Sluijs & Melman, 1990). Experience has shown, however, that these plantings are not very successful due to the differences in growth among the varieties. At the moment the aim is to include only a few main types of wood in the range (ibid.). The possibility for **a number of (relatively non-sensitive) tree species** to grow can in our view count as a minimum engineering requirement for the soil use category 'verges and waste ground'.

2.5 CATEGORY OF USE C: PUBLIC PARKS AND GARDENS & RECREATIONAL AREAS

Defining soil use category

The soil use category 'public parks and gardens & recreational areas' comprises on the one hand parks and gardens and on the other sports fields, playing fields and recreational areas (camp sites).

Ecological parameters

The soil use category 'public parks and gardens & recreational areas' encompasses a large number of manifestations of planting (Boer and Schils, 1993). The minimum parameter for public parks and gardens involves in any event the possibility for woody (trees and shrubs) and herbaceous covers (herbs and grasses) to grow. Although ornamental plants are also used in public parks and gardens, e.g. cultivated roses or bulbs, it is debatable whether the possibility of such plantings should be available in every green amenity.

Compared with the category of use 'verges and waste ground' the engineering requirement for public parks and gardens & recreational areas is less informal as regards species of tree. It is inconceivable for a site to be zoned as a green amenity if no generally used species of plant can grow on the soil in question. Ten general species and genus of tree in green amenities have been selected as **species of special concern** for the soil use category, namely (based *inter alia* on Van Heusden *et al.*, 1994; Grimberg, 1994; Boer & Schils, 1993):

- common oak (***Quercus robur***);
- sycamore (***Acer pseudoplatanus***);
- white horse chestnut (***Aesculus hippocastanum***);
- poplar / white poplar (***Populus spec.***);
- silver birch (***Betula pendula***)
- willow (***Salix spec.***);
- lime (***Tilia spec.***);
- ash (***Fraxinus excelsior***);
- European mountain ash (***Sorbus aucuparia***);
- elm (***Ulmus spec.***).

As regards shrubs, hazel (***Corylus avellana***), hawthorn (***Crataegus spec.***), elder (***Sambucus nigra***), alder buckthorn (***Frangula alnus***) and bird cherry (***Prunus padus***) have been selected as **species of special concern.**

For sports fields and playing fields it is very important that the grass vegetation form a sturdy, compact and flat sod which can also endure being trampled on in winter. Some varieties of grass are more suitable for this than others. On the basis of selection criteria such as sod formation, endurance of trampling and hardiness, perennial rye-grass (***Lolium perenne***) and smooth meadow-grass (***Poa pratensis***) are the species most used on sports fields (Ebskamp & Bonthuis, 1997; Te Velde *et al.*, 1989). From the point of view of recreational sites there is an added requirement that it be possible to cover a stretch of grass for a longer period of time (for example, with a tent). Stoloniferous grasses in particular can survive this treatment, for which reason red fescue (***Festuca Rubra***) is often used in addition to smooth meadow-grass (Te Velde *et al.*, 1989). The above species have been selected as **species of special concern** for the soil use category ' public parks and gardens & recreational areas'.

The public parks and gardens form of use makes higher demands on the fertility of the soil (i.e. the degree to which biological processes in the soil are undisturbed) than do verges and waste ground, but lower demands than gardens and allotments. Although the supply of nutrients has to be guaranteed for a wide range of plant species, endeavouring to achieve high crop production is not a relevant requirement for public parks and gardens. In the case of recreational sites and sports fields balanced fertilising is often applied (Kappen, 1989). Although the activity of **earthworms** has a positive impact on the porosity and homogeneity of sports fields, it can also become too excessive, however, with the result that a 'richer' subsoil produces a top layer that soon becomes nutrient-rich and not compact enough. Sometimes measures such as acid fertilisation are even applied in order to reduce the numbers of earthworms (ibid.). A highly active worm population therefore does not count as an ecological parameter for this soil use category (see § 3.3 and 3.6).

3 TOWARDS A MINIMUM SOIL QUALITY PER SOIL USE CATEGORY

3.1 BIOAVAILABILITY OF METALS

Chapter 2 looked at the ecological parameters for guaranteeing an unrestricted fleshing out of the soil use categories as seen from the human angle. This chapter discusses the relationship between these parameters and the prevention of metals in the soil. An initial aspect is dealt with in this section: the bioavailability of metals in relation to mobility.

Establishing bioavailability

The toxicity of substances to organisms depends to a great extent on the availability of that substance and the exposure route (Tenner *et al*., 1997). In regard to soil examination Professor Mulder (Chemistry, Utrecht) already argued in the mid-nineteenth century ‘ that you cannot deduce from presence what the situation is regarding availability’. According to Lexmond (personal communication, 1997) a use-specific approach to soil quality starts with the abandonment of the total content of heavy metals in the soil as a characterisation measure. Total contents are often immediately associated with the effect, whereas only a given part of the total content causes this effect. Lexmond (ibid.) therefore proposes using the concentration of metal that is established when earth is suspended in a dilute salt solution: **0.01 M $CaCl_2$**, something that is already being done in his department. Van der Guchte (1996) endorses the distinct possibilities of this extraction for metals in view of the fact that it could to a large extent cover the problem of bioavailability. “Mild” extraction approximates the fraction of the metals dissolved in the pore water or soil moisture (Pepels and Lagas, 1993). The value thus determined forms the departure point for the theory of equipartition, which assumes that it is quite possible to predict the internal exposure for soil organisms from the concentrations in the water phase (Lokhorst, 1997). This has been established for certain soil organisms (ibid.), but it is questionable whether this applies generally (for example, to mycorrhized plants or macrofauna species, which can withdraw partially from direct contact with the water phase). As no substantiated and validated limit values are available for the bioavailable contents, it is impossible to make any pronouncement as yet on actual and potential ecological risks based on t he extraction. A comparison with reference samples could perhaps provide a solution. Lexmond (personal communication, 1997) suggests a value of 2 mg/l for zinc. Above this level the more sensitive plants would have problems. All this has been tested in pot trials with French beans and field observations of growth inhibition in broad beans, horse beans and spinach (ibid.).

Van Wensem *et al*. (1994) suggests that when characterising the risk of metal pollution more use should be made of the **ITC** or **internal threshold (body) concentration** (obtained through field studies) and the **concentration factor** (CF) of soil organisms. These aspects are discussed in the next section from the point of view of phytotoxicity.

Bioavailability and soil factors

Lexmond (personal communication, 1997) points to the complexity of the term biological or **bioavailability,** which has meanwhile acquired several meanings. Apart from the type of organism, the bioavailability of heavy metals depends on a range of soil factors, of which Bockting & Van den Berg (1992) give a fairly complete list. The chief soil factors that have an impact on the bioavailability of metals are the **organic matter content**, the **clay content** and the **pH** (personal communication Doelman, 1997; personal communication Lexmond, 1997; Ronday, 1996). With regard to the bioavailability of metals, the **depth** of the pollution is naturally of importance on account of its obvious relationship to exposure. It should be noted that pollutants that have been carried deeper can in time have impacts closer to the surface (Tenner *et al*., 1997) or vice versa (personal communication Faber, 1997). Biological factors such as **bioturbation** and **root depth** play a part in this.

Heavy metals in general adhere firmly to organic matter (even though this differs not only per **element** but also per type of **organic matter** (personal communication Doelman, 1997)) and clay particles in the soil, and are hence less easily available for soil organisms (Tenner *et al*., 1997). Organic matter appears in practice to be able to play different parts where the mobility and immobility of heavy metals is concerned. Whenever organic matter is applied to fix mobile metal, as is done in practice (personal communication Lexmond, 1997; personal communication Doelman, 1997), it is necessary to take into account the mineralisation which will in time make the metal available again. It should be noted here that a high heavy metal content has an impact on the natural processes in a soil. This means, for example, that bioturbation is curtailed and hence the **mobilisation** or **remobilisation** of heavy metals (personal communication Doelman, 1997).

In general, when the pH is lower more metals dissolve (personal communication Lexmond, 1997; Van Straalen and Bergema, 1995; Cairney, 1996). However, various comments can be made on the general trends. In the first place there is the factor of time: experiments have shown that over time heavy metals become more strongly bound to the soil (personal communication Lexmond, 1997). This is linked to the fact that the water solubility of heavy metals in soils contaminated through the actions of man is a factor of 10 higher than in naturally enriched soils (Van Straalen and Verkleij, 1991). Added to this, Bockting & Van den Berg (1992) find that the nature of the pollution (for example, the difference between mining waste and sewage sludge that has been spread) is a very important factor with regard to the mobility of metals and hence their harmfulness to organisms. Doelman (personal communication, 1997) further points to the role of the **heterogeneity** of the soil, which is not taken into account in policy based on standard systems. The level of acidity, for example, can be much lower at the microlevel than the universally accepted pH. This means that the ecosystem can mobilise a great deal on a microscale whereas, overall, immobility will dominate. This also indicates that the distinction drawn in policy between mobile and immobile pollutants is not very clear.

Mobility of the elements

Arsenic is a metalloid and in the environment often acts differently to the heavy metals. Its availability is mainly linked to the quantity of active iron (personal communication Lexmond, 1997). The adsorption of arsenic to organic matter virtually never occurs, but its bonding with particles of clay is important, though (Tenner *et al*., 1997). The pH plays an important part in this (Bockting & Van den Berg, 1992). In the top layer arsenic is generally strongly bound to the soil, not very mobile and relatively speaking not readily available to organisms (Bockting & Van den Berg, 1992; Tenner *et al*., 1997). Of the heavy metals it can be said that **cadmium, nickel** and **zinc** are fairly mobile, contrary to **chromium, copper** and **lead** (Bockting & Van den Berg, 1992). As already stated, the speciation and availability of these metals depends to a substantial degree on the pH of the soil (Tenner *et al*., 1997). For some metals the pH also determines the relative importance of adsorption to organic matter compared with adsorption to clay particles (ibid.).

Soil type correction

The relationship between bioavailability and soil type can have a strong influence on the toxicity of a substance found in experiments (Van Straalen and Bergema, 1995; Faber, 1995; Tenner *et al*., 1997). For purposes of soil protection policy toxicity data are therefore converted to a standard soil (soil type correction). In standardising, however, a correction is only made for organic matter (converted to 10%) and clay (converted to 25%) and not for pH. The above shows that this can be seen as a deficiency. Doelman (personal communication, 1997) and Lexmond (personal communication, 1997) also emphasise this. An attempt to integrate the pH into models was undertaken by Van Straalen and Bergema (1995). Among other things, they found that there was very little experimental data in the literature on lead and cadmium in relation to pH and the data that was available was virtually unusable. In addition, they concluded that for lead in particular there was sufficient cause for concern regarding toxicity if the pH dropped (ibid.). A more successful attempt to integrate the pH was made by Boekhold (personal communication J. van Wensem, 1997).

Metal contents in plants

The transfer of metal from soil to plant depends primarily, of course, on the properties of the element in question - apart from all kinds of time- and place-related factors. This is not only the result of the specific behaviour of an element in the soil, but also of the absorption and transport mechanisms of the plant, which are more or less element-specific (Bockting & Van den Berg, 1992). In the second place, the transfer depends heavily on the **species of plant**. This can be traced back in part to plant-specific interactions between plant and soil and in part to morphological and biochemical properties of the species of plant in question (ibid.). Section 3.2 looks at these aspects in more detail.

Mycorrhizae

Another aspect that has an effect on the bioavailability of heavy metals to plants is their symbiosis (if any) with **mycorrhizae.** The role played by mycorrhizae is not clear and, as a result, is complex (personal communication Doelman, 1997). It appears to be heavily dependent on the type of heavy metal, the species of mycorrhiza and plant, and various soil properties. A study carried out by Diaz *et al*., (1996) shows for two mycorrhized plant species that more Zn and Pb are absorbed at low concentrations of these metals and that absorption is inhibited at higher concentrations. In addition to having a protective effect, mycorrhizae can also encourage the absorption of heavy metals by plant roots (Guo *et al*.; Wuertz and Mergeay, 1997). On the one hand, as well as increasing the absorption of nutrients, the symbiosis also increases the absorption of essential metals such as zinc and manganese when they are available in limited concentrations. However, the moment heavy metals are available in high concentrations, mycorrhizae appear in many cases to protect the plant from excessively high internal concentrations (Turner, 1974; Verkleij, 1994; Galli and Schüepp, 1996; Wuertz and Mergeay, 1997). Sections 3.2 and 3.3 look at mycorrhizal fungi in more detail.

