

**Methodological Framework for Assessment and
Mapping of the Vulnerability of Soils to Diffuse
Pollution at a Continental Level
(SOVEUR Project)**

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SUMMARY

The soil is an important component of terrestrial (agro-)ecosystems. Certain soils are capable of absorbing, retaining and re-cycling contaminants better than others. Potentially toxic and persistent chemicals thought to be held firmly in the soil can be re-mobilized if gradual changes in the capacity controlling factors occur; these changes can be induced by several trigger systems, including human-induced acidification, land use changes and climate change.

A framework for identification of soils most at risk from specified types of delayed-pollution, and delimitation of their geographical distribution using an Environmental Information System, is outlined for future refinement and application in a Central and Eastern Europe wide assessment of soil vulnerability to pollution (SOVEUR project).

A methodology, for application at a continental scale (1:2.5 million), is presented with special reference to pollution by diffusely distributed heavy metals. The overall approach, however, can be adapted to suit other pollutants as well depending on the availability and accessibility of necessary auxiliary data. The methodology will allow generation of maps, and tabular output, which can serve to increase general awareness of soil vulnerability to specific types of pollution. When these maps are combined with combined with auxiliary maps of pollutant loads in a GIS, problem areas can be identified for subsequent studies at regional and national levels.

ACKNOWLEDGEMENTS

Project GCP/RER/007/NET on Mapping of Soil and Terrain Vulnerability in Central and Eastern Europe, SOVEUR, was signed between the Food and Agriculture Organization of the United Nations (FAO) and the Government of the Netherlands, within the framework of the FAO/Netherlands Government Programme. In view of the specific nature of the services to be rendered, the Project activities were implemented under a Contractual Service Agreement with the International Soil Reference and Information Centre (ISRIC). It was carried out in close collaboration with specialists from soil survey institutes in Belarus, Bulgaria, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Moldova, Poland, Romania, the Russia Federation (West of the Urals), Slovak Republic and the Ukraine, who collated the various national data sets using uniform guidelines.

1 INTRODUCTION

1.1 Background

As the 21st century approaches soil degradation and environmental pollution are no longer restricted to isolated incidents and locations; together with global climate change and ozone depletion they are now of international concern (Houghton *et al.*, 1990). The long history of agriculture and settlement, the intensive mining of minerals and lignite, and the widespread (mis)management of soils have caused a complex pattern of soil degradation and pollution all over the world (Barth and L'Hermite, 1987; Nriagu, 1990; Oldeman *et al.*, 1991; Heinisch *et al.*, 1994). This aspect has been stressed also in Agenda 21, accepted by UNCED in 1992.

Historical differences in development policies between East and Western Europe have led to differing environmental strategies. The social and economic structure of many countries in Central and Eastern Europe have changed drastically during the last 40 years, through rapid expansion of heavy industries, the energy and transport sector, increased urbanization, and intensification of agricultural production. Many of these rapid changes occurred without due consideration for possible, adverse environmental effects. Thus, irrespective of the ongoing international efforts to reduce emissions at the source of pollution, there is presently a vast legacy of 'accumulated loads' of pollutants in soils and sediments (Carter and Turner, 1996). Many of these chemicals may get remobilised in a biotoxic form following climatic and land use changes (Stigliani, 1988; Ter Meulen-Smidt *et al.*, 1992). Although emissions and deposition of mercury have been decreasing recently (Pirrone *et al.*, 1996), mercury levels in freshwater fish are still increasing in Scandinavia; mercury accumulated in the soil is now being mobilized by human-induced acidification. Earlier, this acidification led to massive 'die-back' of forests subsequent to: leaching of plant nutrients such as Ca, Mg and K; release of poisonous metals, such as aluminium ions (i.e., a 'chain-of-pollution-events'). With increased acidification, Cd, Zn, Mn and Ni will become more readily accessible to plants. Phosphates, on the contrary, may get bound by Al, making them less easily available. Important nutrients such as Selenium, Boron and Molybdenum also will become less easily available to plants when the soil is acidified.

Environmental policies regarding air and water quality were well developed before soil protection became an important issue, possibly because effects of soil pollution are far less conspicuous. In Europe, the need to protect the soil against pollution was first proclaimed in 1972 in the European Soil Charter. Van Lynden's (1995a) work was part of an initiative of the Council of Europe to develop a European Soil Protection Policy, illustrating the need for new legal instruments for sustainable soil use and protection within Europe. Present concern about adverse effects of point-source and non-point source pollution is related to negative effects on crop quality and quantity, and ultimately on human health and biodiversity. The problems include critical pollution associated with the excessive application of nutrients, the heavy metal content of fertilizers and manure, human-induced acidification, loss of organic matter and decrease in soil biological diversity, and physical degradation and erosion by water and wind.

Estimates of areas of land affected by specific types of human-induced soil degradation in Europe have been presented by Oldeman *et al.* (1991) and revised by Van Lynden (1995b). Ter Meulen-Smidt (1995) reviewed regional differences in inputs and distribution of contaminants in Europe, describing broad regional trends in chemical loads. Csikos (1994) reviewed important 'hot-spots' of pollution in selected Central and Eastern European countries. These studies show that regional and continental estimates of the actual extent of degraded and polluted land, and of areas at risk, remain open to improvement also in view of differences in procedures and threshold levels used (Wieringa, 1995; Van Lynden, 1995a).

Some countries in Europe have started a systematic recording, monitoring and clean-up programme of contaminated sites based on established clean-up methodologies, the implementation of which is often constrained by financial, legal and technical factors. Better data on the extent of soil degradation and pollution, and establishment of monitoring networks to assess the effectiveness of measures that have been put in place, are seen as a pre-requisite to any further coordinated approach to soil protection in the whole of Europe (Wieringa, 1995; Várallyay, 1994; Van Lynden, 1995a). It is in this overall context that FAO and ISRIC initiated the project on Assessment of Soil and Terrain Vulnerability in Central and Eastern Europe (SOVEUR), within the Framework of the FAO/Netherlands Government Cooperative Programme. The project is a sequel to initial discussions during a workshop organized by ISRIC in the framework of the Chemical Time Bombs (CTB) project (Batjes and Bridges, 1991).

1.2 Objectives

The SOVEUR project calls for the development of an environmental information (EIS) system for 13 countries in Central and Eastern Europe (Figure 1). This will be done in close collaboration with soil survey institutes in Belarus, Bulgaria, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Moldova, Poland, Romania, the Russian Federation up to the Urals, Slovak Republic and the Ukraine. The EIS is being developed for a georeferenced assessment of the status of human-induced land degradation, with particular attention to issues of soil pollution, and of soil vulnerability to delayed-pollution at a nominal scale of 1:2.5 million.

The project activities include:

- (1) Refinement of methodological guidelines for the compilation of a soils and terrain digital database for the 13 participating countries (Van Engelen and Wen, 1995; Batjes and Van Engelen, 1997).
- (2) Refinement of methodological guidelines for assessment of the status of human-induced land degradation, with special focus on soil pollution status (Van Lynden, 1997)
- (3) Development of methodological guidelines for assessment of the vulnerability of soils to selected categories of pollutants (*this report*).
- (4) Application of the methodological guidelines, by country, in order to create georeferenced databases on:
 - soil and terrain units;

- soil degradation and pollution status;
- which will subsequently be used, in combination with auxiliary data sources, to:
- assess the relative soil vulnerability (to specified pollutants);
 - determine areas considered at risk from re-mobilization of specific contaminants.

