Helmut Meuser

ENVIRONMENTAL POLLUTION 18

# Contaminated Urban Soils




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# Contaminated Urban Soils

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*To my father*
*who always asked*
*whether I was getting on well with writing  of the book*
*and who died*
*shortly after it was completed*

*"In the fury of the moment I can see Master's hand*
*in every leaf that trembles, in every grain of sand."*
*B. Dylan*

# Foreword

With more than 50% of the world's population already living in towns and cities, migration from rural areas continuing at an alarming rate in developing countries and suburbanisation using more and more land in developed countries, the urban environment has become supremely important with regard to human health and wellbeing. For centuries, urbanisation has caused relatively low level soil contamination mainly by various wastes. However, from the time of the Industrial Revolution onwards, both the scale of urban development and the degree of soil contamination rapidly increased and involved an ever widening spectrum of contaminants. With constraints on the supply of land for new urban development in many countries, it is becoming increasingly necessary to re-use previously developed (brownfield) sites and to deal with their accompanying suites of contaminants. It is therefore essential to fully understand the diversity and properties of urban soils, to assess the possible risks from the contaminants they contain and devise ways of cleaning up sites and/or minimizing hazards.

The author, Helmut Meuser, is Professor of Soil Protection and Soil Clean-up at the University of Applied Sciences, Osnabrück and is one of Europe's foremost experts on contamination from technogenic materials in urban soils. He has many years' experience of research in Berlin, Essen, Osnabrück, other regions of Germany, and several other countries. In this book he has applied his wide-ranging experience to provide a comprehensive review of the subject of contaminated urban soils. One of the book's great strengths is the inclusion of data and other information from studies in urban areas across the world with widely differing climatic conditions, soils and types of industrial and residential development. Commencing with a brief consideration of the geographical basics of urbanization, he goes on to cover in detail the causes and sources of the different types of contaminants found in the soils of towns and cities. These include inputs of heavy metals and organic pollutants in dusts arising from industrial emissions, roads and railways, construction sites, excavations for pipelines, river flooding, horticultural and agricultural sources, derelict land, accident sites and various types of fill.

Particular emphasis is placed on man-made (or 'technogenic') substrates which, together with dust deposits are the most important sources of contaminants in the urban environment. These technogenic materials include construction debris, demolition waste (such as concrete, brick, tarmac, steel), slags, coal ash, mining wastes, municipal solid wastes and sludges and are dealt with in considerable detail. A key for the identification of these materials in field excavations is provided which will be particularly helpful for surveyors, planners, construction engineers and environmental health specialists in assessing brownfield sites. The characteristics and contamination potential of these technogenic materials are considered in detail and followed by a classification of 'anthropogenic' soils. These comprise the 'Anthrosols' developed from cultivated natural soils and 'Technosols' formed on deposited materials. Having discussed the origins and classification of these urban soils and their constituents, the book then progresses goes to the main physical and chemical properties of soils which influence their contamination potential, especially the mobility and bioavailability of heavy metals and organic pollutants. These physical properties include surface sealing, erosion and deflation, compaction, skeleton enrichment, movement of the water table and subsidence. The chemical properties include total concentrations of contaminants, pH, carbon content and biological activity, texture and binding compounds and nutrient materials. Pedogenic processes such as humus formation, pedoturbation, weathering, aggregation and redox (gleying) are also considered in the context of their effects on the behaviour of the range of contaminants.

The book concludes with a major chapter on the assessment of the hazards posed by urban soils and a short, final chapter on the outlook for the future for contaminated urban soils. One of the most important parts of the hazard assessment chapter is the range of soil quality standards in use in different countries and their respective cut-off values for the safe maximum concentrations of many of the most important contaminants. These provide a valuable guide to the concentration levels which are considered hazardous for certain urban land uses and in some countries these are the basis of statutory clean-up regulations. It is likely that these quality standards will be expanded to include more potentially hazardous substances and that the safe maximum levels will be amended in the light of developments in toxicological research. The outlook for the future for contaminated urban soils chapter stresses the need for more attention to be paid to the subject, especially as the contamination potential of urban land increases with time and with the accelerating pace of urbanisation. The greatest contamination risks lie with derelict industrial sites and technogenic substrates, but garden and allotment soils (Anthrosols) used for growing vegetables can also pose a risk to health where they have been contaminated by the addition of various materials including the long-term deposition of pollutant-containing dusts.

This book provides a very thorough and concise review of contaminated urban soils. With its international scope based on examples from the USA, China, India, Germany, the Netherlands, the United Kingdom, France, and other countries it will be particularly relevant to professionals involved in urban planning, construction and environmental health in these and other countries. Likewise, it has also been designed to be a state of the art textbook for use in advanced university courses around the world in Environmental Science, Urban Geography, Landscape Architecture, Waste Management, Ecology and Soil Science. This one volume contains material which is otherwise only partially covered in several existing textbooks and even then not in such a comprehensive way and so it is highly recommended to both students and lecturers as well as practitioners dealing with contaminated land.

February 2010 Brian J. Alloway

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# Contents

# Abbreviations

| | |
|---|---|
| AC | air capacity |
| ADI | acceptable daily intake |
| Ag | silver |
| AMD | acid mine drainage |
| As | arsenic |
| ASM | artisinal and small scale mine |
| Au | gold |
| AWC | available water capacity |
| Ba | barium |
| BCR | Community Bureau of Reference Protocol |
| Be | beryllium |
| BTEX | benzene, toluene, ethylbenzene, xylene |
| Br | bromide |
| C | carbon |
| Ca | calcium |
| Cd | cadmium |
| CEC | cation exchange capacity |
| $CEC_{eff}$ | effective cation exchange capacity |
| $CEC_{pot}$ | potential cation exchange capacity |
| Cl | chloride |
| CN | cyanide |
| Co | cobalt |
| Cr | chromium |
| Cs | caesium |
| Cu | copper |
| DDT | dichlorodiphenyltrichloroethane |
| DHA | dehydrogenase acitvity |
| DM | dry matter |
| DOC | dissolved organic carbon |
| DTPA | diethylenetriaminepentaacetic acid |
| EC | electrical conductivity |
| EDTA | ethylenediaminetetraacetic acid |
| Eh | rodox potential |