3.2 POSSIBILITIES FOR DIFFERENTIATION FROM THE PERSPECTIVE OF PHYTOTOXICOLOGY

If the subecosystem flora were less sensitive to soil pollution than soil fauna or general and microbial processes, there would be no need to devote attention to ecological parameters from the perspective of plant growth as formulated in chapter 2. This is not the case, though. On the basis of an analysis of the **relative sensitivity** of the three subecosystems mentioned to the heavy metals group of substances, Lokhorst (1997) finds that the subecosystem flora is the most sensitive. Although relative sensitivity may differ per metal, the importance of focusing attention on the vegetation has thus been demonstrated. The question now is what information is available and to what extent it can be used in land-use-specific soil characterisation.

From the preceding section it can be deduced that establishing a direct relationship between the total content of a given metal in the soil and the occurrence of phytotoxicity is not automatically possible. Also, a problem when incorporating phytotoxicity data in soil quality characterisation is the fact that in the literature there is very **little information** on dose-effect relationships in plants (NOECs or EC_xs in respect of contents of contaminants in the soil). This is slightly less of a problem for metals than, for example, for PAHs and PCBs, but the latter group of substances are not very relevant from the point of view of plant growth and soil pollution (personal communication Verkleij, 1997). For a functional approach to soil sanitation it is necessary, however, that plants be fully involved in the soil characterisation (ibid.). The general tenor where soil sanitation or standardisation is concerned is that plants are somewhat inferior, and the information used is mostly confined to fast growing crop plants like spinach and lettuce. Lokhorst (1997) found this too. He was searching for the usefulness of the toxicity data used in soil protection policy for purposes of assessing the suitability of polluted sites for specific nature target types (and target species of flora). However, this angle of approach proved too pretentious due to the **lack of toxicity data for target species** in nature policy (ibid.).

In view of the above, the question is to what extent specific information relating to metal toxicity *is* available for the **species of special concern** and taxa selected in chapter 2. Table 3.1 gives the ecological parameters based on plant growth in diagram form. On the basis of this it is clear that differentiation from the point of view of phytotoxicity must be sought in the differences (if any) in the metal tolerance of ornamental plants, crop plants, grasses, heather and species of tree in particular. Information on the functioning of symbiotic interactions in relation to metal contamination would also be useful.

Table 3.1 Qualitative ecological parameters and species of special concern per soil use category

Soil use category	Ecological parameters	Species of special concern
gardens and allotments	- ornamental plants - crop plants - grasses - native flora - symbiotic interactions (mycorrhizae and *Rhizobium*)	(-Trifolium repens)
verges and waste ground	- grassy vegetations - dwarf shrubs - trees - symbiotic interactions (mycorrhizae and *Rhizobium)*	(-Trifolium repens) - *Festuca rubra;* - *Festuca ovina;* - *Agrostis capillaris;* - *Erica tetralix;* - *Calluna vulgaris;* - relatively insensitive tree species
public parks & gardens and recreational amenities	- plants generally used in green amenities - grasses	- *Lolium perenne;* - *Poa pratensis;* - *Festuca rubra;* - 10 trees and 5 shrubs, including sensitive and relatively insensitive species.

Crop and ornamental plants in relation to metal contamination

An initial angle of approach for **crop and ornamental plants** is the **LAC warning levels** of the polluted soils working group of the Agricultural Advisory Committee on Environmental Contaminants (LAC, 1991). The warning levels give the total content of a given substance in the soil above which problems can be expected for agriculture. Phytotoxicity is one of the aspects included. If the warning level is exceeded, it may give the farmer cause for a closer inspection. Lexmond (personal communication, 1997) stresses that it is therefore not an environmental standard but a level below which the risk of problems is negligible and to which conditions are attached. For example, pH and fertilisation must be geared to the crop. A problem relating to the usability of the LAC levels for this present study is that in principle they only apply to soils to which heavy metals have been added ‘recently’. The total heavy metals contents obtained, which according to the LAC indicate a risk of **nascent phytotoxicity**, are probably too low for use-specific soil characterisation because the immobility of metal pollutants from before 1987 is expected to be higher. For the same reason **guidelines for the use of sewage sludge** on agricultural land (based on phytotoxicity criteria) cannot automatically be used (e.g. Chang *et al*., 1992), although the levels established for this purpose in the early 80s are not as precise as the LAC warning levels (personal communication Ernst, 1997). Consequently, for the present study the above levels and guidelines can only be seen as a **rough indication of nascent phytotoxicity** in crop and ornamental plants.

With regard to the metals selected as substances of special concern in §1.3, the LAC warning levels (**based on the phytotoxicity criterion**) are given in table 3.2.

Table 3.2 LAC warning levels based on the phytotoxicity criterion (according to the LAC, 1991). For the levels marked with *, the LAC warning levels from the perspective of human crop consumption are lower than the levels given. For levels marked with **, the LAC warning levels based on the exposure of grazers are more precise. Both comments apply to levels marked with ***.

Metal	Sandy soils	Clay soils	Peaty soils
Arsenic	30 mg /kg	50 mg /kg	50 mg /kg
Cadmium	5 mg /kg*	10 mg /kg*	10 mg /kg*
Chromium	200 mg /kg	300 mg/kg	300 mg/kg
Copper	50 mg/kg**	200 mg/kg**	200 mg/kg**
Lead	500 mg/kg***	800 mg/kg***	800 mg/kg***
Nickel	15 mg/kg	50 mg/kg	70 mg/kg
Zinc	100 mg/kg	350 mg/kg	350 mg/kg

Most studies of the relationship between crops and metal contamination have been carried out from the point of view of consumption by humans or livestock, or both. It is logical, therefore, that in most cases the internal metal contents of the edible parts of the plants should be examined. However, research from a phytotoxicological angle also means that effect concentrations usually do not relate to the content of a metal in the soil, but to the total content in leaf, shoot or root. This is a result *inter alia* of the complexity of the transfer from soil to plant. Table 3.3 gives a large amount of data relating to the metal toxicity of crop plants. The PT_{50} (a 50% reduction in growth in young plants as a result of phytotoxicity) is, according to Chang *et al*. (1992), a good threshold level because above this level a considerable reduction in yield can be expected from the mature plant. According to Macnicol and Beckett (1985) the EC_{10} can be considered to be the critical level, above which phytotoxicity occurs. This largely corresponds with the category of nascent PT (phytotoxicity), which is indicated by Sauerbeck (1989) (in Bockting & Van den Berg, 1992). The maximum exposure content relates to the metal content in the plant, with the growth of the plant being clearly inhibited or the external quality being so much worse that the plant can no longer be considered edible (Bockting & Van den Berg, 1992).

The question now is how **internal contents in the plant** can be of relevance to use-specific soil characterisation in view of the fact that soil sanitation policy makes use of standards based on the total contents of metals in the soil.

In the first place the possibility of abandoning total contents for site-specific soil characterisation is conceivable. A specific definition of the bioavailable part, as stated in § 3.1, is already used in practice for risk characterisation, e.g. by Adviesbureau TauwMilieu (Lokhorst, 1997). The internal metal contents of plants could also be used in diagnostic soil surveys. These might involve **bioindication** based on plants that already grow on the site. Another possibility is to carry out **bioassays** on soil material from the site in question, with plants being examined to see whether critical concentrations have been reached (Mocquot *et al*., 1997). Using species of plants that count as species of special concern for the soil use category is a conceivable angle of approach here. It would require additional research into species of special concern that are suitable for this purpose. A bioassay approach is currently being studied by Faber as well (personal communication, 1997). An additional advantage of bioassays is that any **combination effects of metals** will be brought to light.

In the second place, an attempt can be made to convert internal concentrations in the plant tissue to soil contents. The term **biological or bioconcentration** factor (BCF) is of importance for this. The BCF can be defined as the metal content of the plant (mg/kg dry matter) divided by the metal content in the soil (mg/kg dry matter).

Table 3.3 Metal contamination of the soil in relation to phytotoxicity limits in crop plants, based on (1) Chang *et al*., (1992), (2) Macnicol and Beckett (1985), (3) Sauerbeck (1989) and (4) Bockting & Van den Berg (1992). The contents are expressed in mg/kg dry matter (plant tissue).

Metal	PT_{50} (1)	EC_{10} (2)	Nascent PT (3)	Maximum exposure content (4)
Arsenic	no data available	1 (soya bean) 4 (cotton)	3-10	0.7 (tomato) 1.5 (cabbage) < 2.6 (bean and pea plants) 4.2 (bean) 10 (spinach) 76 (radish)
Cadmium	no data available	4-7 (soya bean) 5-10 (bean) 4-30 (wheat) 8-25 (barley) 20 (carrot) 25 (maize) 30 (radish) 30-35 (rye) 25-67 (lettuce) 90 (tomato) 120 (cabbage)	5-10	no reliable data available
Chromium	no reliable data available	1 (rye) 1 (bean)	2-20	> 0.5 (runner bean) > 1.6 (spinach) 5 (cabbage plant) > 13 (lettuce) > 57 (chard)
Copper	40 (maize)	11 (wheat) 5-21 (maize) 10-15 (lettuce) 15 (bean) 17 (sugar beet)	15-40	> 10 (carrot) > 11 (radish) >12 (runner bean) > 13 and > 23 (lettuce) > 18 (potato) 60 (spinach)
Lead	no data available	no data available	10-20	> approx. 30 (various vegetables) > 35 (lettuce) > 40 (carrots) > 57 (radish)
Nickel	no reliable data available	10-30 (tomato) 16-46 (wheat) 10-79 (maize) 30-40 (cabbage) 40-83 (bean) 120-150 (rye)	20-100	approx. 1,000 (lettuce)
Zinc	375 (bean) 475 (lettuce) 2,200 (maize)	60-250 (bean) 100-150 (sugar beet) 100-<200 (maize) 220 (barley) 108-500 (wheat) 150-530 (lettuce) 350 (tomato) 450 (soya bean) 380-550 (sorghum) 600 (spinach)	150-500	> 1,300 (radish) > 1,800 (lettuce)

The complexity of transfer from soil to plant means, however, that at best a range can be deduced from a BCF. When defining the BCF it is often assumed (implicitly, by presuming that it is a constant value) that the metal content of plants increases linearly with the metal content of the soil (Bockting & Van den Berg, 1992). However, this is too simplified an assumption. The functioning of symbioses with mycorrhizal fungi in relation to metal pollution alone shows this (see later on in this section as well).

Table 3.4 gives a number of examples of BCFs which might explain the relationship between phytotoxicity limits (table 3.3) and the total content in the soil, in which case this would only be relevant to phytotoxicity limits that are lower than the maximum crop contents that have been fixed for human consumption.

Using the BCFs from CSOIL (table 3.4) and the values for nascent PT (table 3.3) phytotoxic total contents in the soil were calculated for the substances of special concern. For **arsenic** this results in a range of 100-333 mg/kg, for **cadmium** it is 1.7-3.3 mg/kg, for **chromium** 67-667 mg/kg, for **copper** 50-133 mg/kg, for **lead** 333-667 mg/kg, for **nickel** 67-333 mg/kg and for **zinc** 50-167 mg/kg. Compared with the LAC warning levels given in table 3.2, the arsenic content in particular is significantly higher. As a range the other values correspond fairly closely to the LAC warning level.

A number of direct **total contents in the soil** relating to nascent phytotoxicity can be found in the literature. Alloway (in Watmough and Dickinson, 1995) gives 3-8 mg/kg for **cadmium**, 60-125 mg/kg for **copper**, 100-400 mg/kg for **lead**, 100 mg/kg for **nickel** and 70-400 mg/kg for **zinc**. The range for lead is significantly lower than the LAC warning level. Finally, NOEC and EC_x values are known for a number of crop and ornamental plants. These are given in table 3.5. The table shows that the spruce already experiences phytotoxicity effects at lower contents of cadmium, copper and lead than the contents given above for nascent toxicity. To what extent a sensitivity of this kind is general in trees is dealt with further on in this section. Finally, with regard to ornamental plants Verkleij argues (personal communication, 1997) that certain ornamental plants could well be very sensitive to metals, for example as a result of inbreeding.