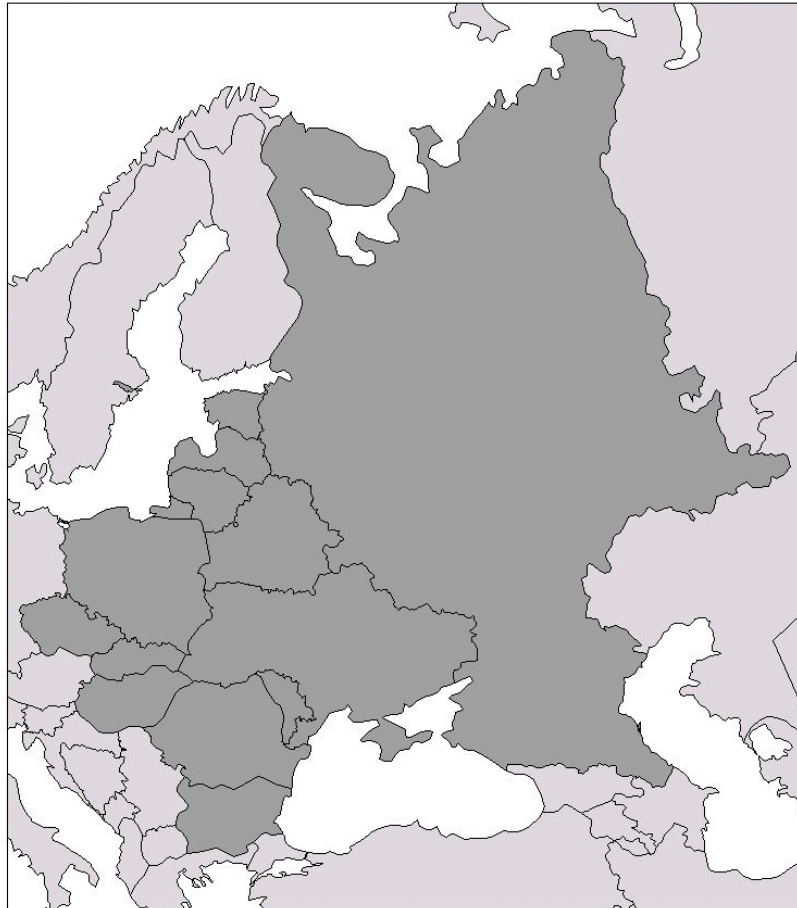


Figure 1. Location of SOVEUR project area in Central and Eastern Europe

The current report proposes methodological guidelines for assessment of soil vulnerability to non-point sources of pollution for application at a continental level. It consists of 4 Chapters. Concepts and definitions of relevance for the assessment of soil vulnerability are reviewed in Chapter 2. Chapter 3 outlines the proposed methodology, including: general procedure (3.1); pollution scenarios (3.2); contaminant accumulation in soils (3.3); relative sensitivity of soils to defined trigger systems or environmental changes (3.4); relative vulnerability of soils to contaminant re-mobilization (3.5); assessment of pollution status as accumulated loads (3.6); identification of areas at risk (3.7); and, mapping approaches, in relation to overall data availability, using GIS (3.8). Conclusions are drawn in Chapter 4.

2 CONCEPTS AND DEFINITIONS

2.1 Sources, processes, pathways and targets of soil contamination

Pollution scenario: Pressure-state-impact-response framework

Pollution scenario identification is the first and perhaps most important step in developing guidelines for soil vulnerability mapping. Figure 2 illustrates how changes in the environmental and societal (sensu politics, economics, demography) setting impact on human activities, and how responses to these impacts influence the environment and society. Under similar conditions of climate, soils and demographic pressures, different impacts and responses can be obtained depending on economic and political circumstances. As illustrated by Figure 2, and McNeill's (1994) review, there remains much uncertainty in the possible impacts of political, technological and socio-economic developments in Europe on land use change during the next decades. An additional source of uncertainty is associated with predicted global warming (Houghton et al., 1990) and possible impacts of increased land degradation (Bouma *et al.*, 1997). Each pollution scenario should consider the *source* and *type* of the contaminant, the *processes* that affect its mobility and toxicity, the *pathways* through which a contaminant and its derivatives are re-distributed in the environment, and finally the *targets* at risk from these compounds (Sheppard *et al.*, 1992; Klijn, 1991; Desaulles, 1991).

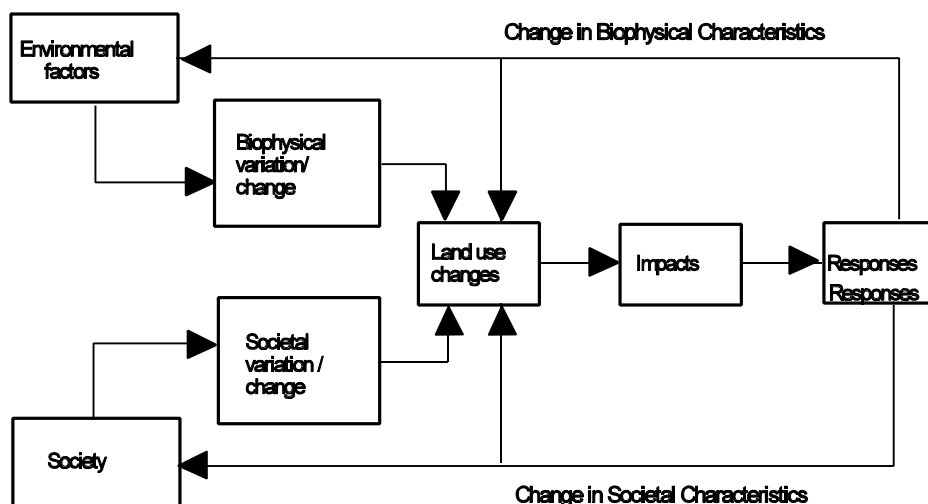


Figure 2. Schematic representation of interactive impact model, with feedbacks (After Kates, 1985)

Contamination and pollution

A widely used definition of *pollution* is 'the introduction by man into the environment of substances or energy liable to cause hazards to human health, harm to living resources and ecological systems, damage to structures or amenity, or interference with legitimate uses of the environment' (Holdgate, 1979). Often, a scientific distinction is made between contaminated and polluted soils. *Contamination* is used for situations where a substance is present in the environment, without causing any obvious harm, while *pollution* is reserved for cases where harmful effects are apparent. In addition, the definition of contaminated soil has the corollary that to be hazardous, pollutant substance must occur in concentrations and quantities above geochemical background levels normally experienced in soils (Bridges, 1987, 1989a).

Sources and types of pollutants

Basically, there are two types of pollutants. *Primary pollutants* cause harmful effects in the form in which they enter the environment, while *secondary pollutants* are formed upon chemical transformation of generally less dangerous precursors after their release in the environment (Alloway and Ayres, 1993).

Alloway and Ayres (1993) reviewed the major source of pollutants, including agricultural sources, electricity generation, derelict gas work sites, metalliferous mining and smelting, metallurgical industries, chemical and electronic industries, general urban and industrial sources, waste disposal, transport, incidental sources (e.g. nuclear accidents and leakage from storage tanks) and long range atmospheric transport, in relation to the environmental media with which they are transported or in which they reside.

Six general kinds of contaminants commonly reach the soil: pesticides of varying chemical composition and reactivity; inorganic pollutants, including cationic (e.g. Pb, Zn, Cu, Cd) heavy metals and other toxic substances such as Fluorine and Arsenic; organic wastes, such as those arising from food processing, feedlots and municipal and industrial activities; salts; radionuclides; and acid rain (Brady, 1990). The main persistent-organic-pollutants (POP) of global concern are: DDT; dieldrin; chlordane; hexachlorobenzene (HCB); toxaphene; furans; aldrin; endrin; heptachlor; mirex; polychlorinated biphenyls (PCBs) and dioxins. Of special concern on a regional scale are emissions of persistent organic pollutants from incinerators, their subsequent binding to atmospheric particulates, and subsequent wet or dry deposition on the soil. Notoriously dangerous substances considered of primary concern are shown on lists, such as the EC Dangerous Substances Directive (EEC, 1981). These substances may occur as solids, gases or liquids, and their source may be either point (local) or non-point (diffuse) (Bridges, 1987).

Point-source pollution

Pollutants emitted from a source are dispersed in the environment in either air, water or the soil, with pollutant transport processes often being much slower in the soil where pollutants tend to accumulate as if in (temporary) sinks. The characteristic element of a point-source is the high local level of toxicity, or corrosivity, that can result in acute damage to plants, animals and inanimate structures. Point pollution by spillage, leakage and poor handling of materials generally can only be mapped adequately at a local scale. Distance from the pollution source (Table 1) has a great effect on the content of (bio-)available forms of heavy metals such as Pb, Cu, Zn, as well as total sulphur, and on the physico-chemical properties and organic compounds of soils (Kowalinski *et al.*, 1989; Lacatusu *et al.*, 1979; Shllaku and Landner, 1995). Factors affecting the proportion of local to long-range emissions and depositions include smokestack height, the velocity of gases and particulate matter leaving the stack, the meteorological conditions, and the particle size of the emitted pollutants (Rautengarten, 1993). In the Czech Republic, soil research by Kozák *et al.* (1995) showed point-contamination to be a bigger problem than diffuse-contamination. Similarly, in Hungary, polluted areas cover 'only' one tenth of the country, but this affects 40% of the total population because of the high population density around industrial centres and major urban areas (Molnár *et al.*, 1995).