| | |
|---|---|
| EPA | Environmental Protection Agency (USA) |
| F | fluoride |
| Fe | iron |
| H | hydrogen |
| HCB | hexachlorobenzene |
| Hg | mercury |
| I | iodine |
| K | potassium |
| $K_{oc}$ | Freundlich constant |
| Mg | magnesium |
| Mn | manganese |
| MSW | municipal solid waste |
| N | nitrogen |
| Na | sodium |
| Ni | nickel |
| O | oxygen |
| P | phosphorus |
| PAH | polycyclic aromatic hydrocarbons |
| $PAH_{EPA}$ | polycyclic aromatic hydrocarbons (EPA list) |
| Pb | lead |
| PCB | polychlorinated biphenyles |
| PCDD | polychlorinated dibenzodioxins |
| PCDF | polychlorinated dibenzofurans |
| PCE | perchloroethylene |
| PCP | pentachlorophenol |
| PM | periculate matter |
| Pt | platinium |
| $qCO_2$ | metabolic quotient |
| Ra | radon |
| Rh | rhodium |
| S | sulfur |
| Sb | antimony |
| Se | selenium |
| SIR | substrate-induced respiration |
| Sn | tin |
| TE | toxicity equivalent |
| Te | technetium |
| TC | total carbon |
| TCDD | tetrachlorodibenzodioxin |
| TCE | tetrachloroethylene |
| TIC | total inorganic carbon |
| TOC | total organic carbon |
| Tl | thallium |
| TPH | total petroleum hydrocarbons |
| TPV | total pore volume |

| | |
|---|---|
| V | vanadium |
| VHC | volatile organic hydrocarbons |
| WRB | World Reference Base |
| Zn | zinc |

# Chapter 1
# Introduction

Many publications which deal with soils in urban environments begin with sentences such as "there is still very little knowledge about soils of the urban environment "or" up to now soil sciences have dealt almost exclusively with soils in agricultural and forested environments." Is this true?

Without a doubt the branch urban soil sciences is still very young compared to the classical soil sciences which traditionally deal with the rural environment. Basically municipal svoils have only been a topic since the middle of the 1970s, particularly in the USA, Germany and Russia. For example, at the Conference of the International Society of Soil Sciences (ISSS) in 1986 in Germany municipal soils in the area of Berlin (West) were presented in detail (Blume 1986). The first well-founded books on this topic dated from the period of the early 1990s, i.e. the books of Bullock and Gregory (1991), which focussed on anthropogenically disturbed soils in United Kingdom and of Craul (1992), which centred on municipal soils in cities in the USA. In the 1990s numerous books of international authors appeared. Most of these were published in the respective national language (Hiller and Meuser 1998; Kollender-Szych et al. 2008).

Generally speaking, the proportion of publications on soils used for urban, industrial and mining purposes has increased in trade journals. This is a consequence of a growing number of research projects in different regions of the world, which varied in their motivation to make urban soils their subject. The ever pressing issues of urbanisation, of land use but, above all, of contamination pushed urban environments increasingly into the focus of science. Working groups formed in various urban agglomerations such as in the Ruhr district (Germany) and in Moscow (Russia), which dealt intensively with the urban locations whose characteristics deviated so strongly from soils of rural environments. Increasing attention was given to urban soils in countries which, by tradition, had experience in soil science research (e.g. in France, in Poland, in Russia, in the USA) and in countries which made the recording and assessment of contaminated locations the focal point (e.g. in Germany and The Netherlands).

We owe it to the initiative of Burghardt (2000) that the first international conference of a newly formed working group of the ISSS, which was devoted to soils of urban and industrial environments and of areas used for traffic, mining and

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military purposes (SUITMA), took place in Essen (Germany) in the year 2000. Further conferences in Nancy (France) in 2003, Cairo (Egypt) in 2005, Nanjing (China) in 2007 and New York (USA) in 2009 followed this one. These conferences documented in an impressive way in how many countries research into urban soils has been carried out. In the meantime investigations into urban soils are being carried out in numerous urban agglomerations. More recently, this is also taking place in developing countries such as China.

Today it can thus be stated that in the meantime a lot of knowledge has been gained on urban soils. The aim of this book is to document the status of the knowledge on hand and to supplement this with the results of the author's own long-term studies in order to provide a comprehensive picture of the physical-chemical qualities of urban soils and, in particular, of their contamination.

In order to place the trigger of the increasing problems in urban environments, the ongoing urbanisation, into its overall historical and economic context, the book begins with a short outline of the basic geographic elements of the urbanisation problem (chapter 2). Chapter 3 deals intensively with the manifold causes of contaminated urban soils such as the influence of dust deposition through industry and traffic, use of fertiliser and sludge but also the influence of former industrial locations and depositions of contaminated materials. The latter aspect leads to the contamination potential of technogenic substrates such as construction debris, slag, ash, garbage, mining waste and various sludges. In Chapter 4 light is shed on these. The aim of the determination key integrated into this Chapter is to give the soil scientists the possibility to recognise such materials when working on site. The different urban soils are presented by means of examples, separated according to Anthrosols and Technosols and assessed with regard to their soil-physical and soil-chemical properties (Chapter 5). The extent and effect of the contamination are strongly influenced by physical and chemical factors. For this reason in Chapter 6 physical characteristics of urban soils such as sealing, compaction and deflation and also chemical characteristics such as pH value, carbon content and chemical binding forms are presented and discussed in relation to the contamination problem in each case. In addition, the nutrient relationships and pedogenesis issues of urban soils are taken up. Chapter 7 deals with the assessment of urban soils. At first, approaches such as the assessment of soil functions, which is presented by means of selected partial functions, are dealt with. However, the assessment of contaminated urban soils is usually carried out on the basis of limit values. For this reason, after a description of the different relevant contaminant pathways, quality standard catalogues and guidelines from numerous countries are presented and discussed comparatively. An outlook including future scenarios complete the book in Chapter 8.