Grasses and heather in relation to metal contamination

With regard to **grasses**, it is the differences (if any) in metal sensitivity among the selected species of special concern that are of particular relevance. *Agrostis capillaris* (personal communication Ernst, 1997; Dickinson *et al*., 1996; Verkleij, 1994), and *Festuca rubra* and *ovina* (Dickinson *et al*., 1996; Turner, 1994, Verkleij, 1994; Van Straalen and Verkleij, 1991) are known to be relatively insensitive to metal pollution, whereas *Lolium perenne* on the other hand can be considered to be a sensitive variety (personal communication Ernst, 1997; personal communication Verkleij, 1997). Unfortunately, it is not possible to link this difference in sensitivity direct to metal contents in the soil. The limit of metal tolerance in the aforementioned relatively insensitive grasses is location-, population- and individual-plant-specific and depends on, among other things, the period of adaptation to specific metals. Species of **heather** are also fairly robust when it comes to holding their own against relatively high metal contents (personal communication Ernst, 1997). No specific tolerance limits are known for the species of special concern *Calluna vulgaris* and *Erica tetralix, though*. The relative insensitivity of species of heather is linked to the good protection provided by the **ericoid mycorrhiza**. This type of mycorrhizal fungus is itself reputed to be relatively insensitive as well (Verkleij and Van Straalen, 1991).

Table 3.4 Minimum, maximum, and average BCFs (based on dry matter contents) of a number of crop plants. If unavailable, a range has been given (after Bockting & Van den Berg, 1992). The BCF is also the one used in the CSOIL model (Van den Berg, 1991).

Metal	Minimum	Maximum	Average	Range	CSOIL
Arsenic	no data available	no data available	no data available	0.001-0.1 (based on: beans, carrots, cabbage, onions, potatoes, tomatoes and radish)	0.03
Cadmium	0.043 (radish) 0.409 (lettuce) 0.533 (carrot) 0.783 (spinach)	0.324 (radish) 2.000 (lettuce) 1.481 (carrot) 3.235 (spinach)	0.171 (radish) 1.046 (lettuce) 0.867 (carrot) 1.851 (spinach)	n/a	3
Chromium	0.005 (radish) 0.017 (lettuce) 0.029 (spinach)	0.005 (radish) 0.017 (lettuce) 0.029 (spinach)	0.005 (radish) 0.017 (lettuce) 0.029 (spinach)	n/a	0.03
Copper	0.031 (radish) 0.098 (lettuce) 0.078 (carrot) 0.139 (spinach)	0.083 (radish) 0.237 (lettuce) 0.262 (carrot) 0.390 (spinach)	0.057 (radish) 0.165 (lettuce) 0.174 (carrot) 0.270 (spinach)	n/a	0.3
Lead	0.001 (radish) 0.015 (lettuce) 0.021 (spinach)	0.006 (radish) 0.063 (lettuce) 0.062 (spinach)	0.003 (radish) 0.036 (lettuce) 0.036 (spinach)	n/a	0.03
Nickel	0.011 (potato)	0.678 (potato)	0.075 (potato)	n/a	0.3
Zinc	0.116 (radish) 0.234 (lettuce 0.605 (spinach)	0.302 (radish) 0.497 (lettuce) 1.714 (spinach)	0.203 (radish) 0.355 (lettuce) 1.087 (spinach)	n/a	3

Trees in relation to metal contamination

The metal tolerance of grasses is often ascribed to their relatively short life. In relation to plant varieties that live longer, e.g. **trees**, it has long been assumed that there has been no development of metal tolerant species and/or ecotypes. Ernst (1990) established that trees are frequently not found on sites contaminated with metals. Although there is still little scientific evidence for metal tolerant populations of tree species (Turner, 1994), differences in metal sensitivity among species are known. Glimmerveen (1996) argues that the birch (*Betula*), pine (*Pinus*), alder (*Alnus*) and willow (*Salix*) are relatively insensitive to metal contamination. Ernst (personal communication, 1997) adds to these the poplar (*Populus*), Turner and Dickinson (1993) and Watmough and Dickinson (1995) the sycamore (*Acer pseudoplatanus*). The same can be presumed of the elder (*Sambucus nigra*) in view of the fact that it is found on sites heavily polluted with metals (Rebele *et al*., 1993). Many other species of tree, including the common oak (*Quercus robur*), are relatively sensitive (personal communication Ernst, 1997).

The differences in metal sensitivity among tree populations are mostly unlikely to be the consequence of the ‘classical evolution of metal tolerance” (Turner, 1994). Alternative angles of approach from a strategy of avoidance, such as phenotypical plasticity or specific mycorrhization, are currently being studied (Watmough and Dickinson, 1995). Turner and Dickinson (1993), for example, collected seedlings of *Acer pseudoplatanus* on metal contaminated and relatively clean sites and studied the growth of both types of seedling on metal contaminated and clean soil. Their study showed no difference in metal tolerance between the types. Some individuals of the two types exhibited phytotoxic growth disorders, other individuals of both types none at all. Most of the seedlings were, however, capable of surviving for three years. In addition to a minor role for basic tolerance, Turner and Dickinson (1993) suspect a role for phenotypical plasticity in the tree species. This enables the seedlings to survive until better environmental conditions arrive, e.g. when the roots reach relatively uncontaminated soil zones. Watmough and Dickinson (1995) demonstrated for the first time multiple metal resistance and co-resistance at a cellular level in the same species of tree. They conclude that the trees have an ability for optional adaptation (acclimatisation to metal stress). The induction of metal resistance can take place in specific circumstances (ibid.).

Table 3.5 Phytotoxicity data for a number of crop and ornamental plants. The effect range gives the range of effect concentrations in mg metal/kg soil (dry matter). The values concerned are standardised values unless stated otherwise between brackets. The data have been borrowed from Lokhorst (1997).

Metal	Effect parameter	Plant	Effect range
Arsenic	NOEC (growth)	cotton	22.3 - 143
	NOEC (growth)	soya bean	< 22.3
Cadmium	NOEC (growth)	grains	6.7 - 69.4
	NOEC (growth)	radish	< 14.3
	EC_{50} (growth)	radish	36.9 - 614
	NOEC (growth)	spruce	1.5
	NOEC	spruce	1.6
	(root length)	spinach	(sandy soil)
	EC_{10} (growth)	wheat	1.1 - 3.4
	EC_{50} (growth)		9.3
Chromium	NOEC (growth)	grains	230- > 1,474
Copper	NOEC (growth)	grains	233-> 516
	NOEC (growth)	spruce	16.5
	NOEC	spruce	16.5
	(root length)		(sandy soil)
Lead	NOEC (growth)	oats	137
	EC_{50} (growth)	oats	> 960
	NOEC (growth)	radish	180 – 12,031
	EC_{50} (growth)	radish	2,215 – 14,426
	NOEC (growth)	spruce	70 (sandy soil)
	NOEC (root length)	spruce	40 (sandy soil)
	NOEC	spruce	< 34
	(formation of	wheat	(sandy soil)
	mycorrhizae)		<125 - > 948
	NOEC (growth)		
Nickel	no data	no data	no data
	available	available	available
Zinc	NOEC (yield)	alfalfa	140 – 466
	NOEC (yield)	maize	82 – 466
	NOEC (yield)	lettuce	85 – 466
	NOEC (yield)	oats	208 – 1,275

Symbiotic interactions in relation to metal contamination

Mycorrhizal fungi boost a plant's absorption of nutrients and water (phytobiont). As stated in § 3.1, they can also have a positive or negative impact on metal uptake by the plant. For the plant they signify an expansion of its root system. About 70 - 90% of seed-bearing plants are estimated to have some form of mycorrhization (Van Straalen and Verkleij, 1991). Within the mycorrhizal fungi a distinction can be drawn between **ectomycorrhizal fungi** (ECM), **ericoid mycorrhizal fungi** and **vesicular-arbuscular mycorrhizal fungi** (VAM). Some types of mycorrhiza have proved very sensitive to metals, while others on the contrary are relatively insensitive (the ericoid mycorrhiza, for example). The following effects have been demonstrated in the presence of increasing quantities of heavy metals in the soil (in Tenner *et al*., 1997):

< 5 mg [Cu, Zn, Pb]/kg: no effect on mycorrhiza;
> 5 mg/kg: no effect on mycorrhiza, increase in uptake by plants;
20-30 mg/kg: 50% of the sensitive species of mycorrhiza disappear;
75 mg/kg: 100% of the sensitive species of mycorrhiza disappear.

In addition to the symbioses of plants with mycorrhizal fungi, another symbiosis is relevant to this study, namely that of **white clover** (*Trifolium repens*) and *Rhizobium leguminosarum*. Although toxicity data are available for white clover, they are in general irrelevant because a study has not been made of whether symbiotic nitrogen fixation works or not. *Trifolium repens* itself appears relatively insensitive to metals (Turner, 1994). Obbhard *et al*. (1994) points out that the reaction of the symbiosis to metal contamination is highly complex, and the nodulation and nitrogen fixation capacity is heavily influenced by soil factors such as soil heterogeneity, pH and nitrate concentration.

3.3 POSSIBILITIES FOR DIFFERENTIATION BASED ON SOIL ECOLOGY

In § 2.2 a simplified division of the soil into subecosystems was decided on. This picture will be refined somewhat in this section. The species of special concern come from the subsystem flora, whereas key species and processes relate mainly to soil fauna and microflora (bacteria and fungi). As already indicated, these subsystems are mainly of importance from the point of view of life support functions.

Decomposition

The central process relating to the life support functions of the soil is the **decomposition** of organic matter (Schouten *et al*., 1997). The TCB (1997) therefore rightly stated that this process is of importance to many forms of soil use and that the associated requirements must be taken into consideration in use-specific soil characterisation. Decomposition encompasses in fact all biological soil functions. During this process mineralisation occurs step by step, soil forming processes take place through the activity of organisms, and the leaching of nutrients is prevented by storage in biomass (Schouten *et al*., 1997).

Functional biodiversity

In chapters 2 and 3 of this report biodiversity of the soil ecosystem is considered from the point of view of functionality in particular. The relationship between biodiversity and the functioning of ecosystems is not clear, however (Hekstra *et al*., 1994; Lawton and Brown, 1993; Woodward, 1993; Giller *et al*., 1997). The possibility of evaluating the importance of biodiversity in regard to nutrient cycles depends on the way in which biodiversity is defined (for example, species diversity or diversity within functional groups), the ability of the ecological sciences to characterise specific functions and to identify critical biotic and abiotic parameters (Beare *et al*., 1995). In our view, making a functional cross section of soil ecosystems provides usable insights for selecting key species and processes for use-specific soil characterisation. Knowledge of life support functions, subprocesses and the groups of species involved are necessary for this purpose.

Indicator system

The indicator system of Schouten *et al*. (1997) provides a useful basis. Unfortunately, no direct relationships with concentrations of soil pollutants have yet been established in this system. They distinguish the following life support functions: (1) decomposition of organic matter, (2) recycling of nutrients, (3) availability of nutrients for plants, (4) soil structure formation and (5) stability of the soil ecosystem. With the exception of the last-mentioned function, it is possible to link relatively simple subprocesses and the associated groups of organisms to these life support functions. As a result of various complex and/or unknown trophic and mutualistic interactions and biological feedback mechanisms it is not easy to do this for the stability of a soil ecosystem. Although the importance of stability is obvious, it appears difficult to fit this function into use-specific soil characterisation. For purposes of the present study, the diagram of the **indicator system for life support functions** drawn up by Schouten *et al*. (1997) has been modified, with on the one hand selective choices being made with regard to subprocesses and taxa and on the other modifications and additions being made as a result of recent scientific opinions and on the basis of the usefulness criterion. Table 3.6 is the result. The selection made takes into account the fact that a careful combination of parameters containing information on the microflora, the soil fauna and an overall biological process in the soil (Van Straalen and Krivolutsky, 1996) results in specific effects or exposure routes being included.

Table 3.6 Functional cross section of the soil ecosystem for use-specific soil characterisation (adapted from Schouten *et al*., 1997).