Table 1. Effect of distance from pollution source on Cd, Pb and Zn content in soils located around some non-ferrous metallurgical factories near Bucharest

| Element | Heavy metal content (ppm) as a function of distance (m) from source | | | Natural content in the soil |
|---------|---|-----------|----------|-----------------------------|
| | < 200 | 200 - 700 | > 700 | |
| Cu | 60 - 188 | 31 - 55 | 24 - 45 | 20 |
| Pb | 93 - 975 | 50 - 160 | 33 - 112 | 12 |
| Zn | 140 - 500 | 90 - 355 | 70 - 150 | 50 - 70 |

Source: Rauta (1995)

Diffuse pollution

Diffuse pollution associated with atmospheric deposition and agricultural activities is ubiquitous and may spread across vast areas. An increasingly important diffuse source is runoff from paved urban areas to surface waters (Rautengarten, 1993). Generally, deposition of non-point-source pollutants occurs in low concentrations, but on the longer term critical loads for these elements can be transgressed (Corwin and Loague, 1996; Nriagu, 1990; de Vries *et al.*, 1993). Being of a regional or transboundary nature, the effects of non-point-source pollutants on soil quality can best be mapped at a regional or continental level (Chadwick and Kuylenstierna, 1990; Glazovskaya, 1991; Prieler *et al.*, 1996).

Critical loads in relation to targets

A *critical load* can be defined as the largest input of a particular contaminant that an ecosystem can tolerate without serious damage to its ecological functions (Nilsson and Grennfelt, 1988). The concept makes use of a biological indicator, a chemical criterion and a critical chemical value. So far, critical loads have been elaborated for acid deposition (acidification) and nitrogen addition (eutrophication) (Agren and Elvinson, 1997). Løkke *et al.* (1996) reviewed the adequacy of the current approach to calculating critical loads. They pointed to the need for a more dynamic approach which takes into account the temporal and spatial variability of ecosystems, rhizosphere complexity and the functional relationships between soil organisms. Critical loads will vary from ecosystem to ecosystem depending on a range of environmental site factors that determine the *sensitivity* of the ecosystem to a unit deposition rate of acid deposition (Chadwick and Kuylenstierna, 1990). As the acidification progresses, there may be a time when the critical load for a certain chemical is exceeded; the soil will then become a supplier rather than a sink of potentially harmful substances (Figure 3). The timing and severity of repercussions of such delayed, non-linear releases as infiltration, run-off or emissions is strongly determined by the nature and intensity of the controlling physico-chemical processes as well as the rate of decomposition and toxicity for biota of the chemicals involved (Connel, 1989; Van Straalen and Verkleij, 1991).

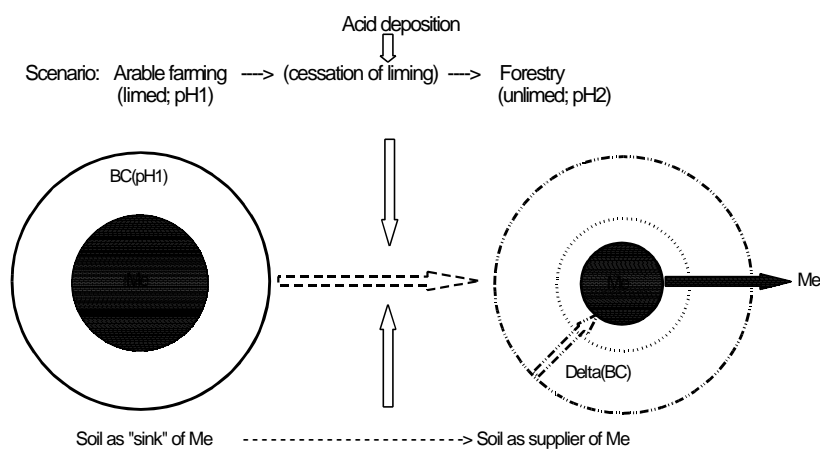


Figure 3. Schematic representation of the effect of cessation of liming, associated with a shift from arable cultivation to forestry, and acid deposition on the buffering capacity of the soil, and subsequent mobilization of heavy metals (Me). [$pH_1 = 6.5$; $pH_2 \ll pH_1$]; BC is buffering capacity with $BC(pH_1) > BC(pH_2)$; Modified after Stigliani (1988)]

Contaminant loading

The current status of trace element concentrations (i.e., *accumulated loads*) in natural or agricultural ecosystems is determined by parent material, natural flows such as soil erosion and flooding, and cumulated anthropogenic additions. Persistent organic pollutants and heavy metals tend to accumulate first in the biologically active topsoil, largely accounted

with the higher organic matter content (Mayer, 1993). Deposition loads on forests are generally larger than on arable land, being higher in coniferous than in deciduous forest (De Vries *et al.*, 1993). Under forest, there will be little mechanical mixing of the surface and subsurface layers unlike agricultural lands, and as a result high concentrations of (atmospheric) contaminants may accumulate in the surface horizons.

The average depth to which heavy metals migrate after deposition varies, penetrating the upper 30-40 cm for arable soils and the upper 15 cm for forest soils (Rauta *et al.*, 1995). As a result, these contaminants will be readily accessible to biota if mobilization/transformation occurs. High concentrations of especially Zn, Cd and Pb in plants, as observed in many localities in the Czech Republic, show that these metals can be concentrated (Kozák *et al.*, 1995). Strongly increased contents of especially Pb have been reported both for land animals and in human blood in proximity of a lead smelter near a metallurgical centre in Kosovo (Shllaku and Landner, 1995). Other examples of adverse effects of a myriad pollutants on flora and fauna abound in the literature.

Pathways of contaminant transfer

Contaminant transfer from soils to plants, and ultimately humans, comprises at least three pathways: uptake through plant roots, adhesion to plant parts, and transfer of volatile contaminants from the soil. The relative importance of these different routes depends on rates of root uptake, ease of volatilization, atmospheric parameters and plant canopy structures, as well as the chemical and physical characteristics of a contaminant. Apart from the basic driving mechanisms for each of the transfer processes (e.g., precipitation and adsorption, volatilization, transformation of compounds, bio-adsorption) features such as isotopic exchange in the soil matrix, temperature-gradients and increased gas pressures caused by biodegradation can be important (Bridges, 1987).

Possible processes/reactions involved in transformation and transport of heavy metals in soil are summarized in Table 2. Pedogenesis may also play an important role in the distribution and partitioning of trace elements in the soil (Xing and Dudas, 1993). As the acidity of rain increase, changes in pH, ionic strength and ion activities of the soil solution will influence the charge characteristics of the soil, and thereby contaminant behaviour (Matschona and Vogt, 1997). Alternatively, industrial activities, such as cement production, locally can increase markedly the pH of snow cover, soils and waters. Changes in atmospheric base cation concentrations also may be important in the CTB context (Hedin *et al.*, 1994), as this will influence the amounts of acidifying compounds that can be neutralized by atmospheric dust particles, further complicating the overall picture. Time lags and temporal variations in soil solution chemistry affect the response of ecosystems or vegetation types to current or historical pollution loads (Løkke *et al.*, 1996).

Table 2. Important reactions affecting heavy metal behaviour in soil

| | Anion exchange | Cation exchange | Acid-base hydrolysis | complexation with organic and inorganic ligands | redox or methylation |
|----------|----------------|-----------------|----------------------|---|----------------------|
| Cadmium | | x | x | x | |
| Arsenic | x | | | | x |
| Mercury | | x | x | x | x |
| Selenium | x | | | | x |
| Lead | | x | x | x | x |
| Zinc | | x | x | x | |
| Copper | | x | x | x | |

Source: Novotny (1995)

Geochemical barriers

Regional differences in static and dynamic soil properties — both horizontally, vertically — largely control a soil's capacity to control movement of pollutants. As such, each soil may be viewed as a chromatographic column or system of geochemical barriers (Glazovskaya, 1991). Similarly, Blum and Wenzel (1991) made an overview of necessary parameters for the assessment of soil vulnerability, with respect to:

- (1) the general mechanical filtering capacity of the soils (particle size distribution, bulk density);
- (2) the physico-chemical buffering potential (CEC);
- (3) the resistance of the soil to acidification and alkalinisation (pH, Base Neutralizing Capacity, Cation Exchange Capacity);
- (4) ease/resistance to metal mobilization (pH, CEC, clay and sesquioxide content, OM content) or pesticide buffer potential (pH, CEC, clay and sesquioxide content, OM content, soil hydrology/redox) for the considered contaminant.

Effects of contaminated soil involve aspects of the environment about which very little is known (Sheppard *et al.*, 1992). Nonetheless, a wealth of information on the behaviour of inorganic and organic contaminants, as influenced by soil-environmental factors, is available from the literature (Adriano, 1992; Blume and Brümmer, 1987a, 1987b; Blum, 1990; Briggs, 1990; Fränze, 1987; Hesterberg *et al.*, 1992; Fine *et al.*, 1997). Less information is available on how the various environmental triggers and capacity controlling soil factors may change over time scales of decades to centuries to affect the mobility and biotoxicity of chemicals (Ronse *et al.*, 1988; Stigliani, 1988; Arnold *et al.*, 1990; Scharpenseel *et al.*, 1990).