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# Chapter 2
# Geographical Basics

**Abstract** For a better understanding of the current existence of contaminated urban soils some geographical aspects are initially introduced after a definition of the most important terms of reference for this book. The historical development of city growth and its relation to soil contamination is described taking particular consideration of soil contamination of ancient times and the influence of the Industrial Revolution in Europe. Furthermore, the process of urbanization in less developed and developing countries in Africa and Asia and its impact on soil losses and degradation forms a part of this chapter. The theoretical scheme of city structure and urban land-use types in association with their contamination pattern are also points of consideration.

**Keywords** City structure • Historical development • Industrial revolution • Urbanization • Urban land-use types • Urban sprawl

## 2.1 Definitions

Soils are defined as dynamic natural bodies composed of mineral and organic solids, gases, liquids and living organisms which have properties resulting from integrated effects of climate, organisms, parent material and topography over periods of time and which can serve as a medium for plant growth (Brady and Weil 2008).

Urban soils are soils in urban and suburban areas consisting of anthropogenic deposits with natural (mineral, organic) and technogenic materials, formed and modified by cutting, filling, mixing, intrusion of liquids and gases, sealing and contamination (Burghardt 1994; Stroganova et al. 1998). According to the definition urban soils are related to specific areas, namely urban and suburban areas. In this sense, industrial, traffic and mining areas may also belong to the areas of concern. Currently, the abbreviation SUITMA for Soils of Urban, Industrial, Traffic, Mining and Military Areas is frequently used.

Contaminated or polluted soils are soils with physical, chemical and biological materials that can potentially have adverse impacts on human health or environmental

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media. The materials must be undesirable substances which are normally not or to a less extent present in the environmental media like soils (Asante-Duah 1996). The terms 'contaminated' and 'polluted' always involve the assessment of measured concentrations in comparison with either empirical determined thresholds or with background values resulting from naturally occurring concentrations (bedrock concentrations) and non-site-related, ubiquitous concentrations (see Section 3.2).

In this book the term 'contaminated' will not relate to thresholds and man-made quality standards, since the defined thresholds and quality standards applied by environmental agencies vary from country to country (see Section 7.4.2). The non-site related concentrations do not seem to be a suitable level either due to the dependence on site factors such as topography, climate, etc. Alternatively, naturally occurring concentrations based on the continental earth crust could be taken, but in the case of exceeded values an adverse impact on human health is not inevitably present.

In urban geography city size classification includes different definition approaches. Based on the United Nations definition, inhabitants of towns range between 10,000 and 100,000 people, in cities a population of 100,000 to 1 million is supposed. If the population exceeds 1 million inhabitants and the population density is higher than 2,000 inhabitants $km^{-2}$, it is called a metropolis (UN 1993). The terms metropolis, metropolitan area and agglomeration are used synonymously. In this book the terms agglomeration and (big) city are mostly used but not clearly separated.

A metropolis of global scale reveals particular characteristics such as presence of international authorities, economic institutions and private enterprises, broadcasting houses, publishing houses and telecommunication centres of international reputation, as well as famous cultural buildings like theatres and museums, beneficial traffic connections like international airports or big harbours. Moreover, such cities are globally well-known and they have frequently capital status. It should be noted that the boundaries of all areas defined above are politically determined and can subsequently change with reference to urban expansion.

The increase in share of urban population in relation to the total population in the area of concern is termed urbanisation. It is characterized by an increase in the amount of cities, in the population density and in rural-to-city migration. It leads to spatial expansion of city areas and the establishment of agglomerations.

## 2.2 Historical Development of Urbanization

In Europe and North America the process of intensified urbanization started during the Industrial Revolution. Inventions and innovations in originally agriculture-based countries led to a conversion process from the primary sector to the secondary industrial sector. The possibility to smelt iron and other ores with hard coal instead of charcoal, the invention of blast furnaces and other heavy industry equipment powered by steam machines initiated a complete transformation of the industrial status quo. Mechanization in factories and general acceleration of production processes resulted in an increase in produced goods but also in an increasing demand on resources and employees. The coal mining and metal processing industries expanded and population migration into urban and industrial areas occurred.

The process started in the late eighteenth century, firstly in the United Kingdom. In 1850 labour forces in the industry sector already made up 48% of the population. In the nineteenth century countries such as France, Germany and Belgium followed. Between 1870 and 1914 the era of Industrial Revolution can be understood, if the rapid increase of coal and steel production is taken into consideration (Kiesewetter 2006). For instance, in the United Kingdom in 1870 125,500 t of hard and lignite coal were mined, in 1913 292,000 t were registered. In Germany coal mining resulted in 26,400 t in 1870, 43 years later 190,100 t. Between 1860 and 1914 steel production in the United Kingdom rose from 150,000 to 7,790,000 t and in Germany from 50,000 to 18,960,000 t (Table 2.1). In 1900 in Western European countries the labour force of the industrial sector ranged from 29% to 39% of the population. Some decades later in North America same tendencies were observed. To a less extent in Eastern European countries and in some Asian countries a comparable historical development took place.

After the first World War the tertiary sector, the services sector, became more significant, in particular in European and North American countries. Ultimately, in 1990 71% of the labour force was employed in the tertiary sector in USA, 69% in United Kingdom, 64% in France, and 57% in Germany. This sector was strongly connected with improved living standards and a wasteful lifestyle, causing more land consumption particularly in the city periphery. Both the development of industry during the epoch of the Industrial Revolution and the development of the services sector led to increasing urbanization in the developed countries. In the former socialist countries in Europe the development of the tertiary sector started with some delay, but the tendencies were rather similar (Kiesewetter 2006).