Life support functions	Subprocesses	Key species and groups
Decomposition of organic matter	- fragmentation - conversion into organic substrate	- earthworms - micro flora
recycling of nutrients	- microbial activity - microflora grazing	- microflora - earthworms
nutrient availability for plants	- nutrients and water uptake - nitrogen fixation - nitrification	- mycorrhizal fungi - *Trifolium repens / Rhizobium* - nitrifying microflora
soil structure formation	- bioturbation / aggregate formation	- earthworms

Earthworms as key group

It is obvious that the subecosystem soil fauna can be further subdivided, using a functional classification (Faber, 1991), for example, or by distinguishing scale-related “soil fauna subecosystems” (Pokarzhevskii, 1996). **Earthworms** have been selected as the key group within soil fauna. This selection can be easily justified from the point of view of functional significance, and on the basis of the ecological parameters discussed in chapter 2 also appears to provide opportunities for differentiation of the soil use categories, because in soils with high biological activity the importance of earthworms is greater than in soil types having moderate to little biological activity (in Gleichman-Verheijen *et al*., 1991).

Earthworms (Lumbricidae) make an extremely important contribution towards the formation of the soil structure, the conversion of organic matter and hence soil fertility (Marinissen, 1995; Brown, 1995; Coleman and Crossley, 1996). It is therefore not surprising that a reduction in earthworms can result in a lower crop plant biomass (Stinner *et al*., 1997). The function of earthworms in relation to soil processes is not uniform, however, but varies per ecological category and type. In this context the distinction between **‘epigeics’** (relatively small varieties found in the humus layer), **‘endogeics’** (varieties found in the soil) and **‘anecics’** (relatively large varieties found in both compartments) is relevant (Brown, 1995; Beare *et al*., 1995; Lavelle *et al*., 1995; Schouten *et al*., 1997). The first category is of particular importance in the initial stages of the decomposition process (fragmentation), the second category absorbs relatively large quantities of soil material (ingestion) and provides for significant improvements in the soil structure and soil aeration (bioturbation and aggregate formation and stability) and the last category, finally, buries the humus in the soil (the mixing of decaying leaf matter with other soil fractions promotes decomposition and mineralisation processes) and forms extensive systems of vertical tunnels which are of importance to, among other things, the gas and water regimes of the soil (Brown, 1995). All categories of earthworms also have a direct or indirect impact on the microflora of the soil ecosystem. In addition to their impact via habitat quality or providing a niche through ingestion, this impact involves *inter alia* dispersion and selective grazing (Beare *et al*., 1995; Brown, 1995). In our view Schouten *et al*. (1997) wrongly ignore the selective grazing on microflora by earthworms. All in all it appears that earthworms are involved in many subprocesses of life support functions, which makes them eminently suitable as a key group for use-specific soil characterisation. Added to this, it is known that earthworms respond relatively swiftly to disturbances (Marinissen, 1995). Furthermore, a relatively large amount of knowledge is available on the relationship between earthworms and metal toxicity.

Table 3.7 Metal toxicity data for earthworms (from Lokhorst, 1997). The effect limits are approximated using the method of calculation given in the text.

Metal	Species	Effect parameter (reproduction)	Metal content in standard soil (in mg/kg)	Specific effect limits (in mg/kg)
Arsenic	no data available	no data available	no data available	no data available
Cadmium	*Dendrobaena rubrida*	NOEC	24	EC_{10} : 27
	Lumbricus rubellus	NOEC	13	EC_{25} : 54
	Eisenia fetida	NOEC	18	
	Eisenia andrei	EC_{10}	19	
Chromium	*Eisenia fetida*	NOEC	405	EC_{10} : 405
	Eisenia fetida	NOEC	405	EC_{25} : 810
Copper	*Dendrobaena rubrida*	NOEC	150	EC_{10} : 132
	Dendrobaena rubrida	NOEC	183	EC_{25}: 264
	Eisenia fetida	NOEC	500	
	Eisenia fetida	NOEC	1,000	
	Eisenia andrei	NOEC	142	
	Lumbricus rubellus	NOEC	40	
	Lumbricus rubellus	NOEC	99	
	Lumbricus rubellus	NOEC	17	
	Allolobophora caliginosa	NOEC	94	
Lead	*Dendrobaena rubrida*	NOEC	736	$EC_{10:347}$
	Dendrobaena rubrida	NOEC	741	$EC_{25:755}$
	Dendrobaena rubrida	NOEC	171	
	Dendrobaena rubrida	EC_{70}	775	
	Lumbricus rubellus	NOEC	241	
	Eisenia fetida	NOEC	1,000	
Nickel	*Lumbricus rubellus*	NOEC	65	EC_{10} : 143
	Eisenia fetida	NOEC	233	EC_{25}: 328
	Eisenia fetida	EC_{40}	583	
Zinc	*Eisenia fetida*	NOEC	318	EC_{10} : 802
	Eisenia fetida	NOEC	1,273	EC_{25}: 1,603
	Eisenia fetida	NOEC	1,273	

Earthworms in relation to metal contamination

Practical examples relating to earthworms provide some insight into the direct and indirect effects of metal contamination on the functioning of a soil ecosystem. Disrupted functioning can be recognised especially by a thick layer of non- or poorly decomposed organic matter. A study carried out by Spurgeon *et al*. (1996) looked at *inter alia* the abundance of earthworms in a gradient of metal contamination (cadmium, copper, lead and zinc). In the immediate vicinity of a smeltery (300 mg Cd/kg, 3,000 mg Cu/kg, 15,000 mg Pb/kg en 35,000 mg Zn/kg) it turned out that there were no earthworms at all, resulting in a substantial accumulation of non-decomposed leaf matter. Although the concentrations of metals decreased exponentially with the distance to the smeltery, there was a reduction in the total abundance of earthworms up to a distance of 3 kilometres (ibid.). In orchards in the fruit growing region of the Netherlands no earthworms were found at copper contents of 60 mg/kg (in the top 10 cm) (in Tenner *et al*., 1997). In a different study it was demonstrated that earthworms are only found in low to very low numbers at copper concentrations of up to 50 mg/kg (ibid.).

With regard to indicating minimum soil quality requirements for earthworms, it would be worthwhile using toxicity data on at least one variety per ecological category per metal. This is not feasible with the data available. Table 3.7 presents metal toxicity data on earthworms. It only includes the effects on reproduction because reduced reproduction greatly minimises the restorative capacity of a population following a perturbation, which could result in extinction (Marinissen, 1995). The data have been taken from Lokhorst (1997).

On the basis of these data it is possible to differentiate according to soil use, because certain forms of use require lower earthworm activity than others. This differentiation can be translated into an approach (naturally open to discussion) consisting of determining a geometric average EC_{10} and EC_{25} per metal (more about this in § 3.4). A rough method of calculation analogous to Van Beelen and Doelman (1997) is sufficient in the context of this study to obtain ranges of maximum values (effect limits):
The EC_{10} is approximated by equating NOEC and EC values up to EC_{20} to EC_{10}, dividing EC_{20} to EC_{50} by 3 and dividing EC_{50} to EC_{10} by 10. EC_{25} is approximated by multiplying NOEC and EC values up to EC_{20} by 2, equating EC_{20} to EC_{50} to EC_{25} and dividing EC_{50} to EC_{95} by 3. Effects stronger than EC_{95} are not used. The total reliability of the approximations increases with the spread of effect parameters

As an alternative to the use of total contents in the soil, the key group earthworms can function extremely well as a bioindicator of soil quality (Schouten *et al*., 1997). For example, it is possible to look at species diversity per ecological category or body concentrations of metals. References for the various soil use categories can be established on the basis of inventory data in similar (but clean) soils.

Nitrification as key process

In the context of this study microbial processes and taxa related to the life support functions given in table 3.6 are particularly relevant. In addition to the micorrhizal fungi discussed in the preceding section they are mainly the various microbiological transformations within the nitrogen cycle. In general, there is a high degree of functional redundancy within the microflora. Soil processes are measured as sum parameters, with the result that it is not known which species do and which do not experience an effect (Tenner *et al*., 1997). The less functional redundancy there is the more this disadvantage diminishes, as there are fewer species making up the sum parameter. Also, processes for which there is little functional redundancy are more vulnerable to species shifts. A useful angle of approach for identifying suitable **key processes** is to compile an inventory of specific subprocesses for which there is little functional redundancy within the nitrogen cycle. This applies in particular to **biological nitrogen fixation** and to **nitrification**.

Although biological nitrogen fixation occurs fairly generally in nature, it involves only a relatively small number of micro-organisms. The use of nitrogen fixation as a general key process is not advisable, however, because in spite of the marginal number of species, very different types of organisms are involved. For example, asymbiotic nitrogen fixation in the soil is carried out by aerobic (*Azotobacter*), micro-aerophile (*Klebsiella*) and anaerobic, organotrophic (*Closstridium*) bacteria and free-living cyanobacteria. Symbiotic nitrogen fixation is known *inter alia* in bacterial associations (*Rhizobium, Bradyrhizobium*) with Leguminosae and symbioses of actinomycetes (*Frankia*) with non-papilionaceous angiospermae (for example, *Alnus*). The species in question are, moreover, frequently confined to a narrow, species-specific range of environmental conditions (Beare *et al*., 1995). From the point of view of use-specific soil characterisation the ***Rhizobium/Trifolium repens*** association is the one that is of the most practical and indicative use as a key process. As already shown in § 3.2 it is difficult to link toxicity data to this association and so this will not be worked out in any more detail.

Table 3.8 Metal toxicity data for nitrification (based on Van Beelen and Doelman, 1997 and Lokhorst, 1997). It is not known whether a correction has been made for soil type for values marked (*). The effect limits have been approximated using the method of calculation given in the text.

Metal	Effect parameter	Metal content in standard soil (in mg/kg)	Specific effect limits (in mg/kg)
Arsenic	no data available	no data available	no data available
Cadmium	EC_{32}	100*	EC_{25} : 171
	EC_{94}	675	EC_{50} : 237
	EC_{74}	529	
	EC_{70}	593	
	EC_{77}	560*	
Chromium	NOEC	119	EC_{25} : 104
	EC_{59}	220	EC_{50} : 157
	EC_{87}	236	
	EC_{81}	260*	
Copper	EC_{60}	364	EC_{25} : 316
	EC_{31}	311	EC_{50}: 399
	EC_{43}	279	
	EC_{75}	136	
	EC_{75}	1,364	
	EC_{75}	13,636	
	EC_{45}	320	
Lead	NOEC	1,486	EC_{25} : 1,932
	EC_{26}	1,137	EC_{50} : 2,632
	EC_{10}	1,019	
	EC_{7}	943	
	EC_{14}	1,035*	
Nickel	NOEC	250	EC_{25} : 217
	EC_{68}	1,250	EC_{50} : 410
	EC_{64}	295*	
Zinc	NOEC	208	EC_{25} : 529
	NOEC	1,403	EC_{50} : 710
	NOEC	13,345	
	NOEC	241	
	NOEC	225	
	NOEC	40	
	EC_{17}	254	
	EC_{58}	364	
	EC_{24}	305	
	EC_{39}	275	
	EC_{40}	325*	

The process of **nitrification** in the soil is of substantially practical importance. Ecological relevance in the various steps of the nitrogen cycle, in particular in relation to soil fertility, has been much studied (Rother *et al*., 1982). From this point of view ammonification and nitrification appear to be highly essential. However, functional redundancy with respect to ammonification is considerable, whereas only a small number of species are involved in nitrification (personal communication Doelman, 1997; Van Beelen & Doelman, 1997; Health Council, 1991; Rother *et al*., 1982). In addition to a number of bacteria genuses involved in chemo-autotrophic nitrification, e.g. *Nitrosomonas*, *Nitrococcus* and *Nitrobacter*, there are a number of bacteria (*Arthrobacter*) and actinomycetes that play a part in heterotrophic nitrification (Beare *et al*., 1995). Marginal redundancy makes the process very susceptible to perturbations in environmental conditions. Perturbation in the balance of the two phases within the nitrification process (conversion of ammonium into nitrite and conversion of nitrite into nitrate) can result *inter alia* in an accumulation of the toxic intermediary nitrite in the soil (Dusek, 1995). In our view, nitrification is a useful key process in use-specific soil characterisation. The process is eminently suitable for prognostic or diagnostic tests of soil quality (Doelen and Vonk, 1994; Health Council, 1991), provides links for differentiation via soil use requirements (soil fertility) and, in addition, much is already known about the relationship between nitrification and metal contamination.