2.2 Soil vulnerability in relation to Chemical Time Bombs

The concepts of soil sensitivity, susceptibility and vulnerability often are used interchangeably in the literature, forming a basis for confusion (e.g., Van Duijvenbooden and van Waegeningh, 1987; Van Lynden, 1995b). The initial SOVEUR workshop defined vulnerability, in relation to 'chemical time bombs', as the 'capability for the soil system to

be harmed in one or more of its ecological functions' (Batjes and Bridges, 1991). These functions include production of humus, filtering, storage, buffering and transformations of heavy metals, persistent organic chemicals and other xenobiotics. Additionally, the protective role of soil and the genetic reserve of its flora and fauna are significant issues. Important processes (triggers) influencing a soil's capacity to hold various contaminants and pollutants include: acid precipitation; liming of agricultural land; eutrophication; salinization; water erosion; and, changes in climate, hydrological conditions and land use (Hesterberg, 1993; Scholten and Szabolcs, 1993; Mayer, 1993; Brüggemann *et al.*, 1997).

Soils are chemically and biologically complex media comprising weathered and newly formed mineral fragments, organic matter in various stages of decomposition, microorganisms, and solutes and gases in its pores. In Europe they are diverse, ranging from shallow and stony Lithosols to poorly drained Histosols rich in organic matter (FAO/Unesco, 1981; EEC, 1985; Jamagne *et al.*, 1993). Depending on its inherent properties such as content of clay, organic matter and calcium carbonate and cation exchange capacity, each soil will react in different ways to similar pollution and environmental changes.

Table 3 lists seven important 'capacity controlling properties' affecting soil buffering and maximum retention capacity for heavy metals and persistent organic chemicals. In case of other contaminants, different sets of 'capacity controlling properties' may have to be considered. Any scenario causing an environmental change that affects these capacity controlling properties will subsequently affect the maximum 'sink capacity', either favourably or unfavourably (Hesterberg *et al.*, 1992).

A 'strong delayed response' may be observed in soils which can store large amounts of potentially mobilizable chemical compounds; these sudden occurrences of pollution are the 'chemical time bombs'. Alternatively, so-called 'weak rapid responses' or 'whimpers' occur where compounds are easily lost in continual small amounts, without a trigger effect. These are not 'chemical time bombs' but rather what the SOVEUR workshop participants termed 'more or less' polluted areas. These areas will be shown on the soil degradation status map for the area (Van Lynden, 1997).

In the CTB-sense, the most vulnerable soils are those with high but finite capacities for storage of potentially harmful and mobilizable chemicals (Stigliani, 1993; Batjes and Bridges, 1993). In addition, time-delayed and then sudden, non-linear releases of pollutants are important in the CTB concept (Ter Meulen-Smidt *et al.*, 1993, p. 265). Consequently, the chemicals of concern with respect to CTB-occurrences are the long-lived species most resistant to chemical decomposition, especially heavy metals and persistent organic-chemicals.

Table 3. Important soil capacity-controlling properties for heavy metals and persistent organic chemicals

| Capacity controlling property | Possible detrimental environmental effects |
|--|--|
| Cation- or anion exchange capacity (CEC resp. AEC) | Soil having a low CEC or AEC has a low capacity to retain cations, such as heavy metals, or anions, such as Arsenic and organic anions, by sorption. The size of CEC and AEC mainly depends on clay content and type, organic matter content and type, and soil pH. |
| Soil reaction (pH) | A decrease in soil pH (generally) increases heavy-metal solubility, decreases CEC, and alters microbial composition and activity |
| Redox potential (Eh) | A decrease in redox potential (more reducing moisture conditions), dissolves iron and manganese oxides, which mobilizes oxide-sorbed toxic chemicals. Increasing redox potential mobilizes heavy metals by dissolving metal sulfides. |
| Organic matter content/quality (OM) | Decreasing OM content reduces CEC, soil pH buffering capacity, the sorption capacity for toxic organic compounds and soil water-holding capacity; alters physical structure (e.g., increases soil erodibility and compaction hazard) and decreases microbial activity. |
| Soil structure | Altering soil structure can reduce drainage and thereby increase redox potential (more oxidizing conditions), increase soil erodibility, affect the rate of chemical release to drainage water, and alter soil pH. |
| Salinity | Increasing salinity solubilizes toxic chemicals by altering the ion-exchange equilibrium, increasing soluble complexation, and decreasing chemical thermodynamic activities in solution. Further, it can decrease microbial activity. |
| Microbial activity | Altering microbial activity and population ecology can reduce toxic degradation of organics (increase in toxic build-up), and alter redox potential and pH. |

After: Hesterberg *et al.* (1992).

Summarizing, the 'chemical time bomb' (CTB) concept stresses:

- (1) the (changing) capacity of the soil reservoir to hold or release contaminants, and
- (2) a trigger system.

The severity, nature and timing of the impacts resulting from CTBs vary with (Batjes and Bridges, 1993):

- (1) the degree of *loading* of the soil with a particular chemical;
- (2) the capacity or *propensity* of the soil to retain a chemical;
- (3) the type and intensity of the *triggers*;
- (4) the *sensitivity* of individual soils to the respective triggers.
- (5) the *targets* affected by the released pollutants.

The combined interpretation of items (2), (3) and (4) permits mapping of the *relative* vulnerability of soils to a pre-defined pollution scenario. In a Geographic Information System (GIS), the *relative* 'vulnerability map' can be overlaid onto a map of current (accumulated) loadings (1) to show where chemical time bombs are prone to occur. At the

(sub)continental scale, such preliminary analyses can provide the basis for identifying areas at risk from Chemical Time Bombs (CTB) where more detailed studies of soil pollution would be useful in so-called regional ‘windows’, of higher spatial resolution. These in turn can be used for identification of areas, which may require urgent remediation at the local level (Figure 4).

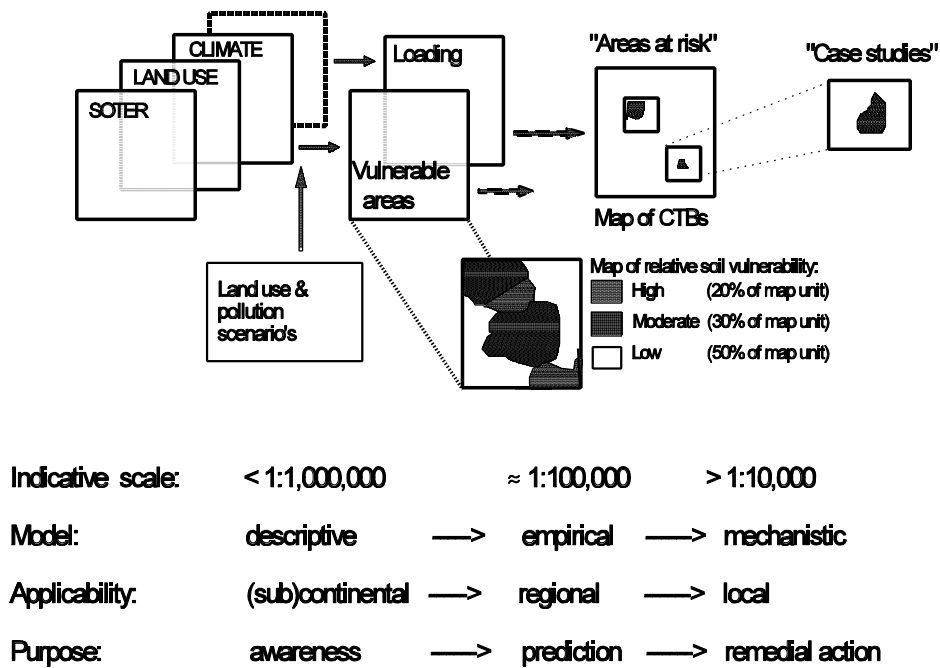


Figure 4. Schematic approach to studying soils at risk from chemical time bombs in relation to scale

2.3 Practical implications

The preceding, non-exhaustive, introductory review has shown that the study and estimation of pollutant accumulation and migration under changing environmental conditions is complex, irrespective of scale. This is so in view of the great number of chemicals, soils and sediments. Therefore, a simplified and explicit methodology is needed for the current, sub-continental assessment of soil vulnerability to pollution, for which a methodological framework is proposed in Chapter 3.