**Table 2.1** Development of coal production from 1870 to 2005 and steel production from 1860 to 2007 in different European countries in 1,000 t (Data from different sources, e.g. Kiesewetter 2006, UN 2009, European Navigator 2006, UK Steel 2008)

| | Coal production | | | | | |
|---|---|---|---|---|---|---|
| Country | 1870 | 1890 | 1913 | 1960 | 1992 | 2005 |
| Belgium | 13,700 | 20,400 | 24,400 | 27,100 | 218 | uv |
| France | 12,200 | 25,100 | 37,600 | 57,700 | uv | uv |
| Germany | 26,400 | 70,200 | 190,100 | 132,600 | 72,200 | 28,000 |
| United Kingdom | 125,500 | 185,500 | 292,000 | 196,700 | 84,900 | 20,500 |
| | Steel production | | | | | |
| Country | 1860 | 1890 | 1913 | 1958 | 2007 | |
| Belgium | 3 | 240 | 2,470 | 6,007 | 10,700 | |
| France | 42 | 570 | 4,690 | 14,633 | 19,300 | |
| Germany | 50 | 2,260 | 18,960 | 22,785 | 48,600 | |
| United Kingdom | 150 | 3,640 | 7,790 | uv | 14,300 | |

uv = unknown values

The growth of area occupied by the city can be historically recognized in the capital of Russia, Moscow, with 10,500,000 inhabitants. In 1495 the town covered the area within the Kremlin walls, an area of 0.2 $km^2$, only. The growth slowly continued in the Middle Ages, achieving 18.7 $km^2$ between 1630 and 1640. Just before the Industrial Revolution in 1864 the territory of Moscow covered 72.1 $km^2$. Rapid growth took place during the following decades and when the Moscow Ring Railroad was completely finished in 1917 the territory made up 212.1 $km^2$. At the end of the 1960s 356.7 $km^2$ were occupied by the agglomeration, continuing urban sprawl. Today, the modern city of Moscow covers an area of 1,091.0 $km^2$.

Intensive soil surveys corresponded with the historical development. It was almost impossible to detect the original soil of Moscow, mainly soddy-podsolic soils and glacial, glaciofluvial and mantle deposits, in the upper horizons. Above the buried natural soil deposited material with a thickness of approximately 2 m and composed of clayey texture with a high humus content and timber and saw residues was present. This dates back to the twelfth to sixteenth century, when the city was constructed of wood. This layer was overlaid by another anthropogenic horizon which was mainly sandy and poor in humus that was enriched with construction rubble such as bricks and stones. The layer relates to the seventeenth to nineteenth century, when intensive construction activities occurred. Finally, a modern cultural horizon containing modern construction debris like asphalt and concrete covered the last one. The anthropogenically deposited layers ranged in ancient Moscow between 2 and 3 m, in wet depressions it was possible to find thicknesses of 7–10 m and the maximum thickness was 20 m.

Research projects in downtown Moscow demonstrated that a historical contamination has to be taken into account when assessing the current contamination level. In deep soil layers related to the fifteenth to nineteenth centuries an accumulation of trace elements such as arsenic, copper, lead, and zinc has been found, reflecting metal working, leather treatment and other commercial activities of former times. Copper, for instance, ranged between 43 and 359 mg $kg^{-1}$, lead between 13 and 265 mg $kg^{-1}$, and zinc between 76 and 265 mg $kg^{-1}$, exceeding clearly the background levels. Table 2.2 presents a soil profile considering the depth 0–220 cm that was possible to relate to the period of time in which the artificial deposits happened. In addition, the upper layer seemed to be strongly influenced, resulting from current contamination sources.

The relatively high contaminant level is confirmed when considering areas of long-term anthropogenic use. In the catchment area of the Russian capital Moscow some areas established by mankind for a long time have been analyzed in detail. For example, the Rozhdestvenskiy Convent, established in 1386, supplied very high values in subsoils (Pb: 33–373 mg $kg^{-1}$, Zn: 55–492 mg $kg^{-1}$). It was possible to detect similar results in the Zachat'evskiy Convent, established in 1623 (Pb: 40–873 mg $kg^{-1}$, Zn: 36–597 mg $kg^{-1}$) as well as the Yusupovskiy Park, established in the seventeenth century (Pb: 131–245 mg $kg^{-1}$, Zn: 41–151 mg $kg^{-1}$). Obviously, lead and zinc are typical parameters, proving the long-term human influence, while Cd, Cu, Co, Ni, and V revealed low results at the same sites (Stroganova et al. 1998).

**Table 2.2** Heavy metal content (mg kg$^{-1}$) in deposited, cultural layers of a soil profile in downtown Moscow associated with the period of time of depositing (Data from Stroganova et al. 1998)

| Depth, cm | Century | As, mg kg$^{-1}$ | Cr, mg kg$^{-1}$ | Cu, mg kg$^{-1}$ | Ni, mg kg$^{-1}$ | Pb, mg kg$^{-1}$ | Zn, mg kg$^{-1}$ |
|---|---|---|---|---|---|---|---|
| 0–25 | Beginning of the 20th | 74 | 98 | 197 | 24 | 1,321 | 1,552 |
| 25–70 | 18th–19th | 36 | 90 | 149 | 29 | 242 | 192 |
| 70–100 | 18th | 16 | 61 | 88 | 24 | 103 | 151 |
| 100–115 | 18th | 13 | 50 | 64 | 12 | 146 | 128 |
| 115–145 | 17th–18th | 25 | 92 | 43 | 23 | 152 | 107 |
| 145–160 | 17th | 33 | 92 | 80 | 28 | 265 | 256 |
| 160–170 | 17th | 16 | 83 | 83 | 35 | 55 | 134 |
| 170–195 | 17th | 18 | 91 | 81 | 36 | 54 | 104 |
| 195–220 | 16th | 15 | 81 | 359 | 29 | 89 | 398 |