Nitrification in relation to metal contamination

A great deal is already known about the effects of heavy metals on microbial communities and processes. Recent overviews are given in Baath (1989) and Wuertz and Mergeay (1997). Various studies show that compared with most other microbial processes nitrification is highly sensitive to heavy metals (Van Beelen and Doelen, 1997; Dusek, 1995; Health Council, 1991; Rother *et al*., 1982).

Table 3.8 presents metal toxicity data for nitrification. The data are taken from Lokhorst (1997) and Van Beelen and Doelman (1997). On the basis of these data it is possible to differentiate according to soil use because certain forms of use require or permit a lower soil fertility (nutrient availability for plants) than others. This distinction can, for example, be translated into an approach (naturally open to discussion) that consists of determining a geometric average EC_{25} and EC_{50} per metal (more about this in § 3.4). As in the case of earthworms, a rough method of calculation analogous to Van Beelen and Doelen (1997) can suffice as a method to obtain ranges of maximum values (effect limits):
The EC_{25} is approximated by multiplying NOEC and EC values up to EC_{20} by 2, equating EC_{20} up to EC_{50} to EC_{25} and dividing EC_{50} to EC_{95} by 3. EC_{50} is approximated by multiplying NOEC and EC values up to EC_{20} by 3, equating EC_{20} to EC_{50} to EC_{50} and dividing EC_{50} to EC_{95} by 2. Effects stronger than EC_{95} are not used. The total reliability of the approximations increases with the spread of effect parameters.

3.4 MINIMUM SOIL REQUIREMENTS OF SOIL USE CATEGORIES

Sections 3.2 and 3.3 look at the extent to which the subecosystems flora, soil fauna and general and microbial processes present opportunities for differentiating soil quality requirements. In spite of the fact that certain metal toxicity data are lacking, the inventory has produced many leads.

Plant growth occupies a central position in use-specific soil characterisation. As indicated in § 2.2, the presence of specific plant species alone gives substance to the soil use category. Chapter 2 gives species of special concern and ecological parameters per soil use category. The gardens and allotments category sets the highest requirements for plant growth. In these cases nascent phytotoxicity in sensitive species must be ruled out. The public parks & gardens and recreational amenities category likewise sets fairly high requirements. Admittedly, the varieties in question here are the common ones, but the failure to grow of common species of plant such as the common oak is in our view too great a restriction on the soil use. Also, the relatively sensitive grass species *Lolium perenne* - used for sports fields - imposes limitations on allowing heavy metals in the soil. Verges and waste ground impose the lowest soil quality requirements from the perspective of plant growth. All selected species of special concern for this soil use category are relatively insensitive to metals. This applies both to the grass species and to the species of heather. As regards planting trees, relatively insensitive species like poplar, willow, birch, sycamore and alder will also suffice.

To what extent can this differentiation based on plant growth be translated into total contents in the soil? In principle, there are no specific toxicity data on the species of special concern. Nonetheless, an attempt has been made – based on the available numerical values and qualitative views of the relative metal tolerance of types of plants from section 3.2 and expert judgement - to give an estimated **maximum value per soil use category**. It should be emphasised here that this must be taken as a very broad range. The values are given in table 3.9 (as F). The table also includes the maximum metal values for earthworms (R), nitrification (N) (both calculated on the basis of the method described in § 3.3) and by way of illustration the intervention values (I) of the Soil Protection Guideline (VROM, 1995). This gives a picture of the relative sensitivity of the subecosystems to the various metals. The value of the most sensitive subsystem is printed bold. This can be seen as a range of the metal content, with the soil being suitable for the form of use in question from the perspective of ecological parameters. Plant growth often proves to impose the strictest requirements on soil quality (particularly in the case of arsenic, cadmium, copper, nickel and zinc). This corresponds with Lokhorst (1997). For earthworms it has been assumed that they are of eminent importance in gardens and allotments (EC_{10} values from table 3.7) and of less importance for the other soil use categories (EC_{25} values from table 3.7). In our opinion the importance of nitrification in the verges and waste ground category is relatively small compared with the other two soil use categories (EC_{50} values and EC_{25} values respectively from table 3.8).

Table 3.9 Minimum soil quality requirements based on plant growth (F), earthworms (R) and nitrification (N). The intervention values (standard soil) from soil sanitation policy are also given (I). See text for explanation.

Metal	Gardens and allotments	Public parks & gardens and recreational amenities	Verges and waste ground
Arsenic	**F = 20 mg/kg**	**F = 100 mg/kg**	**F = 300 mg/kg**
	R = n/a	R = n/a	R = n/a
	N = n/a	N = n/a	N = n/a
	I = 55 mg/kg	I = 55 mg/kg	I = 55 mg/kg
Cadmium	**F = 3 mg/kg**	**F = 10 mg/kg**	**F = 50 mg/kg**
	R = 27 mg/kg	R = 54 mg/kg	R = 54 mg/kg
	N = 171 mg/kg	N = 171 mg/kg	N = 237 mg/kg
	I = 12 mg/kg	I = 12 mg/kg	I = 12 mg/kg
Chromium	F = 200 mg/kg	F = 300 mg/kg	F = 600 mg/kg
	R = 405 mg/kg	R = 810 mg/kg	R = 810 mg/kg
	N = 104 mg/kg	**N = 104 mg/kg**	**N = 157 mg/kg**
	I = 380 mg/kg	I = 380 mg/kg	I = 380 mg/kg
Copper	**F = 60 mg/kg**	**F = 100 mg/kg**	**F = 200 mg/kg**
	R = 132 mg/kg	R = 264 mg/kg	R = 264 mg/kg
	N = 316 mg/kg	N = 316 mg/kg	N = 399 mg/kg
	I = 190 mg/kg	I = 190 mg/kg	I = 190 mg/kg
Lead	F = 400 mg/kg	**F = 500 mg/kg**	**F = 700 mg/kg**
	R = 347 mg/kg	R = 755 mg/kg	R = 755 mg/kg
	N = 1,932 mg/kg	N = 1,932 mg/kg	N = 2,632 mg/kg
	I = 530 mg/kg	I = 530 mg/kg	I = 530 mg/kg
Nickel	**F = 50 mg/kg**	**F = 100 mg/kg**	**F = 300 mg/kg**
	R = 143 mg/kg	R = 328 mg/kg	R = 328 mg/kg
	N = 217 mg/kg	N = 217 mg/kg	N = 410 mg/kg
	I = 210 mg/kg	I = 210 mg/kg	I = 210 mg/kg
Zinc	**F = 100 mg/kg**	**F = 200 mg/kg**	F = 1,000 mg/kg
	R = 802 mg/kg	R = 1,603 mg/kg	R = 1,603 mg/kg
	N = 529 mg/kg	N = 529 mg/kg	**N = 710 mg/kg**

	I = 720 mg/kg	I = 720 mg/kg	I = 720 mg/kg

4 ALTERNATIVE ANGLES OF APPROACH TO MINIMUM SOIL QUALITY

4.1 SURFACE LAYERS

Chapter 3 looked at the minimum soil quality for the various soil use categories. It goes without saying that this relates mainly to the quality of the upper layer of the soil, that is to say the layer in which the most relevant ecological soil functions take place. An alternative angle of approach to minimum soil quality is to produce a clean upper layer on top of the pollution and to find the minimum thickness for this layer. Faber (1997) already indicated that the requirements a soil must meet for a given use need not necessarily result in standards for contaminants but, for example, in **land-use requirements for surface layers**. In the context of a use-specific approach to the soil, the question of the extent to which the various soil use categories could result in differentiated land-use requirements for surface layers is relevant. A number of practical examples and results of studies were examined for this purpose.

The surface layer principle

Applying a surface layer as a soil sanitation technique stems from the idea that restoring the multifunctionality of the soil is in general not necessary in urban areas (Van Wachem *et al.*, 1987). The goal of the surface layer principle is to make a contaminated area suitable for a specific, high-grade use (residential, recreational, etc.) in a relatively inexpensive but environmentally sound manner. The essence of soil sanitation according to the surface layer principle is to remove any possibility of contact between the pollution and routes of dispersion (Roeloffzen & Driessen, 1989). A surface layer consists of a clean covering layer with additional, isolating facilities on top of contaminated soil (Van Wachem *et al.*, 1987). The surface layer principle was opted for at an early stage in large cities like Amsterdam (Van Wachem *et al.*, 1987) and Rotterdam (Roeloffzen & Driessen, 1989) in particular for reasons of returns on costs, in spite of the fact that the desirability and risks of surface layers were still the subject of much discussion both at the Environment Ministry (among others Keuzenkamp, 1988) and in the environmental movement (among others Van Pelt, 1988). Keuzenkamp (1988) argued that it was in theory possible to virtually eliminate the restrictions on use of a surface layer and to design a specific surface layer for each form of use, but that in practice there was still too little experience in applying and managing isolating facilities at soil sanitation sites and with measures for making these sites usable again (to a limited degree). Van Pelt (1988) argued that a surface layer was not a satisfactory alternative to soil sanitation and attested only to a short-term view. As she sees it, surface layer cleanups may be fast and initially inexpensive, but they only defer the problem of soil pollution and even enlarge it, because replacement of the isolation materials, which will in time be necessary, will be a technically difficult and expensive undertaking as a result of the buildings that will by then have been erected (ibid.).

The Griftpark

The Griftpark was the experimental site for large-scale soil sanitation operations in the Netherlands. In the case of the Griftpark, on the site of a former gasworks, the city of Utrecht consciously opted to apply the surface layer principle (Van Pelt, 1993). It was unwilling to make any concessions to soil quality, but did not have the money for a complete cleanup (personal communication Leurink, 1997). The measure was taken in combination with the ICM (Isolate Control Monitor) strategy for the core of the pollution, including lowering the groundwater level. The proposed thickness of the surface layer varied from a maximum of 1.5 metres where trees grow and an average of 1 metre. Contact with the upper layer of the pollution was minimised by coarse sand and gravel to obstruct capillary rise and prevent roots growing through in combination with non-woven cloth and in special cases bentonite or foil with a supporting layer (Van der Drift *et al.*, 1992). In December 1997, the Griftpark was finally completed. In the end the thickness of the surface layer departed from the proposed thickness given above: where areas were sown with grass it is 60 centimetres, the planned birch wood was given 3 metres of clean soil, while a boggy layer of clean soil a few dozen centimetres thick was deemed enough for surface rooting alders and willows (Didde, 1998).

Steendijkpolder-Zuid

Another example of the use of surface layers is the cleanup of the Steendijkpolder-Zuid in Maassluis in the late eighties (TCB, 1988). Entirely for reasons of environmental hygiene, the ingestion of soil by children and absorption via crops in allotments, it was decided to apply a surface layer above the pollution which consisted partly of polluted dredging sludge and partly of dumped refuse. The new surface layer to be applied consisted of 70 cm of humus, 30 cm of coarse sand and a regulating layer of 30 cm of gravel with permeable plastic film on the top and bottom (as a barrier for roots and soil organisms and to avoid pores between the gravel filling up with sand or dredging sludge). All this was based on reports from the engineering consultancy DHV (ibid.). The TCB's recommendation (ibid.) was in agreement with the 70 cm of humus but it questioned the need for the other measures. The groundwater level in the polder was 65 cm below surface level. The saturated zone therefore formed a barrier to the deep rooting of trees in particular (there was no cereal farming). The argument of digging work for the 30 cm sand layer was refuted: mixing soil with dredging sludge would not bring any serious pollution to the surface. With regard to the isolating layer the TCB calculated that the effects of upwards transport by plants, via groundwater and as a result of earthworm activity, had been overestimated.

Surface layer thickness from the ecological point of view

Although a reasonable amount of experience has been gained in applying surface layers in the practice of soil sanitation it is worthwhile reconsidering a number of aspects and placing them in the context of setting ecological parameters for use-specific soil quality. Additional information has meanwhile become available on the requirements imposed by ecology in particular on the thickness of the surface layer. The rooting depth of flora especially is a relevant engineering requirement. In addition to the type of plant, it depends in particular on the soil structure and groundwater level (Van Wachem *et al.*, 1987).