3 METHODOLOGICAL FRAMEWORK

3.1 General

3.1.1 Reducing functional complexity in relation to scale

GIS-based modelling of contaminant mobilization and solute transport basically involves linking three different kinds of activities, namely (1) model building to describe the physical and chemical processes (incl. transport, transformation, speciation and bio-accumulation processes; Novotny, 1995), (2) data collection to provide values for model parameters and attributes, and (3) GIS to provide an organized data structure for the automated handling of the input data to the computations and for the display of results (Burrough, 1996).

The scale of a map determines the detail of information that can be shown, and in soil vulnerability studies it determines the soil-environmental factors which may be considered at the specified scale of interest (Klijn, 1991; Wessman *et al.*, 1992; Corwin, 1996). At a scale of 1:2.5 M, as adopted for SOVEUR, soil mapping units are compound. When assessing the vulnerability of these soil associations or complexes, special attention must be paid to their most vulnerable component soils. In other words, one should not determine an 'average' vulnerability for the respective components of compound map units, nor look only at the vulnerability of the spatially dominant soil type. The relative propensity of certain pollutants to accumulate in each soil type of a compound mapping unit can be tabulated when using the methodology developed for SOTER, the World Soil and Terrain Digital Database (Van Engelen and Wen, 1995). Methodological guidelines for compiling the soil and terrain data for Central and Eastern Europe are discussed elsewhere (Batjes and Van Engelen, 1997).

For the current assessment of 'soil vulnerability', data are needed on pH, calcium carbonate content, clay content, clay mineral type, texture and organic matter content by soil unit (*sensu* FAO/Unesco, 1990). In addition, the site relief, lithology, total depth of soil, water regime and length of growing period, being closely related to the ability of soils to resist deleterious changes, are needed also. Depending on the chemical substances and environmental processes under consideration these factors must be given different 'weightings'. This means that several soil vulnerability maps, each corresponding to a well-defined pollution scenario, can be compiled from the same base map and set of key attributes (Van Duijvenbooden and Breeuwsma, 1987; Klijn, 1991; Varallyay, 1991). The relevant spatial and attribute data can readily be manipulated in a GIS, using a system of matrices (Fränzle, 1987; Blume and Brummer, 1987a, b).

The assessment of soil vulnerability to pollution by chemical compounds is complex, irrespective of temporal and spatial scales. There is a wide range of chemical compounds and their derivatives, each of which may react differently to changing soil-environmental factors and processes. Despite considerable knowledge of the behaviour of organic and inorganic substances in simple laboratory systems, it is still poorly known how these substances behave in complicated soil systems, under changing boundary conditions.

Organic contaminants transported in the vapour phase, for example, will differ from those transported in the liquid phase (Fine *et al.*, 1997). Knowledge of soil biology and its relationship to solute sorption, degradation and sequestration at the field scale is also incomplete (Wagenet and Hutson, 1996). The presence of competing metals in sewage sludge, for instance, may increase the general availability of other metals and competition of the various metals for sorption sites will be complex (Schmitt and Sticher, 1986). Organic-rich linings and fillings in root-derived soil macropores may exhibit higher sorptivities for Cd and Mn than homogenized soil, particularly at pH 4 to 5 (Turner and Steele, 1988). Heavy metals, further, may exist in various oxidation states, complexed forms, or as co-valently-bound organic compounds, which vary in bio-availability, toxicity, volatility and geochemical mobility (Adriano, 1992). Possible effects on the speciation and mobility of heavy metals in soils and sediments of various environmental processes, such as human-induced acidification, land management and changes in sea-level associated with climate change, are still poorly quantified (Stigliani, 1988; Bouma *et al.*, 1997).

The above considerations indicate, again, that a key issue in any assessment of 'soil vulnerability' will be how to decrease the functional complexity in relation to the adopted scale of mapping (Van Duijvenbooden and Van Waegeningh, 1987; Andersen and Gosk, 1989; Desaulles, 1991; Giordano *et al.*, 1995; Batjes, 1996). The smaller the map scale, the greater the need will be for an explicit methodology to simplify the information available and remain realistic (Van Duijvenbooden and Breeuwsma, 1987; Chadwick and Kuylenstierna, 1990; Desaulles, 1991; Glazovskaya, 1991; Giordano *et al.*, 1995).

Both soils and contaminants should be grouped according to their environmental behaviour (Fränze, 1987; Glazovskaya, 1991), in relation to pre-selected trigger-systems. Soil maps then can be used in conjunction with physico-chemical data on contaminant behaviour to make rational judgements about possible contaminant behaviour (Livens and Loveland, 1988; Bogaci and Florea, 1986; Glazovskaya, 1991).

3.1.2 Modelling approaches

Describing and modelling of contaminant behaviour in the soil is complex, because the physical and chemical properties and dynamics of any soil compartment (e.g., horizon) are governed by external forces such as (acid) precipitation, air temperature and solar radiation. The principles embodied in conventional water flow equations and convective-dispersive solute transport equation are (only) sufficient to describe fate of chemicals in homogenous, uniformly packed soil columns (Jury, 1996). Contaminant (solute) transport models have been reviewed by Bonazoukas (1987), Sheppard *et al.* (1992) and Wagenet and Hutson (1996), while Coughtry (1988) reviewed mathematical models for radionuclides transport in the soil. One of the major problems that must be faced in linking models of solute transport in the vadose zone to GIS is the inherent incompatibility in scale (Jury, 1996; Wagenet and Hutson, 1996). Soil physical properties governing unsaturated flow, for example, are highly variable over short distances and tend to become uncorrelated over distances of several metres. Generally, the sample spacing adopted in routine soil surveys or in continental data compilation activities based on available data, as adopted for

SOVEUR, is much greater than the distance over which transport parameters vary. Therefore, some compromise between rigour and practicality must be adopted (Jury, 1996; Desaulles, 1991; Prieler and Anderberg, 1996). Very accurate predictions of contaminant migration on time scales of days and large spatial scales in uniform porous or soil media will not serve the regulators (Sheppard *et al.*, 1992), who more commonly need to know about the future regional impacts of contaminants in soils. During the State-of-the-Art-Conference on CTBs, at Veldhoven (Ter Meulen-Smidt *et al.*, 1993 p. 265), it was stated that even though not all processes and mechanisms are understood, it should be possible to use 'quick and dirty' methods to identify areas at risk to give policy makers an indication where certain types of CTBs may occur. However, it will not yet be possible to specify when and with which intensity and these CTB-events may occur. Additional fundamental research should be executed to arrive at better, quantitative predictions and better remediation technologies for use at the local level (see Figure 4).

3.1.3 Conceptual framework

In view of the limitations of complexity, data availability or accessibility, scale, process-knowledge and time, the SOVEUR project will focus on developing and implementing an explicit, rule-based assessment-level model that can be used at a continental scale of 1:2,500,000. Generalized relationships between external triggers, associated changes in the capacity controlling properties of different soils and subsequent pollutant behaviour, will have to be considered in this conceptual model to identify areas prone to CTBs.

Figure 5 illustrates, in a simplified manner, that there are two main ways in which contaminant saturation can occur in the soil with subsequent triggering of CTBs. Of these, the second mode (b) may be far more important than the first mode (Stigliani, 1993 p. 19):

- (a) simple saturation over time. As the system approaches maximum soil sorption capacity, the ratio of the concentrations of dissolved (mobilized) chemical to the amount sorbed to the soil increases (i.e., no marked trigger system);
- (b) saturation as described for (a) plus triggering which causes a rapid change in a soil property that reduces the maximum capacity for retention of the contaminant.

Vulnerability class membership can be defined by specifying the ranges of a limited number of key property values that an individual soil must meet (Varallyay, 1991; Glazovskaya, 1991; Rauta *et al.*, 1993). In first instance, the focus should be on the impact of selected environmental and policy driven changes on the behaviour of selected contaminants, such as heavy metals.

For short-term risk assessment (5-15 yr) it will be sufficient to understand how the *capacity controlling properties* (CCP) of a soil determine contaminant mobility, and hence their leaching and bio-availability. In the longer-term, changes in geochemical cycles become important. These can be induced, for example, by changes in land use policies, continued acid precipitation and hydrological changes, that gradually affect the size of the 'capacity controlling properties' (CCP) so as to suddenly cause strong increases in contaminant concentrations over a short-time period (Salomons, 1995). Consequently, it will be necessary to bring together scientists from various disciplines (soils; biogeochemistry; land use

change; climate change; GIS) to solve different pieces of the puzzle schematized in Figure 5.