Metal extraction method: $HNO_3$

The described context that soils function as records of urban development was also found outside of Europe. In Nanjing (China) with a population of 5,300,000 soils were sampled at every 5 cm for heavy metal determination purposes. Charcoal from several layers was found and dated using $^{14}C$ to recognize archaeological horizons termed cultural layers. All cultural layers started about 1,700 years ago above the original loess indicated high concentrations of Cu, Pb and Zn but not of Co, Cr and Ni. The lead content varied from 100 mg kg$^{-1}$ to more than 2,000 mg kg$^{-1}$, Cu varied from about 100 mg kg$^{-1}$ to more than 5,000 mg kg$^{-1}$, Zn from 70 mg kg$^{-1}$ to more than 2,000 mg kg$^{-1}$. There were several historical periods in which heavy metals accumulated, such as ancient ore smelting and use of materials containing heavy metals for handicraft manufacture. It was concluded that the source of Pb in cultural layers might come from lead ores of former times. Similar interpretation was made related to copper and zinc (Zhang et al. 2005a).

In the vicinity of areas with long-term metal mining processes the accumulation of metals in soils can reach the highest levels by far. Such historical accumulation areas are, for example, in Eastern Turkey (Ergani Maden Copper Mine close to the river Tigris), where copper was extracted 9,000 years ago. Other examples of ancient mining activities like the copper ore mining and smelting regions of Iran, Iraq and Israel were reported. In the Bronze Age, 4,600 years ago, tin and copper were extracted and smelted together with charcoal. Later on, 2,000 years ago, iron smelting became widespread. The metal extraction activities reached their peak in Roman times in the form of weapon manufacturing (bronze, iron) as well as the production of more peaceful things such as ornaments (copper, gold, silver), plumbing and coffins (lead), coinage (bronze), etc. Moreover, some minerals were used as pigments in order to colour ceramics and glass (cobalt, nickel). The soil contamination derived from historical ore mining processes, however, was usually limited to the small-scale sites of extraction and processing (Evans 2005).

The different sources of contaminants which are deposited onto the soil surface have an impact on the long-term contaminant uptake of long living plants like old trees. Accordingly, trees can be witnesses of historical development as well. The lead content in wood of the American beech trees in Virginia, USA showed an increasing trend, in particular a sharp increase in the late 1800th century, when the industrialization began (increase from less than 100 to more than 450 μg $kg^{-1}$). The high level continued, indicating a visible second peak in the 1950s (300–400 μg $kg^{-1}$) when leaded petrol was burnt in vehicles. After the 1990s with the beginning of unleaded gasoline utilization, the Pb values in the wood of the trees were reduced again reaching values less than 300 μg $kg^{-1}$ (Pierzynski et al. 2005).

Another witness of historical development is the type and distribution of anthropogenic artefacts present in soil profiles. Human behaviour has left its fingerprint in urban history, and in turn contamination. Beginning in the Middle Ages inhabitants of urban areas threw organic garbage, litter and other organic materials away next to their houses. This permanent activity produced waste materials without any consideration of sanitation. Organic material was incorporated into the soil, together with some mineral artefacts.

The production of urban wastes changed drastically in the eighteenth century, the beginning of the period of the Industrial Revolution. Industrial waste was increasingly thrown away, often causing soil contamination based on the kind of material such as ash and slag. The opportunity to use hard coal for smelting iron led to an expansion of iron production for domestic articles and particularly for machinery and manufacture. The mechanized production opened the way for the establishment of large-scale factories, which were built first in the United Kingdom and some years or decades later in many countries all over the world. The flourishing industry needed more and more metals extracted, resulting in increasing prospecting for mineral resources. Hence, the exploitation was intensified, especially in the nineteenth century, causing large devastated areas in the developing countries like Belgium, France, Germany and the United Kingdom, on the one hand, and increasing worldwide exploitation in countries which supply raw materials, on the other hand. Consequently, soil contamination caused by metal extraction took place in the developing countries to a greater extent and, additionally, in, for instance, African countries whose economies were originally based on agriculture (and still are today) (Evans 2005).

Increasing urbanisation since the Industrial Revolution changed the residential areas considerably as well. Construction operations led to the production of construction debris and wastes that were deposited close to the buildings after the humic topsoil had been excavated. The stockpiled topsoil, however, returned after building operations. In city areas only little natural soil remained, instead mixtures of soil and construction debris were refilled. If the area was not redeveloped well, apart from construction rubble foundations, concrete floors and underground cellars can be discovered as well, leading to rehabilitation difficulties later on (see Section 5.4.1). Industrial wastes in urban areas have been masked by distinct anthropogenic influences of the period after the First World War. The frequent land use changes associated with excavation and fill operations altered the soil below the surface time and time again.

Long-term urban utilization led to increasing contaminant values. Different contaminant inputs such as waste additions, long-term application of ash and

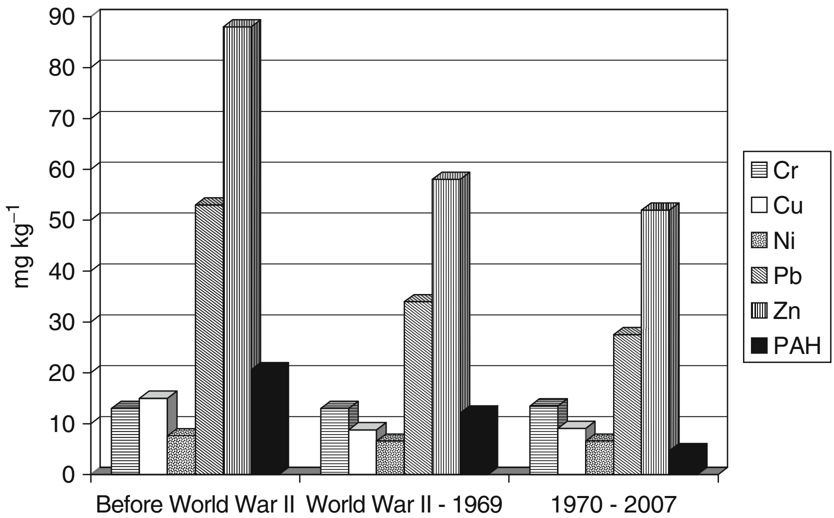


**Fig. 2.1** Topsoil concentration of some contaminants related to the age of buildings constructed nearby in Münster, Germany. Investigated areas of different land-use types: 285. Sampling depth: ranging from 0–10 to 0–35 cm. Metal extraction method: aqua regia