Table 4.1 Minimum thicknesses of soil layers for four types of vegetation (in Cairney, 1996). It is assumed that (1) 30% of the available thickness is of good quality and (2) water retention is sufficient.

Vegetation	Thickness of the surface layer
grasslands only	150 mm
crop and ornamental plants and grasses	200-300 mm
scrub	500 mm
fruit and other trees	1 to 1.5 metres

Table 4.2 Mechanical resistance for root growth (in Cairney, 1996).

Soil density (g/cm^{3})	Effect on root growth
1.37	root growth is impeded
1.37 to 1.77	root growth declines linearly
1.74 to 1.83	root growth stops entirely
1.55 (clay soils)	root growth severely restricted
1.85 (sandy soils)	root growth severely restricted

Root depth and mechanical resistance

Cairney (1996) considers various aspects of surface layers on the basis of data from the literature. He argues simply that an adequate thickness for a surface layer can be provided for the specific types of vegetation chosen (table 4.1) *and* that plants can be prevented from rooting deeper by compacting the contaminated subsoil to the right density (table 4.2). Compacting is much easier than precluding contact between roots and the deeper pollution by installing a foil barrier or influencing the level of the groundwater. It appears that at certain critical soil densities root growth is increasingly restricted and can even be stopped entirely. As far as is known this phenomenon has not been used before in connection with soil sanitation, whereas **compacting soil** is routine in creating sufficient load-bearing capacity in the soil for building purposes. Wachem *et al.* (1987) do propose a surface layer structure with sharp transitions in types of soil, however. If the groundwater level is too close to the clean surface layer, Cairney (1996) considers an intermediate layer necessary to prevent capillary rise.

Covering polluted river sediments

With regard to the **root depth of species of plants** research has also been carried out at the AB-DLO Research Institute for Agrobiology and Soil Fertility in Haren (Van Driel *et al.*, 1995; Van Noordwijk *et al.*, 1995). The aim of the research was to establish the requisite thickness of a clean covering layer on polluted river sediments for agricultural purposes. In an experiment with a layer of clean soil of up to 70 cm thick it turned out that a significant effect from the polluted under-layer could be perceived in 60% of the plants. In an experiment with a covering layer of up to 1.6 metres thick a significant effect could be perceived in 50% of the plants (table 4.3). What was striking was that after 11 years of research no perceivable migration of heavy metals had taken place from the polluted subsoil to the clean covering layer (ibid.), not even as a result of a higher groundwater level (Van Noordwijk *et al.*, 1995). This would mean either that plants send roots down into the polluted sediment, or roots have some kind of impact in their immediate micro-environment on the upward migration of heavy metals, whereas this does not result in significant increases when the concentration of metals in the surface layer is measured. The conclusion of Van Driel *et al.* (1995) is that a clean layer of soil of more than 1.6 metres deep is necessary in order not to exceed the maximum acceptable concentrations in food and food crops (grains). Another conclusion of the study was that a lower groundwater level results in deeper root growth and would necessitate an even thicker layer of soil (Van Noordwijk *et al.*, 1995) if no isolating intermediate layer were installed.

Surface layer thickness for gardens and allotments

For the soil use categories dwellings with gardens and dwellings with vegetable gardens the Association of Municipalities in the Netherlands VNG (Moet, 1995) assumes a minimum depth of 1.5 metres, in which the soil quality is not permitted to exceed the critical values in connection with the absorption of substances by plants (followed by consumption) and possibilities of contact during digging work. This is amply sufficient if use is made of a separating intermediate layer. It is a wider margin than the one opted for by the TCB for creating vegetable gardens (see above), in which planting certain trees (root depth 1 to 1.5 metres, see table 4.1), for example, can present problems. De Ruiter (personal communication, 1997), incidentally, considers a surface layer thickness of 25 cm sufficient for vegetable gardens which, strictly speaking, focus on vegetable-growing. In that case it is not necessary for the layer beneath to be entirely clean, but it does need to be functioning properly.

Surface layer thickness for the other categories

For the other two forms of soil use the VNG (Moet, 1995) has set the highest contents in the upper 0.5 metre as the standard. A clean layer of this depth is pointless, seeing that trees - an indispensable part of green amenities as evidenced by the available information - root deeper than half a metre and are also relatively sensitive to heavy metals (§3.2). Also, in practice a maximum planting depth of 1.2 metres is adhered to when planting trees (Vegter *et al.*, 1995). However, it may be possible to opt for a variation in soil depth, as applied in the Griftpark (personal communication Leurink, 1997), because a surface layer is not required under paved surfaces, for example (personal communication De Ruiter, 1997). Various scientists do warn against the temptation of permitting high concentrations of contaminants in these places, though. For example, Ernst (personal communication, 1997) points to the mobilising effect of salt on bound heavy metals and the idea of adsorption of heavy metals to porous asphalt (this has not yet been demonstrated, incidentally). Doelman (personal communication, 1997) sees hazards in the mobilisation of toxicants by micro-organisms in anoxic conditions.

Table 4.3 The thickness of a clean covering layer (up to a maximum of 1.6 metres) on polluted river sediment, whereby a number of crops have not (yet) exhibited any effects from cadmium, copper and zinc respectively (Van Driel *et al.*, 1995). NNOEL = no no-effect level found; NEO = not enough observations.

Crop	No effect depth		
	Cd	Cu	Zn
celeriac, tuber	1.2	0.4	1.2
celery, foliage	1.6	1.6	1.2
endive, foliage	1.2	NEO	0.8
potato, tuber	1.6	NNOEL	NNOEL
winter wheat, grain	NNOEL	NNOEL	NNOEL
winter wheat, stalk	NNOEL	NNOEL	NNOEL
summer wheat, grain	1.2	NNOEL	NNOEL
summer wheat, stalk	NNOEL	0.8	NNOEL
barley, grain	1.6	NNOEL	NNOEL
barley, stalk	0.6	NNOEL	NNOEL
maize, 1985	1.2	0.4	1.2
maize, 1990	1.6	0.4	1.6

Conclusion

There appears to be little point in approaching use-specific differentiation in surface layer structure or thickness from the point of view of soil use categories. It is far more logical to opt for a more site-specific approach seeing that it is very easy to make very local differences in the surface layer. In some cases a deeper soil is necessary, whereas other situations would require no surface layer at all. From the point of view of optimising controllability Roeloffzen & Driessen (1989) advocate the use of uniform surface layers, however, geared to the most sensitive use on a site. In conclusion, it can be stated that it is necessary to know how the layer to be covered is contaminated as well as the proposed plants in order to establish both the structure of the surface layer and its thickness. Furthermore, the role of capillary rise does indeed appear to have been overestimated, whereas root depth on the other hand appears to have been underestimated. Impeding root growth can be an option for reducing the minimum thickness of the surface layer, however.

4.2 PHYTOREMEDIATION AND GROWTH OF VEGETATION ON HEAVY METAL CONTAMINATED SOILS

This section approaches the subject from the angle of the possibility and impossibility for vegetation to grow on soils contaminated with heavy metals. The central issue is what plants can still grow and to what extent can the growth of vegetation contribute to an improvement in the quality of the soil? Presenting the question this way produces a close link to a topical subject in phytotoxicology, phytoremediation (personal communication Verkleij, 1997; personal communication Lexmond, 1997; personal communication Doelman, 1997).

Plant growth and development of vegetation on soils contaminated with heavy metals
The example given below shows that much is still possible where plant growth on metal contaminated soils is concerned. The aspect of **soil heterogeneity** probably plays a big part in this, however. Within a radius of 1 to 2 kilometres around a copper smeltery in Poland various species of plant were found in patches on soil heavily polluted with copper and lead (Rebele *et al.*, 1993). The average copper contents of the top layer of soil were more than 15,000 mg/kg, the lead contents more than 2,000 mg/kg. The dominant species were field bindweed (*Convolvulus arvensis*), couch grass (*Elymus repens*), wood small-reed (*Calamagrostis epigejos*) and elder (*Sambucus nigra*). The metal

contents in the shoots of mugwort (*Artemisia vulgaris*) varied from 665-2,340 mg/kg (dry matter) for Cu, 215-2,301 for Zn, 189-1,031 for Pb and 0.75- 12.4 for Cd (ibid.).

In practice it appears that once plant growth has been achieved on a metal contaminated site, other species can establish themselves spontaneously (personal communication Lexmond, 1997). To this end it may be necessary to encourage plant growth in the first place by mixing the soil with, for example, compost, lime and Berengiet in order to immobilise the metals and by sowing metal tolerant species. This type of approach ties in with the theme of **phytoremediation**.

Phytostabilisation and phytoextraction

Verkleij (personal communication 1997) distinguishes two approaches to phytoremediation, namely **phytostabilisation** and **phytoextraction**. Phytostabilisation involves heavily polluted soils, in which soil additives (for example, iron and aluminium silicates) bind the heavy metal irreversibly. At the same time metal tolerant grass seed is sown and a small dose of nitrogen administered. After 5 to 6 years the normal vegetation reappears with a biodiversity comparable to non-contaminated soils. Phytoextraction is worthwhile in cases of lightly contaminated soils (just above the intervention values) and uses hyper-accumulative species of plants such as Alpine pennycress (*Thlaspi caerulescens*) and species of *Brassica*. After 10 to 15 years the plants are harvested and it is possible to calculate on the basis of pilot experiments how long it takes before a soil drops below the intervention value. The study being carried out on this subject is being coordinated by the Vrije University, Amsterdam and was commissioned by the European Community.

Phytoremediation using trees

In addition to herbs and grasses, trees can also be used for fixing heavy metals. Experience has been gained with, among others, silver birch (*Betula pendula*), sycamore (*Acer pseudoplatanus*) and various species of willow (*Salix* spec.) (Duncan *et al.*, 1995; Labrecque *et al.*, 1995; Glimmerveen, 1996). The advantages of using trees for the phytoremediation of metal contaminated sites include the cost and the fact that trees perform an erosion-reducing function, initiate soil development and improve the visual quality of the site (Glimmerveen, 1996). Species of willow have therefore been developed which combine swift growth and metal tolerance, with the aim of achieving maximum metal uptake from the soil and cleaning of the soil compartment. The reason why this technique is not yet in general use is a lack of information on the routes by which the metals are passed on through the ecosystem (ibid.).

Conclusion

The question is what import can this knowledge have in the context of functional soil sanitation? On the one hand it is good to know that there are specific species that with a little help and soil heterogeneity will still grow. Phytoremediation is one possible method of treating the soil, resulting in a specific use being possible again in time. When it comes to be deemed acceptable for use of a site, e.g. a green amenity, to be impeded for a number of years, it is conceivable that a higher concentration of heavy metals may be permitted temporarily provided the contamination is reduced in the course of time by means of a form of phytoremediation. At the same time, a maximum needs to be fixed in connection with the risk of secondary poisoning. All this requires additional research.

4.3 ECOLOGICAL FUNCTIONS OF THE SOIL USE CATEGORIES

The problem of secondary poisoning occurs mainly when the ecological or natural function of the soil use categories is considered. Chapters 2 and 3 contain an ecological substantiation of use-specific soil characterisation from an anthropocentric point of view, as is evidenced *inter alia* by the fact that biodiversity is considered to be a functional property. As already indicated in § 2.1, in certain cases this conflicts with the actual situation. This section therefore looks on the one hand at how use-specific soil characterisation is connected with themes like ecological green management

(Boer & Schils, 1993) and the intertwining of beneficial functions and natural functions (Vissers *et al.*, 1995). Each soil use category is considered individually.

Table 4.4 Various types of synanthropy (after Faber, 1997).

Obligatory synanthropy (eusynanthropy)	Species that only occur and reproduce within human settlements. Many of these species are cosmopolitan. For example, various spiders, woodlice and insects, swifts, urban pigeons, brown rats and a number of species of bat.
Optional synanthropy	Species that enjoy optimum chances of survival in human settlements. Populations occur from which immigration can take place. For example, common rough woodlouse, honey bee and the mole cricket.
Permanent synanthropy	Species that spend there entire life cycle in human settlements.
Temporary synanthropy	Species that are found in human settlements at set times (for example, to overwinter) or in given conditions. No independent populations are formed in this case. E.g.: starlings, finches and great tits.
Partial synanthropy	Species that during a given phase of life (possibly even changing daily) belong to the urban biotic community

Eusynanthropic organisms as ecological parameter?