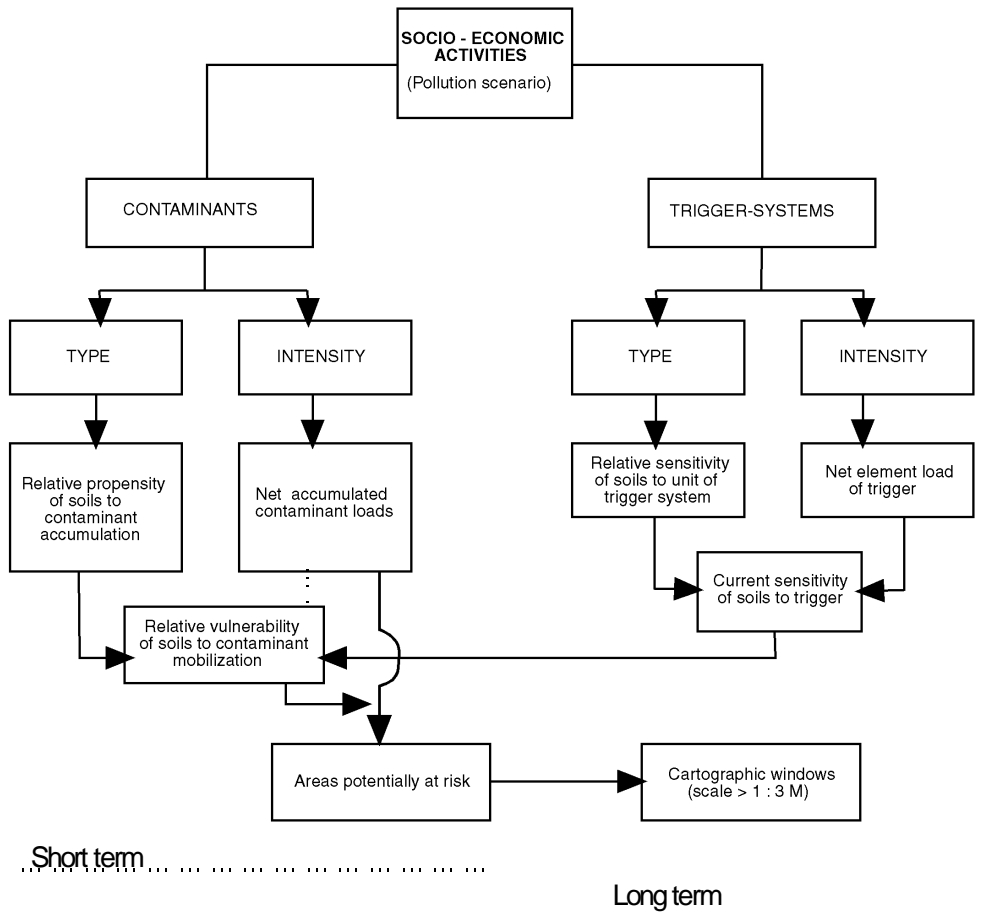


Figure 5. Conceptual framework for assessment of the vulnerability of soils to pollution

3.2 Pollution scenarios

3.2.1 Sources and types of contaminants

The first critical step in scenario development for Figure 5 will be to identify substances known to cause harm at the regional level which may accumulate in the soil for which triggers are known. Sources of various contaminants can be characterised according to five major sectors from which they originate (Table 4).

Table 4. Examples of contaminants derived from human activities

| Activity | Examples of derived contaminants |
|--------------------|---|
| Industry | <ul style="list-style-type: none"> — Zn, Cd, Pb, Hg, Fe, Mn, Cr, Ni, Cu from metallurgy — Pb, Al, Ga in tilings — Pb, Fe, Zn, Ni, PCB, oils, phenyls and related substances from chemical wastes — Pb, Cd, Oils, Phenols from fuel combustion — Ga, Se from microelectronics |
| Energy | <ul style="list-style-type: none"> — Dioxin, halogenic organo-carbons, NH₃, Sulphur, NO_x, aerosols and some heavy metals from power companies — Radionucleides from nuclear accidents — Radioactive gases (Rd) from coal burning |
| Agriculture | <ul style="list-style-type: none"> — Cd contaminants in P-fertilizers — P₂O₅, NO_x and K₂O in manure — Hg and Cu in pesticides — halogenated organic compounds — mineralisation and secondary salinization |
| Urban and domestic | <ul style="list-style-type: none"> — Dioxin, PACs and some heavy metals — NH₃, SO₂, As, dust from heating — Cd, Pb, PACs and oils from traffic |
| Military | <ul style="list-style-type: none"> — Cr, Ga, Ti, Se and radionucleides from armament industry/use |

After: Csikos (1996)

During a preceding workshop, the following contaminants have been earmarked for possible consideration in SOVEUR: heavy metals, including Pb, Cd and Cu; persistent organo-chloric compounds such as some pesticides and PCBs; and other xenobiotics such as F, Bo and As (Al). These chemicals are potentially toxic, and are mobilizable and persistent. Further, they are a source of public concern because of past, present and possible future inputs (loadings) into the environment.

3.2.2 Processes and trigger systems

Several important natural and man-made processes and ‘triggers’ are currently modifying the *relative* vulnerability of soils [or capacity of the soil to hold the contaminant] in Central and Eastern Europe with respect to the considered compounds. These include: acidification; land use change; eutrophication; salinization; erosion; climate change; and hydrological changes. The effect of these often interacting trigger(s) is to sharply reduce the capacity of the soil to buffer, filter or store the contaminants which are then released into the environment as pollutants (Hesterberg *et al.*, 1992). A review by Stigliani (1993) suggests that even before climate change begins to be an important factor affecting CTBs, other changes in the short term, in particular land use changes, may be of more importance.

Land use is a major determinant of (current) soil pH and organic matter content, which in turn are important regulators of heavy metal mobility in the soil (Hesterberg, 1993). Important driving forces of land use change in Europe are (Kostrowicki, 1984; Prieler *et al.*, 1996b; Bouma *et al.*, 1997): (1) agriculture, in particular agricultural policy; (2) forestry; (3) environmental conditions; (4) social context, including the regions history; (5) policies related to land use planning. In the context of the current project, information on the *type* and *intensity* of the various trigger-systems, will largely have to be derived from auxiliary sources.

3.3 Propensity of soils to contaminant accumulation

3.3.1 Capacity controlling parameters

The choice of the basic soil parameters determining contaminant behaviour is essentially dictated by the availability of sufficient profile data of adequate quality at the considered spatial scale (Van Duijvenbooden and Breeuwsma, 1987; Batjes and Bridges, 1994). The most important capacity controlling factors affecting heavy metal binding are: soil depth, texture, content and type of organic matter, soil pH-redox conditions, the content of Fe-, Al- and Mn-oxides, and the type of heavy metals (Kabata-Pendias *et al.*, 1992; Livens and Loveland, 1988). Sorption and desorption kinetics of heavy metals in soils are non-linear and further complicated when several metals compete for the same sorption sites, and as new derivatives are formed during their movement through the soils. Easily mobile trace elements like Zn and Cd exist mainly as organically bound, exchangeable, and water soluble species. Cu and Mo predominate in organically bound and exchangeable species, and their behaviour is strongly influenced by changing soil conditions. Slightly mobile metals, such as Pb, Ni and Cr, are mainly bound in silicates (Kabata-Pendias, 1995).

The fate of POPs in the soil is determined by a number of processes, including photolytic, chemical and microbial transformation, sorption, plant uptake, transport and volatilization, the interaction of which is complex and non-linear (Tiktak *et al.*, 1996). Adsorption of pesticides to soils is determined largely by the character of the pesticide and soil to which they are added. The existence of functional groups such as -OH, -NH₂, -NHR, -CONH, -COOR and -⁺NR₃ in the chemical structure encourages adsorption especially on the humus (Brady, 1990). In general, the larger the pesticide molecule, everything else being equal, the greater is its adsorption. Some pesticides, such as Diquat and Paraquat are also adsorbed by silicate clays; this adsorption tends to be pH-dependent with maximum adsorption occurring at low pH where protonation occurs. The majority of pesticides degrade rapidly enough to prevent build-up in the soils, but some of these may be contaminated with As creating another source of pollution (Brady, 1990).

From the above, it follows that the basic soil data for our study should include:

- (1) depth of soil horizons
- (2) pH and soil carbonate status
- (3) soil texture
- (4) organic matter content

- (5) soil mineralogy
- (6) soil drainage class (redox conditions)
- (7) salinity

These soil criteria must be combined with (external) environmental features such as the site relief, lithology, depth of groundwater table, water regime and length of growing season to assess the *relative* vulnerability of each soil for the considered chemical. The water regime, for example, will reflect the effects of climate and groundwater on contaminant behaviour, while differences in soil temperature may affect the speed and intensity of physico-chemical processes.