problematic organic manures and the atmospheric deposition left their fingerprints in the long course of time. In the past, it was documented that the lead concentrations in garden topsoils increased with the age of the houses present. In Münster (Germany) with its population of 270,000 soil contamination maps were produced taking the age of residential and industrial areas into consideration. The city is designated as an administrative area and to a less extent as an industrially influenced city but intensive vehicle traffic and emissions of some industrial complexes may enhance the topsoil concentration in any case. Residential areas were separated into three classes, firstly constructive phases before the Second World War, secondly between the Second World War and 1969 and thirdly between 1970 and the realization of the research project in 2007. For some parameters such as Pb and Zn the contaminant concentration was related to the age of the building construction phase (Fig. 2.1). Some elements did not react or only to a very small extent (e.g. Cr, Ni). Organic pollutants like the Polycyclic Aromatic Hydrocarbons ($PAH_{EPA}$) tended to reveal a slight dependence on the period of construction.

## 2.3 Present Urbanization Process

The population living in an urban environment (cities, agglomerations) shows a tendency to grow and simultaneously the degree of urbanisation is becoming higher. In general, the degree of urbanization exhibits extreme differences. In the developed countries such as France, Germany, United Kingdom and USA it is higher than 70%, in less developed and developing countries such as Kenya and India the values

**Table 2.3** Degree of urbanization (%) in different countries in the course of time (Data from World Gazetteer 2008)

| Country | 1950 | 1970 | 1990 | 2005 | 2020 (prognosis) |
|---|---|---|---|---|---|
| France | 55 | 71 | 74 | 77 | 80 |
| Germany | 54 | 72 | 73 | 73 | 76 |
| United Kingdom | 55 | 77 | 89 | 90 | 91 |
| USA | 64 | 74 | 75 | 81 | 85 |
| Brazil | 36 | 56 | 75 | 84 | 90 |
| China | 13 | 17 | 27 | 40 | 53 |
| India | 17 | 20 | 26 | 29 | 34 |
| Russia | 44 | 63 | 73 | 73 | 74 |
| Kenya | 6 | 10 | 18 | 21 | 27 |
| Nigeria | 10 | 23 | 35 | 46 | 57 |

can fall below 30% (Table 2.3). The differences, however, have reduced in the last few decades, and they are reducing continuously. In principle, the following tendencies can be ascertained:

- The current process of urbanization in some developing countries such as China and India is much higher than the urbanisation process which happened in developed countries a long time ago.
- The current urbanization in less developed countries is caused by rural-to-urban migration combined with the increase in the birth rate.
- The urbanization based on migration relates mainly to the booming agglomerations, because the job opportunities seem to be more favourable than in smaller cities; the development led to overurbanized mega-cities that are nowadays concentrated in less developed and developing countries (e.g. Mexico City, Rio de Janeiro, Sao Paulo, Buenos Aires, Tehran, Cairo, Lagos, Lahore, New-Delhi, Karachi, Mumbai, Hyderabad, Dhaka, Kolkatta, Jakarta, Beijing, Tianjin, Shang Hai, Manila).

Nine out of ten of the biggest cities in the world are located in less developed and developing countries. In particular, in developing countries the population density of the agglomerations is extremely high. For instance, in Mumbai (India) with 21,300,000 inhabitants the density is 47,900 $km^{-2}$, while among the European agglomerations the capital of the United Kingdom, London, with 13,200,000 inhabitants has only 8,400 people $km^{-2}$. Because of the increasing population uncontrolled urban sprawl occurs. In less developed countries the enormous sprawl is linked to poor housing in slums.

As mentioned before, in the developed European and North American countries the process of urbanisation lasted a relatively long period of time, so that the city expansion occurred at least in partly controlled conditions. In the last 2–3 decades in East Asian countries, particularly in China and India, the industrialization and urbanization process occurred at a speed the world has never seen before. The extremely rapid increase in population in cities due to rural-to-city migration is continuously leading to enhanced population density and huge urban sprawl.

The development can be impressively observed in the Yangtze Delta in China, where an urban expansion of 322 km$^2$ each year has taken place since 1984. Two Chinese cities called Nanjing and Suzhou with a population of 5,800,000 inhabitants have been investigated with reference to urban sprawl based on satellite image comparison for the period from 1984 to 2003. The yearly rate of increase of the urban area varied between about 5% and 12% until the year 2000 and afterwards the rates exploded, in particular in Suzhou, indicating an increase of approximately 25% from 2000 to 2003 (Fig. 2.2). Contrary to the development caused by the Industrial Revolution in Europe, this extreme urban sprawl was not able to prevent unplanned and environmentally unfriendly city expansion and land consumption. Subsequently, in Nanjing, for instance, the loss of soils for construction purposes covered fertile and agriculturally valuable soils such as Argosols and Cambisols (Table 2.4). In county towns and suburbs located on the periphery of Suzhou most of the areas occupied by urban sprawl belonged to the good-quality soils used for