An initial aspect is the occurrence of synanthropic species in h uman settlements and the question of whether these organisms should be taken into account in the use-specific characterisation of soil quality. Faber (1997) distinguishes various forms of synanthropy (table 4.4) and stipulates the **eusynanthropic flora and fauna** as ecological parameter. Taking these species as a basis, assuming there is already sufficient knowledge about them, need not by definition signify a tightening up of soil quality requirements, as the species may also be relatively insensitive to the pollution in question. Also, even if a specific use can be allocated to a polluted soil, it does not mean that the eusynanthropic species in question can no longer go anywhere else. As there is a great lack of information, it is not feasible for the time being to involve these species in use-specific soil quality characterisation.

Table 4.5 Selected higher species of animal as species of special concern in the soil ecology of Amsterdam (Tenner *et al.*, 1997).

in soil:		mole and/or shrew or mouse-like rodents in general
on the soil:	herbivores	hare, rabbit, northern vole, squirrel, seed- and berry-eating birds (finches, sparrows, gold finches, green finches).
	omni- / carnivores	hedgehogs, all reptiles present and amphibians, birds of the thrush family (blackbird, song thrush, etc.), small insectivores (pipits, wagtails, wrens, etc.), pheasants, partridges.
	top predators	mustelines, foxes, domestic cats, birds of prey and owls.

Synanthropy and secondary poisoning

Where the ecological function of the soil use category is concerned it is certainly important that attention be focused on synanthropy. In a study of the ecological aspects of soil sanitation policy in Amsterdam, the IVM (Institute for Environmental Issues; Tenner *et al.*, 1997) selected various higher animals as species of special concern in the study (table 4.5). The departure point was that it should remain possible to find in Amsterdam species currently found there (and which thus exhibit synanthropy). This also focused attention on secondary poisoning as it necessitates the continued existence of the food of top predators. At the same time this food must not contain high (harmful) concentrations of contaminants. The result of the inventory is given in brief in table 4.6 and shows not only how limited conclusions could be drawn but also how careful conclusions are concerning poisoning. The table shows that if a copper content of 200 mg/kg is permitted for verges and waste ground (see table 3.9), there is a definite risk of negative impacts on herbivores and top predators.

Table 4.6 The effects of substances and groups of substances on higher species of animal for the Amsterdam situation in the area of concentration between the ecotoxicological (eco IV) and the human toxicological intervention value (human IV) (Tenner *et al.*, 1997). + + = fairly high degree of certainty of effects on the organism in question; - - = probably no effects; + ? = not enough data available, effects likely; - ? = not enough data, probably no effects; ? = not enough data available; [a]: for lead shot; [b]: effects probably marginal.

Substance (group)	Eco IV - human IV	In the soil	Herbivores	Omni-/ carni-vores	Birds	Top predators
Arsenic	40 - 678	+?[b]	?	+?[b]	?	?
Copper	190 - 31,300	?	++	?	+?	++
Mercury	10 -197	?	?	?	++	++
Lead	290 - 530	+?	--	+?	-/++[a]	-?
Zinc	720 - 56,500	+?	?	?	?	?
PAHs	40 - 11,800	-?	-?	-?	-?	-?
PCBs	-	?	?	+?	++	++
Mineral oil	< 500	-?	-?	-?	-?	-?

Urban green structures

In various policy plans (including the Fourth Policy Document on Physical Planning, the Third Policy Document on Water Management and the Nature Policy Plan) much emphasis is placed on creating more space for nature, including within the built environment (Boer & Schils, 1993). Consequently, the **ecological management of green amenities** is increasingly being given a place in the green policy of local and provincial authorities. By analogy with the national network of important ecosystems (EHS), a network of ecosystems is also being sought in and around towns (inter alia Bergakker & Lampert, 1994; De Bruin *et al.*, 1995; Denters, 1995). These systems include not only green facilities such as parks and recreational areas but also allotment complexes and other gardens (Vissers *et al.*, 1995; Denters, 1995), verges and waste ground (Niemeijer and Verburg, 1995a; Denters, 1995; Van der Weijden and Schippers, 1996), and industrial sites (TCB, 1993; Vissers *et al.*, 1995; Denters, 1995). Plenty of examples of these are available. The animal species listed in table 4.5, for example, can be found in principle in each of the spatial elements referred to and that ultimately is the aim (see also Bergakker & Lampert, 1994).

Gardens and allotments

It can be stated simply that the soil quality requirements set in chapter 3 change little in the existing ecological function of gardens and allotments. However, if in the end critical values are chosen which are closer to the values adhered to by the VNG (see §4.5) - for the original category of dwellings with gardens, for example - issues such as secondary poisoning could start to play a part. Table 4.6 gives an indication of the risk of an effect for lead, mercury, copper, arsenic and zinc. If a site containing polluted or slightly polluted soil is planned for dwellings with gardens or for allotments, it is advisable not to attach an ecological function to it at the same time.

Verges and waste ground

The verges and waste ground category of soil use presents the most opportunities for lowering soil quality requirements (see chapter 3). However, it cannot be reconciled with an ecological function. It is exactly in these verges that many birds of prey forage for food, and it is well known, for example, that the mole (*Talpa europea*) and the common shrew (*Sorex araneus*) can accumulate very high levels of heavy metals from the soil (in: Van Straalen *et al.*, 1994). A verge on contaminated soil cannot therefore constitute part of a network of ecosystems, although it is impossible to establish with certainty that at values resulting from chapter 3 negative effects will actually occur as a result of secondary poisoning. Nonetheless, it is advisable to take steps to separate a verge or waste ground on contaminated soil from green zones that are being managed on an ecological basis. If the verges are being grazed, it is necessary to impose stricter requirements for copper.

Public parks & gardens and recreational areas

It is even more natural that the public parks & gardens and recreational areas category should be included in urban ecosystems. From this point of view, it is not advisable to relax soil quality requirements to any degree. A surface layer could provide a solution in this case.

4.4 THE QUALITY OF GREEN WASTE

Permitting a lower soil quality has consequences not only for the ecology in and on that soil but also for its use. The residual products of soil use are also affected. This section looks briefly at the relationship between use-specific soil characterisation and the quality and re-use of green waste.

Table 4.7 The maximum contents of heavy metals (mg/kg dry matter) in compost, very clean compost and cattle feed. [a] according to the Decree governing the quality and use of other organic fertilisers (BOOM, as amended per 1/1/1995) (in: Sluijsmans, 1995); [b] calculated from animal feed legislation in the Netherlands, part 1. (In: Niemeijer and Verburg, 1995a).

Substance	Compost [a]	Very clean compost [a]	Cattle [b]
Organic content	> 20 %	> 20 %	
Arsenic	15	5	4.4
Cadmium	1	0.7	1.1
Chromium	50	50	-
Copper	60	25	37 (sheep)/39 (cattle)
Lead	100	65	44
Nickel	20	10	-
Zinc	200	75	275

In view of the lower requirements governing the quality of soil in verges and on waste ground, the chemical quality of green waste (and compost) plays a part in this soil use category especially. Verge mowings are in principle suitable as cattle feed if they meet the standards set by animal legislation. For intensive cattle farming its nutritional value is too low, however. Another possible use of green waste is as green manure, by ploughing into arable land. The most structural method of processing mowings is to compost them (Sluijsmans, 1995). These possibilities will be curbed if soil quality is reduced, resulting in higher contents of heavy metals in the verge mowings (see table 4.7 for maximums). However, this need not impose a limitation on opting for flexible soil quality requirements. It can be seen as an acceptable impediment to use, which means that care must be taken in dealing with the material. Needless to say, if the use-specific critical values employed by the VNG are used, the processing of the green waste requires close control. The maximum contents of heavy metals in compost, very clean compost and cattle feed can in that case easily be exceeded.

4.5 FUNCTION-SPECIFIC SOIL SANITATION IN OTHER COUNTRIES

Function-specific soil sanitation is already part of soil policy in various countries. These include the United Kingdom, Germany and Canada. This section discusses briefly the function-specific aspect of the soil policy of these countries (largely based on Visser, 1993).

The United Kingdom

The policy of the United Kingdom is based on soil quality having to be fit for its present or immediate use in the future and that sanitation should not be focused on all conceivable uses of the soil in the future (**"fitness for purpose" principle**). It is therefore a functional approach. At the same time attention is paid to additional cleanup methods which can achieve a greater fitness for purpose at marginal cost. The guidelines for assessing and restoring contaminated sites are based on what are known as "trigger concentrations" for specific contaminants and the intended use of the site. A distinction is drawn between two trigger values: a limit or threshold value and an action value, corresponding with the target value and intervention value in Dutch policy (compare Tenner *et al.*, 1997). The values are given for different forms of land use and two forms of pollution, namely (1) inorganic pollution: (a) private gardens and plots and (b) parks, recreational areas and public spaces; and (2) pollution caused by former coal furnaces: (a) private gardens and plots, (b) areas of natural beauty, (c) buildings and (d) paved surfaces. Table 4.8 gives the limit and action values for the heavy metals that are also involved in this study (substances of special concern). Concrete numbers are given for limit values, but these are often lacking for the action values. There are no values for groundwater either. In general, the values are deemed to be site-specific and are based on expert judgement. In addition to human toxicity - ingestion or inhalation of soil, skin contact and consumption of contaminated plants - phytotoxicity, chemical threat from building materials and the risk of fire and explosion are also taken into account. The values have no status or formal background.

Table 4.8 Trigger concentrations (mg/kg dry matter) for soil quality characterisation in the United Kingdom. The values are approximate and / or provisional. Only the substances relevant to this study are given. TBE = to be established

		Trigger concentrations	
Substance	Soil function	Limit value	Action value
Arsenic	private gardens and plots	10	TBE
	parks, recreational areas and open spaces	40	TBE
Cadmium	private gardens and plots	3	
	parks, recreational areas and open spaces	15	TBE
Chromium (III + VI)	private gardens and plots	600	TBE
	parks, recreational areas and open spaces	1,000	TBE
Copper	any use with plant growth	130	TBE
Lead	private gardens and plots	500	TBE
	parks, recreational areas and open spaces	2,000	TBE
Nickel	any use with plant growth	70	TBE
Zinc	any use with plant growth	300	TBE

Germany

In Germany as well the approach to soil quality characterisation and sanitation is ultimately fairly ad hoc, although **reference values** have been established for the environmental impact assessment of heavy metals and PAHs based on 526 reference sites, which has resulted in two categories of concrete values. Reference level 1 points to a multifunctional soil. Between level 1 and level 2 soil is suitable for most agricultural purposes (see table 4.9.A). Values that exceed the second level point to site-specific limitations on soil use. Where sanitation is concerned, the goals are related to present or future land use. The basic idea for establishing the associated values is securing soil functions in accordance with a concept for soil use in which public health takes priority; in any event human toxicology is the guideline. Actually establishing values is further left to the individual states in Germany. Table 4.10.B gives an example of provisional trigger concentrations for the state of Hamburg.

Table 4.9. A. Reference values I and II (mg/kg dry matter) for soil quality characterisation in Germany. Only the substances relevant to this study are given. B. Provisional trigger concentrations for heavy metals in soil for various exposure routes in the state of Hamburg. [a]: for sandy soils with a normal humus content and pH between weakly acid and weakly basic.

A.			**B.**		
Substance	Multifunc-tionality (I)	Suitable for agri-cultural use (II)	Farming of food crops [a]	Public health	
				long term	acute
Arsenic	20	40	50	100	100
Cadmium	0.6	1.5	2	40	40
Chromium	-	100	100	200	500
Copper	40	60	100	500[b]	3,000
Lead	50	100	300	500	3,000
Nickel	-	50	100	300	4,000
Zinc	120	200	500	2,000	2,000

Also interesting is Eikmann and Kloke's system of characterisation from 1991. It is the so-called **'three sectors system'** and aims to determine the possible uses of contaminated soil. The system gives values for eleven heavy metals and three organic substances for a wide range of soil uses. It divides the pollution into three sectors, namely: protect, tolerate and clean up. In itself this is not startling, although the relationship it establishes with soil use is. Figure 4.1 shows the model for the system for soil use in the city. Table 4.10 gives the appurtenant values for the substances that are relevant to this study.