Information on the above parameters can be combined into a simple matrix table, for subsequent computerization, permitting assessment of the propensity of a certain soil for contaminant accumulation (Table 5). Although not directly quantifiable at scale 1:2.5 million, the 'immobilization capacity' permits ranking of soils indicating which soils may give rise to persistent problems if contaminants were deposited uniformly.

Table 5. Conceptual framework for assessment of soil propensity to contaminant accumulation

| Soil Component [†] | Soil and terrain properties | | | | | Relative propensity to cont. accumulation | | | |
|-----------------------------|-----------------------------|------|-------------------|---------|-----|---|----|----|----|
| | pH | OC | CaCO ₃ | texture | ... | Pb | Cd | Hg | .. |
| RO001/1/1 - Orthic Luvisol | 7.5 | 3.3% | 2.2% | SCL | | H | H | H | - |
| RO001/1/2 - Orthic Gleysol | 6.5 | 3.0 | 0.2 | CL | | M | M | M | - |
| - | - | - | - | - | | H | M | M | - |

Note: Each Soil Component in SOTER can be characterized by its dominant FAO-Unesco (1990) soil units, using regionally representative profiles. Classes of relative propensity for element-accumulation will range from 'lowest' (L) to 'highest' (H); ratings shown are hypothetical only.

3.3.2 Compilation of regional parametric overviews of soil attributes

Based on the soil profile data available by soil unit (from SOTER activity), regional parametric overviews can be generated as single value maps (i.e., as medians). Where measured soil data are lacking, use may have to be made of auxiliary sources to fill eventual gaps (FAO, 1995; Batjes *et al.*, 1997). A selection of these single value maps can then be used to identify areas of land with differing propensities for accumulation or leaching of heavy metals and other contaminants to the groundwater.

The type of pollutant and research purpose will determine which single value maps are of importance in a special case (Table 6). Maps of CEC, organic matter, and clay will be of importance for retention/leaching of heavy metals, while a map of organic matter content will give some information on the propensity for organic micropollutants to accumulate in a particular soil (Klijn, 1991; Van Duivenbooden and Breeusmwa, 1991).

Table 6. Soil characteristics determining the accumulation/leaching of selected contaminants

| Soil characteristic | Type of pollutant | | | |
|----------------------------|-------------------|---------|-----------|----------------------|
| | Heavy metals | Nitrate | Phosphate | Org. micropollutants |
| <i>Chemical</i> | | | | |
| pH | X | X | X | X |
| Redox potential | X | X | | X |
| CaCO ₃ content* | X | | | |
| Organic matter* | X | X | | X |
| Clay content* | X | | | X |
| Sesquioxide (Al, Fe) | X | | | |
| Cation exchange capacity* | X | | X | X |
| <i>Physical</i> | | | | |
| Depth to groundwater | X | X | | X |
| Residence time of water* | | X | | X |
| <i>General</i> | | | | |
| Thickness of vadose zone* | X | X | X | X |

* Attributes as used in a 1:400,000 scale study for the Netherlands (Van Duijvenbooden and Breeuwsma, 1987).

3.4 Relative sensitivity of soils to various trigger systems

Chadwick and Kuylenstierna (1990) defined sensitivity as 'the responsiveness of an important variable to an independent forcing variable'. They differentiated between the 'absolute' site-specific sensitivity, indicated by the current chemical status of the soil or the surface water of a site, together with the tolerance limit of species, and the 'regional sensitivity' which compares the ability of one region to withstand the effects of acidic inputs to another. Important factors that determine the sensitivity of a site to acidic deposition, for example, are: bedrock lithology, soil type, land use and rainfall (Chadwick and Kuylenstierna, 1990). Soils that are most sensitive to acidification are those with a low acid neutralizing capacity (ACN), i.e., little residual calcite, a low cation exchange capacity, and low contents of minerals that consume protons or basic cations by dissolution (Hesterberg, 1993). For example, sandy soils of low organic matter content have a low ACN and are therefore sensitive to more rapid acidification than clayey soils. The most sensitive soils to acidification are found in areas where the bedrock consists predominantly of gneiss, granite and other slow-weathering, acid rocks (Chadwick and Kuylenstierna, 1990). De Vries *et al.* (1993) proposed a conversion between FAO soil types and assumed dominant parent material in order to estimate weathering rates.

Summarizing, soil sensitivity to a particular trigger system can be described as the result of:

- (1) the inherent resistance/resilience of the soil system to the trigger (type);
- (2) the intensity of the trigger (e.g., from acid deposition or exceedance maps).

Rating systems for assessment of the relative sensitivity of soils to particular trigger systems, such as acidification, can be derived from the literature (Varallyay, 1991; Rauta, 1991).

3.5 Relative vulnerability of soils to contaminant mobilization

The solubility, and consequently mobility, of solid-bound metals in soils can be changed by three major processes which induce changes in speciation (in solution, at the surface or in the bulk of solids): (a) lowering of pH, (b) changing the redox conditions, (c) increasing the concentration of inorganic salts and of natural or synthetic complexing agents.

The relative vulnerability of soils to element mobilization is obtained by overlaying the maps of propensity to contaminant accumulation with the map of relative sensitivity to contaminant mobilization as induced by the specified trigger system (Figure 5). In each case, a rule-based rating system will have to be developed and tested for its applicability.

3.6 Pollution status

Soil vulnerability must be specified with respect to agents, causes and effects (Desaules, 1991). In order to assess soils “at risk” from a defined pollution scenario, information on net accumulated loads for the considered pollutants must be overlaid onto the *relative* vulnerability maps as shown in Figure 5. Such databases of “pollutant loads” will have to be obtained from auxiliary sources, and their access may be limited (see Webster, 1997).

3.6.1 Sources

A generalized sources map would be useful, by contaminant, in that these sources will determine the main (historical) input of chemicals into the environment. Pragmatically, the sources can be categorized by the human activities from which they derive (Table 4) and the way in which the contaminants are emitted/distributed (Fraters, 1996):

- (1) Agricultural activities, such as application of fertilizers and pesticides with their contaminants, or salinization associated with poor irrigation practices.
- (2) Energy use by power plants, including emissions of heavy metals, acid and corrosive substances and accidental releases of radionuclides.
- (3) Derivatives of industrial activities, such as mine spoil, PCB and diazine release, dumping of chemical wastes, and emissions from cement factories (dust, F, heavy metals) and NP-fertilizers manufacture (F, SO₂, NO_x).
- (4) Urban, commercial and domestic activities, including incineration of waste and landfilling
- (5) Military activities, such as waste containing heavy metals and persistent organic compounds or radionuclides.

Prieler *et al.* (1996a) illustrated the importance of regional differences and changes in time, with respect to total loads of heavy metals to the soil and the share of agricultural or

atmospheric load in this total. Agricultural loads, by contaminant, in principle can be derived from simplified mass balances, requiring statistical data on land use, crop areas, crop productivity and animal density; data on total fertilizer consumption by (administrative region in a) country; data on fertilizer application by crop and region; data on atmospheric deposition; data on pollutant excretion by animal head (e.g., cows, pigs, sheep and goats); and ratios of micro/macro-nutrients in commercial fertilizers and manure.

Suspension of soil particulate matter into the atmosphere is commonly modelled using an atmospheric dust load approach. Modelling the transport of contaminated particles/aerosols suspended from ground level sources generally employs statistical trajectory dispersion models. Deposition of contaminants back to the surface is based on deposition velocities to model dry deposition and washout ratios are used to treat wet deposition (Sheppard *et al.*, 1992 p. 372). Data on atmospheric emissions of Cd, Pb and Zn in Europe during 1955-1987 have been compiled by Olendrynskin *et al.* (1995). Atmospheric deposition will often be presented on a grid level (i.e., 150 x 150 km² EMEP grid), as opposed to data on agricultural inputs/loads that will have to be generated by administrative region.

Uncertainties related to model and data errors are prone to be significant at the considered scale (Loagu and Corwin, 1996; Prieler *et al.*, 1996), and results will mainly be applicable to large areas.