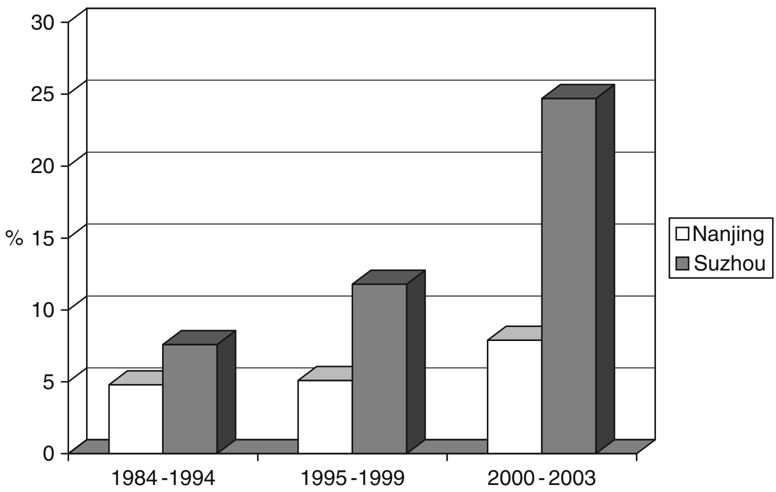


**Fig. 2.2** Yearly rate of increase of the urban area in Nanjing and Suzhou, China (Data from Zhang et al. 2005b, 2007)

**Table 2.4** Lost percentage of soils in Nanjing, China from 1984 to 2003 (Data from Zhang et al. 2007)

| Soil order | Characteristics | Lost (%) |
|---|---|---|
| Anthrosols | Long-term management of soil for agriculture and other anthropogenic uses | 39.9 |
| Argosols | Illivial accumulation of clay, udic moisture regime | 17.8 |
| Cambisols | Partly low grade of development, udic moisture regime | 41.2 |
| Gleysols | Redoximorphic conditions, stagnic moisture regime | 0.1 |
| Primosols | Lithologic character | 1.0 |

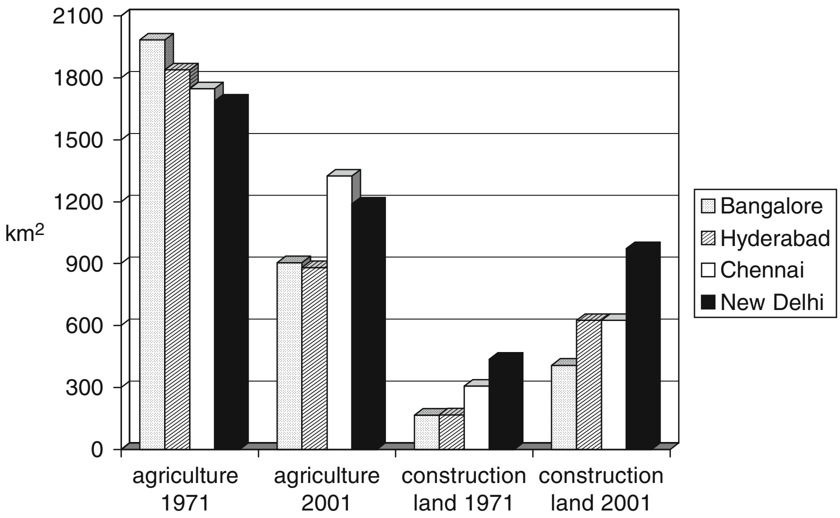


**Fig. 2.3** Development of agricultural and construction land use from 1971 to 2001 in four Indian agglomerations (Data from Ramachandran 2003)

agriculture. The ratio between such soils and less fertile and degraded soils was 95:1. In the city itself the ratio was nearly 1:1. Apparently, half of the new construction areas occupied brownfields and abandoned land (Zhang et al. 2005b, 2007).

The loss for construction purposes relates mainly to the agricultural sector. Thus, the conflict between urban and agricultural land use became a subject of controversy. The example of four Indian cities impressively showed the expansion of construction land at the expense of cropland and pastures (Fig. 2.3) (Ramachandran 2003).

The rapid process of urbanization is not only limited to the East Asian countries mentioned. In other spatially more limited developing regions of the world we can observe comparable tendencies. The objective of a study covered the extent of changes in land use in East Delta, surrounded by Port Said and Cairo, Egypt with reference to both the expansion of agriculture over non-arable lands and the urbanization over arable lands. The study made use of maps in 1952 and recent Landsat TM images in 1989, 1995 and 2000. There was considerable urban expansion between 1952 and 2000. In 1952 the urban area of selected settlements (small and medium-sized towns) varied between 0.11 km² (in El-Tell El-Kebir) and 8.44 km². In 2000 the urban area increased, ranging from 0.96 km² (in El-Tell El-Kebir) to as much as 1,442 km² (in Ismailia, 1,000,000 inhabitants). On the contrary, areas were transferred to new reclamation desert land consisting of cultivated land, bare land and other land-use types. In a pilot area, investigated in detail, the changes in land use were designated as area loss to urbanization within the old agricultural Delta land to about 94.08 km² and the area of the new reclaimed desert land to about 267.54 km². Thus, the net result was about 173.46 km². Two aspects, however, should be considered when calculating the net land use development. Firstly, as shown in the example mentioned, the quality of the lost land was not better than the quality of the gained land from the agricultural point of view. Secondly, it cannot be expected that the reclaimed land may always compensate the area urbanized (Abdel-Salam et al. 2005).

Another example presented deals with the development of East African countries. In Kisumu, Kenya, today a densely populated town with one of the highest population growth rates in East Africa, there was the problem of land consumption and limited supply of land for urban expansion. Although the surroundings of Kisumu included rain-fed mixed farming, large-scale sugar cane farming and fishing in the neighbouring lake belt, the agriculture did not provide enough livelihood support as a result of frequent drought periods and flood occurrences in the floodplains. Consequently, a rural–urban migration of the population occurred leading to unplanned and uncontrolled urban expansion. In 1986 the population counted 200,000 inhabitants and in 2013 about 800,000 are already expected (D'Costa and Omoto 2000a).