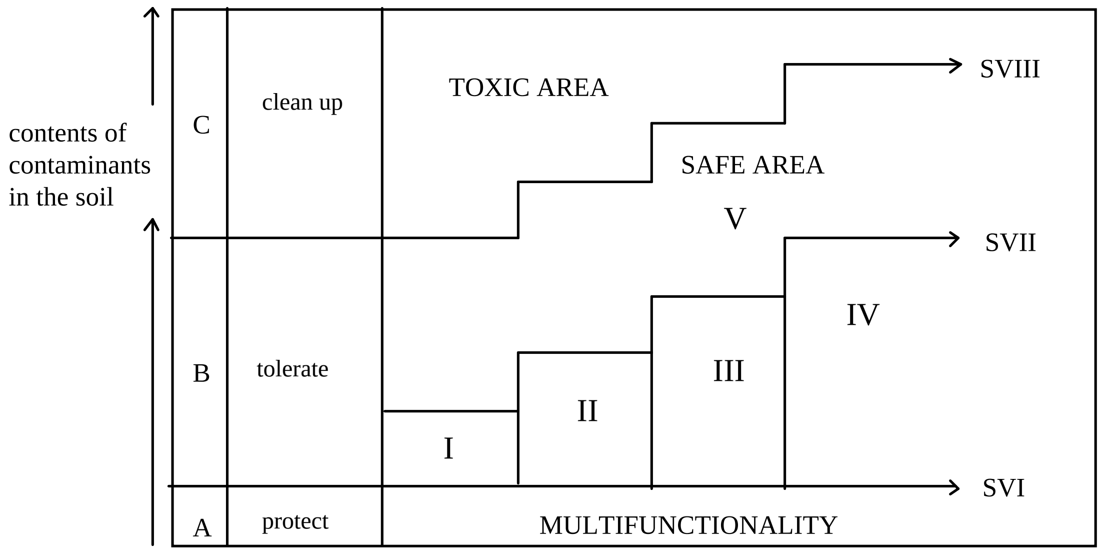

Figure 4.1 The three sectors system of Eikmann and Kloke. A model for the possible uses of polluted soil in urban areas. SV = soil value (see table 4.10); I = children's playgrounds; II = private gardens,

plots, sports and play areas; III = parks and recreational areas; IV = industrial sites; V = site- or function-specific possibilities for use.

A proposal was recently made to produce uniform, scientifically substantiated use-specific values for the whole of Germany. These are what are known as ‘soil screening values’; above these values detailed site-specific surveys are required. For the time being the values are based only on human toxicological information (Caracas, 1997; see table 4.11).

Table 4.10 The three vectors system of Eikmann and Kloke translated into reference values for the heavy metals relevant to this study. SV = soil value (see figure 4.1).

Soil use		Arsenic	Cadmium	Chromium	Copper	Lead	Nickel
multifunctionality	SV I	20	1	50	50	100	40
children’s playgrounds	SV II	20	2	50	50	200	40
	SV III	50	10	250	250	1,000	200
private gardens and plots	SV II	40	2	100	50	300	80
	SVIII	80	5	350	200	1,000	200
sports and play areas	SV II	35	2	150	100	200	100
	SV III	90	5	350	300	1,000	250
parks and recreational	SV II	40	4	150	200	500	100
areas, non-paved and low vegetation	SV III	80	15	600	600	2,000	250
heavy and light	SV II	50	10	200	300	1,000	200
industrial areas, non-covered	SV III	150	20	800	1,000	2,000	500
heavy and light	SV II	50	10	200	500	1,000	200
industrial, covered and overgrown	SV III	200	20	800	2,000	2,000	500
agricultural area,	SV II	40	2	200	50	500	100
orchards, gardens	SV III	50	5	500	200	1,000	200
non-agricultural	SV II	40	5	200	50	1,000	100
ecosystems	SV III	60	10	500	200	2,000	200

Table 4.11 Proposals for uniform ‘soil screening levels’ in Germany (in mg/kg; see text).

Substance	Children’s playgrounds	Parks and recreational areas	Residential areas	Commerce/ industry
Arsenic	20	40	20	140
Cadmium	10	50	20	60
Chromium	200	1,000	400	1,000
Lead	200	1,000	400	2,000
Nickel	70	350	140	900

Canada

Canadian policy distinguishes three different soil use categories. Initially these were agriculture, residential areas/parkland and commerce/industry (Gaudet & Cureton, 1993). Interim restoration criteria were formulated for these three categories (table 4.12) in anticipation of a protocol which would enable criteria to be developed on a more scientific basis. Various proposals for these were made in a draft version of this protocol. It draws a distinction between human- and ecotoxicology and whichever produces the tightest standard is decisive. As regards the soil use categories, commerce is added to the second category (CCME, 1993). In addition, a careful search was made for ecological receptors and exposure routes per soil use category. Ultimately it was stated that **agricultural lands** could not contain any contaminants that imposed restrictions on the maintenance of microbial and non-vertebrate populations, crop farming and stockbreeding for consumption purposes. Protection must also be offered to local and passing game and to native flora. The regulations governing maintenance of microbial and non-vertebrate populations, the growth of ornamental and native plants as well as local and passing game also apply to **residential areas and parklands**. In the case of **industrial land use** the same criteria apply as to the previous category of use, only the level of protection has been set lower (ibid.). It is not known to what extent the procedure developed has already resulted in use-specific intervention criteria per heavy metal.

Table 4.12 Interim restoration criteria for the soil in Canada (CCME, 1993). Only the values for the heavy metals relevant to this study have been incorporated.

Substance	Agricultural area	Residential area / parklands	Commerce / industry
Arsenic	20	30	50
Cadmium	3	5	20
Chromium	750	250	800
Copper	150	100	500
Lead	375	500	1,000
Nickel	150	100	500
Zinc	600	500	1,500

Conclusion

The divisions into soil use categories employed by the United Kingdom, Germany (the Eikmann and Kloke system) and Canada exhibit a clear overlap with the division in this study. The gardens and allotments category corresponds with the ‘private gardens and plots’ category of Eikmann and Kloke and the United Kingdom, and the ‘farming food crops’ category established by the state of Hamburg. The verges and waste ground category compares with the ‘industrial areas’ in the Eikmann and Kloke system and with the interim restoration criterion ‘commerce / industry’ of Canada. Finally, the public parks & gardens and recreational areas category relates to the categories for parks and recreational areas of Canada, the United Kingdom and the Eikmann and Kloke system. Information on the way in which the foreign divisions were made is not available unfortunately. Information on the way the ultimate values were fixed is also very limited. However, it is possible to establish that public health constitutes a primary parameter in fixing all kinds of critical values for soil sanitation. Canada in particular also takes into account ecological aspects. It is striking that in spite of the background against which the values have been fixed, they are virtually incomparable with the VNG values resulting from CSOIL (see table 4.13). When categories that are comparable in any way are compared, the VNG values are in every case by far the highest. The indicative figures from table 3.9 on the other hand correspond reasonably well with the foreign values.

Table 4.13 The VNG critical values (mg/kg dry matter) according to CSOIL for four forms of soil use (Moet, 1995). The values apply to an organic matter content of 2, 4, 6, 10 and 20%.

Substance	Dwellings with vegetable gardens	Dwellings with gardens	Dwellings without gardens, traffic, social/cultural, work	Recreation, parks & gardens
Arsenic	150	680	6,700	1,400
Cadmium	4.2	35	3,200	660
Chromium	620	2,200	16,000	3,300
Copper	2,600	16,000	100,000	92,000
Lead	330	1,500	12,000	2,400
Nickel	1,100	6,600	100,000	33,000
Zinc	7,100	56,000	100,000	100,000

5 SUMMARY AND CONCLUSIONS

It was recently decided in soil sanitation policy to opt for a functional approach, focusing on what the soil can still be used for under what conditions. It is therefore a question of the minimum soil quality requirements for achieving the desired use. It is advisable to involve relevant ecological parameters at an early stage in the functional approach because otherwise there is a danger of no attention being paid to aspects of soil quality other than human exposure to pollutants. The aim of the exploratory research described in this report was to provide an initial impetus for use-specific minimum soil quality requirements for urban areas as seen from the ecological angle.

In use-specific ecological soil characterisation within urban areas a division into three soil use categories is most advisable, namely (1) gardens and allotments, (2) verges and waste ground, and (3) public parks & gardens and recreational areas. In this research ecological parameters (species of special concern, key species and processes) and soil quality requirements were linked to these categories from an anthropocentric point of view. If a full ecological function is also allocated to soil use categories, the logic of abandoning the pursuit of restoring multifunctionality is diminished because it is necessary to take into account the intrinsic value of biodiversity (including synanthropy / eusynanthropy), ecological infrastructure and aspects of secondary poisoning in ecosystems. The degree to which the ecological function can be included in the soil characterisation can be made per site, if necessary.

Although plants can still grow on soils heavily contaminated with metals, the forms of use impose specific requirements on the types of plant that should be able to grow. Species of special concern are plant species which logically occur with a given soil use. They therefore provide departure points for the use-specific differentiation of soil quality requirements. This was considered in the research from the point of view of metal contamination. The gardens and allotments category imposes the highest requirements on plant growth: a range of plant species needs to be able to grow in these gardens, among them ornamental and crop plants, as well as species of grass and native herbs. In verges and on waste ground on the other hand a number of relatively metal-insensitive species of grass, tree and heather will suffice. If not being able to re-use verge mowings is acceptable as an impediment to use, relatively high metal contents can be permitted from the point of view of plant growth. This category of use may also present opportunities for applying phytoremediation techniques. The public parks & gardens and recreational areas category falls between the other two categories as regards soil quality requirements for plant growth seeing that a number of generally used species of tree and grass are fairly sensitive to metals.

In addition to plant growth, life support functions can be seen as an ecological parameter for soils in general within the overall function of decomposition. It is obvious that some forms of use impose different requirements with regard to an aspect such as nutrient availability for plants than do others; in this respect allotments impose the highest requirements and verges and waste ground the lowest. In relevant subprocesses relating to life support functions, both species of soil fauna and micro-organisms are involved. By making a functional cross section of the soil ecosystem and applying criteria like functional redundancy within subprocesses, ecological relevance and the availability of information, earthworms were selected as key group and nitrification as key process. In this way the most relevant routes of exposure to substances in the soil were covered. It is advisable in use-specific soil characterisation to devote attention also to critical symbiotic interactions (mycorrhizae, nitrogen fixation), although their relationship with metal contamination is highly complex.

Although deducing numerical values for minimum soil quality was not a specific aim of the research, an attempt was made to produce ranges for arsenic, cadmium, copper, lead, nickel and zinc using metal toxicity data relating to earthworms and nitrification and the available phytotoxicity data. Although rough figures, they provide an indication that an approach from the human toxicological angle proves inadequate where use-specific soil characterisation is concerned. The present research shows, among other things, that the VNG metal values for minimum soil quality obtained from CSOIL are absolutely inadequate from an ecological point of view. This applies in particular to forms of soil use in which the risk of human exposure is marginal. Plant growth in particular imposes much stricter requirements on maximum metal contents. Phytotoxicity consequently deserves a much more prominent place in soil characterisation than is the case at present. For the rest, the VNG values can be said to be very high even in comparison with function-specific numerical values from other countries, even though they are based only on human toxicological information.

Further - numerical - ecological substantiation of use-specific soil quality requirements is desirable, whereby it is advisable to look at total contents as well as the bioavailable part of metals. This will also do more justice to the distinction made in policy between the approach to mobile and relatively immobile contaminants. A set of specific bioassays could also be involved in use-specific soil characterisation. A number of selected species of special concern (for example, grasses like *Lolium perenne*, *Festuca rubra* and *Agrostis capillaris*), the various categories of earthworm ('epigeics', 'endogeics' and 'anecics') and nitrification (in particular chemo-autotrophic) present useful possibilities for this purpose.

As regards using surface layers, regulations governing development for surface layers are not advisable per soil use category. This type of use deserves a more site-specific approach, taking into account among other things the aspect of plant growth in relation to the thickness of the clean top layer. The extremes are formed by deep rooting species of tree (1.5 to 3 m) and paved surfaces (no surface layer necessary). Impeding root growth presents an option for reducing the minimum thickness of the surface layer.

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