3.6.2 Enrichment factors

The natural concentrations of trace elements and radionuclides in soils depend upon the amounts present in the parent rocks from which the soils formed and upon soil forming processes closely related to climatic conditions (Kabata-Pendias and Pendias, 1992; Novotny, 1995). Natural concentrations of heavy metals in unpolluted (pristine) soils are known to vary considerably, and can be high for Radon and Pb, so before any discussion of contamination it is desirable to define a *baseline* for these soils (Bridges, 1989b). Geochemical baselines, however, represent concentrations specific to one area and time and as such they need not necessarily be true background values. In contrast, *geochemical backgrounds* should represent natural element concentrations, which ideally exclude human influences; thereby they permit calculation of *enrichment factors* or 'coefficients of technogenic concentration' (Glazovskaya, 1991), as an indicator for the magnitude of environmental pollution. Enrichment factors calculated on the basis of normalization to Al-content show that several metals, and in particular Cd, Pb and As are concentrated in the topsoil in comparison to their occurrence in the Earth's crust (Kabata-Pendias, 1995).

Concentration enrichment may occur during erosion, due to particle selectivity and the positive correlation between contaminant retention and clay and organic matter content (Ghadiri and Rose, 1991; Novotny, 1995). Korentajar *et al.* (1993) showed that, at the soil surface, up to 50% of Cd applied with fertilizer was lost due to water erosion. During this process, clay particles at the surface are preferentially removed with adsorbed trace elements. Heavy metals, bonded to suspended solids transported by erosion, tend to accumulate in aquatic bottom-sediments (Molnár *et al.*, 1995).

An overview of trace elements contents in main soil-parent rocks is listed in Kabata-Pendias and Pendias (1992, p. 63). Values presented for geochemical backgrounds often are difficult to compare due to differences in analyses, as a result of which they may have limited validity geographically (Edelman and de Bruin, 1986). Similarly, Kabata-Pendias and Pendias (1992) reviewed total (soluble) contents in the surface layer of soils of Central and Eastern Europe, pointing at the limited comparability of the various data by FAO-Unesco soil unit.

3.6.3 Net accumulated loads

Studies of environmental pollution by toxic (persistent) chemicals require analyses by main source in a long-term perspective, since contaminant accumulation rates from diffuse source are slow and subject to changes in land use and emissions (Prieler *et al.*, 1996a). Upon the identification of materials in which the contaminants (chemical) occur, the emissions and depositions to air, water and soil for each need to be estimated quantitatively (Rautengarten, 1993; Stanners and Bourdeau, 1995 p. 36-39). The net, accumulated loading with any particular contaminant, in principle, can be estimated using a simple mass balance, depending on the availability of the various types of input data at the considered spatial and temporal scales (Fraters and van Beurden, 1993; Prieler *et al.*, 1996).

Net element accumulation can be defined as the difference between the input and output of an element from a soil. Preparation of an accurate mass balance (by contaminant) may be difficult at a regional scale, considering the complexity of soil processes controlling the (im)mobilization and transport of the contaminant in soils and the variability of soil and land use in each spatial (administrative) unit. Thus, simplifications are needed.

Net element accumulation, in the case of heavy metals (HM), can be computed with a simple mass balance:

$$\begin{aligned} \text{HM}_{\text{net load}} = & \text{HM}_{\text{atm. dep.}} + \text{HM}_{\text{manure}} + \text{HM}_{\text{fertilizer}} + \text{HM}_{\text{sewage sludge}} \\ & + \text{HM}_{\text{irrigation water}} \pm \text{HM}_{\text{water erosion}} \pm \text{HM}_{\text{wind erosion}} \pm \text{HM}_{\text{leaching/cap. rise}} \\ & - \text{HM}_{\text{uptake with harvested product}} - \text{HM}_{\text{geochemical background}} \end{aligned}$$

with:

HM_k , the element (contaminant) under consideration, and its source or fate (k)

$$\text{HM}_{\text{av. uptake with harvested product per ha}} = \frac{\sum(A_i * Y_i * C_i)}{\sum A_i} \text{ for } i= 1 \text{ to } n$$

where:

A_i is the area under the specified crop (ha)

Y_i is the yield of this crop (1000 kg ha⁻¹)

C_i is the concentration of HM in g ton⁻¹ dry matter (mg kg⁻¹)

i , refers to the crop/vegetation under consideration (and time period)

Many products/substances contain heavy metals as impurities or contaminants, so that the focus in SOVEUR may have to be limited to say: Cd, Zn and Pb added with phosphate fertilizers and manure, initially. Nitrogen and Potassium fertilizers may be omitted, in this context, since their contribution to total agricultural load of the heavy metals should be very low and hence negligible (Prieler *et al.*, 1996a). Heavy metals may occur also in agricultural-lime, but data on their concentrations may be too uncertain to be included in a regional study. Metal loads applied with sewage, sewage sludge, wastes and pesticides also may be difficult to include both for lack of data and their much lower and variable contribution to the total load.

3.7 Areas at risk from chemical time bombs

Areas at risk from (delayed) pollution on the long-term can be assessed by analysing effects of plausible scenario's, which may be defined as so-called 'windows of opportunity' as suggested by Bouma (1994), for future heavy metal loads and land use changes. The combined identification of current risk and scenario analysis will provide the means to derive policy-relevant recommendations for decision makers (Fraters and van Beurden, 1993; Prieler and Andelberg, 1996).

Overlay of the map of relative vulnerability with a map of accumulated loads will combine available information on (1) soils that have a high relative propensity to accumulate specific heavy metals, (2) soils that are sensitive to particular trigger systems, and (3) administrative regions with high element loads, thus permitting the identification of areas at risk from CTBs.

3.8 Mapping approach in relation to data availability

The mapping resolution adopted for the SOVEUR programme is 1:2.5 million, with a publication scale of 1:3 million. The integrated nature of a (sub)continental assessment of soil vulnerability to pollution requires access to a wide range of uniform and comparable data. Functional regions with respect to certain pollution types can be defined as geographical regions with similar environmental characteristics such as climate, geological structure, topography, soil type, hydrology and land use. The baseline data for the study can be mapped either on a grid or polygon basis, the latter being the case for the soil and terrain data (Batjes and Van Engelen, 1997). Land use and land cover data, however, often will be in a raster format (RIVM/SC-DLO, 1996).

Generally, map units on small scale maps will have a higher internal variability than units on larger scale maps. Detailed definitions of representative profiles for small scale maps can be quite misleading as they suggest the possibility of making rather specific interpretations for extensive areas of land, so that, as an alternative approach, median values for the main properties by soil unit may have to be used instead. Determining realistic levels of soil profile data generalization when using small scale soil maps thus is considered critical in SOVEUR. Also, at the considered scale it is recommended that at most 4 to 5 classes of susceptibility or vulnerability be discerned (Sverdrup *et al.*, 1990; Chadwick and Kuylenstierna, 1990; Rauta *et al.*, 1993).

Van Woerden *et al.* (1995) presented a comprehensive catalogue of international data sets compiled in support of Integrated Environmental Assessment and Modelling. Other databases are available via UNEP-GRID and from the European Union, the later often with significant constraints on 'data-release' to various user-groups (Jones and Buckley, 1997). Van Lynden (1995a p.30) gives a summary of existing databases and monitoring systems related to soil problems in Europe. Summary data on the acid neutralizing ability of main rock types are presented in Chadwick and Kuylenstierna (1990).

The availability, compatibility and accessibility of the various data sets needed for assessment of soil vulnerability, must be inventoried by country by the national partners in SOVEUR. Results of this inventory will allow the SOVEUR workshop to identify which type of soil vulnerability mapping studies can be carried out during the project period, based on the overall availability of data and current scientific understanding of the processes involved.

4 CONCLUSIONS

The current report presents a conceptual framework for identifying areas of soils considered vulnerable to defined types of chemical time bombs, for use at a (sub)continental level. The methodology will allow generation of maps, and tabular output, which can serve to increase general awareness of soil vulnerability to specific types of pollution, allowing problem areas to be identified for subsequent (remediation) studies at regional and national levels.

As has been the case for preceding studies of the status of human-induced soil degradation (Oldeman *et al.*, 1991; Van Lynden, 1995b), the soil vulnerability mapping component of the SOVEUR project mainly aims to increase geographical awareness of areas potentially at risk from specified types of pollution. Target beneficiaries are Ministries and planning bodies in the collaborating countries who can use the database and derived maps for policy formulation at the regional and national level.

An important task of the SOVEUR workshop, scheduled for early October 1997, will be to assess for which pollutants and triggers (as well as regions and time periods) soil vulnerability maps should be compiled first, keeping in mind prevailing constraints of data availability/accessibility and time; full cooperation from agencies and institutions with essential data and information will be essential. In cases where certain data on pollution are lacking, a practical solution may be to define several windows of 'opportunity' (after Bouma, 1994). This should then permit to define plausible CTB scenario's. Rating systems for these studies may have to be refined by small groups of experts, by scenario. These rule-based models can first be tested and evaluated after the soil and terrain digital (SOTER) database for the region becomes available.

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