Rapid population growth leads to uncontrolled urban expansion impacting both urban and peri-urban areas. The expansion is related to a lack of accurate and timely data on soil quality promoting land use planning. Hence, in some regions the limits of urban expansion might become visible in the near future. A good example of such a situation is the city of Nakuru, Kenya with a current population of about 500,000 citizens. Processing and marketing of agricultural products, tourism, manufacturing, commercial and administrative services are the economic pillars of the city. Before 1900, the area was part of savannah grassland and later on the first farms were introduced. Urbanization started in the 1980s and has continued rapidly. In 2020 the population might reach more than 1 million people. Approximately 70% of the population live in primitive fragile settlements with pit latrines. Household waste is disposed of close to or in the residential housing, drinking-water is taken from boreholes. The opportunities for city expansion have been investigated by subdividing the territory into five units. Unit 1 is a mountainous region with steep slopes, shallow soils and bare rocks unsuited for urban development. In future, the area will only be used for forest and tourism because of the wild animals living there (e.g. lions). The soils of unit 2 are volcanic deposits, well drained, deep and fertile and subsequently preferentially used for agriculture. But, in general, in this unit settlements can probably be established by converting agricultural land into urban land-use types systematically. Unit 3 consists of undifferentiated volcanic rock and deposits, causing very frequent land subsidence and ground collapse. Accordingly, serious problems would arise, if urban development took place. Unit 4 which consists of old lake plains is already covered by the present town and densely urbanized. Unit 5 around Lake Nakuru represents the famous protected National Park, particularly well-known for its flamingoes. Urbanization is generally not feasible and uses only compatible with wildlife conservation are allowed. In conclusion, the urban expansion was strongly restricted and was mainly concentrated in unit 2. Since this unit was the only one offering good quality soils for agriculture like vitric Andosols, urban expansion and agricultural use came into conflict with each other (D'Costa and Omoto 2000b).

Generally speaking, the urbanization caused by rapid increase in population means a major problem in less developed and developing countries, because city structures are not well planned. This leads to cities growing in size and population due to uncontrolled migration and therefore a deterioration of already existing problems. Such problems are, for instance, inadequate waste disposal systems creating contaminated soils and water resources. In spite of some rules and protection acts in developing countries like India and China, effective implementation is difficult to realize because of various political and financial problems as well as slow government machinery including lack of co-ordination among different authorities involved (bureaucracy).

In industrialized, well-developed countries different reasons may exist to explain urbanization. One of the most important reasons for land consumption and urban expansion is linked to the progressive development of tourism. The impact of tourism is very serious in the Mediterranean basin, where urban and tourist settlements have involved the main coastal plains (Fig. 2.4). The losses on the north–eastern coast of the island of Sardinia, 1,700,000 inhabitants, with the municipalities of Arzachena, Golfo Aranci, Loiri Porto San Paolo, Olbia, and Palau, Italy, amounted to about 80,000 ha including the main buildings and road networks during a period from 1958 until 1998. The results showed large soil consumption; about 17% of this loss has involved land suitable for agricultural use. Of interest are the observed soil losses that were not caused by a natural rise in population but were driven by the strong tourism development. Additionally, it should be kept in mind that a large percentage of the new buildings and settlements are inhabited only during the summer season (Madrau et al. 2005).

**Fig. 2.4** Tourist beach lifestyle in Alanya, Turkey; at seaside of Mediterranean countries such as Spain, Italy, Greece and Turkey considerable percentages are built-up areas for tourism purposes

A new phenomenon called shrinking cities occurred in the last few decades on the East coast of the USA, in Europe and in the South-Eastern part of Asia. Detroit (USA) with currently 820,000 inhabitants, for example, is a major manufacturing centre serving as home of the American automobile companies. It is accustomed to the economic cycles of the auto industry. Until the middle of the twentieth century car manufacturing increased, leading to a rapid migration of people from abroad for working purposes to the automobile factories. In 1900 the population was 190,000 whereas in 1960 it was 1,700,000 people. Consequently, urban sprawl occurred, especially in the suburbs, where workers bought houses. Growth at all environmental costs happened. A rise in automated manufacturing and competition from Asian countries in the areas of production and labour costs caused a steady decrease and transformation of the jobs in the region. Although new technologies such as biotechnology and information technology developed and transformation from manufacturing to service industries such as internet companies and entertainment institutions occurred, the unemployment rate increased. Occurrences in the first decade of the present century like a recession in 2001 and the 9/11 terror attack exacerbated the situation additionally. As a consequence the city began shrinking and in 2007 the population was only 950,000. This resulted in an increasing number of brownfields, more and more vacant accommodation and ultimately an increasing tendency towards poor housing (Vogel 2005).

It was possible to observe shrinking to a spectacular extent in former socialist European countries due to the transformation of the whole economic system. In Leipzig (Germany) with a current population of 510,000 inhabitants, a city in the former German Democratic Republic (GDR), the population grew until 1930, reaching a maximum of 700,000 people. Urbanization took place mainly in the suburbs surrounding the city centre. After the Second World War the city authority was ignorant of the problem of urban sprawl. In particular, in the suburbs many typical socialist buildings were constructed. After 1989 urban sprawl continued for a short time, because new enterprise zones and shopping malls were built to accommodate the expected western lifestyle of the population. Actually, economic shrinkage took place and a high percentage of the city population tended to migrate to the more developed Western part of Germany. In 2003 the population was approximately 500,000 and because of more than 55,000 empty flats some parts of the city seemed more or less like suburbs of a ghost city (Banzhaf and Kindler 2000). Both examples demonstrate that shrinking cities may cause a high percentage of abandoned and derelict land as well as brownfields in future.

## 2.4 City Structures and Their Impacts on Contamination

Figure 2.5 presents the idealized city structure. Cities are built-up in concentric order. The city centre is surrounded by a suburban zone containing residential and industrial suburbs or mixtures of both. A second outside ring called urban fringe is still influenced by urbanization to a lesser extent and with lower population density.

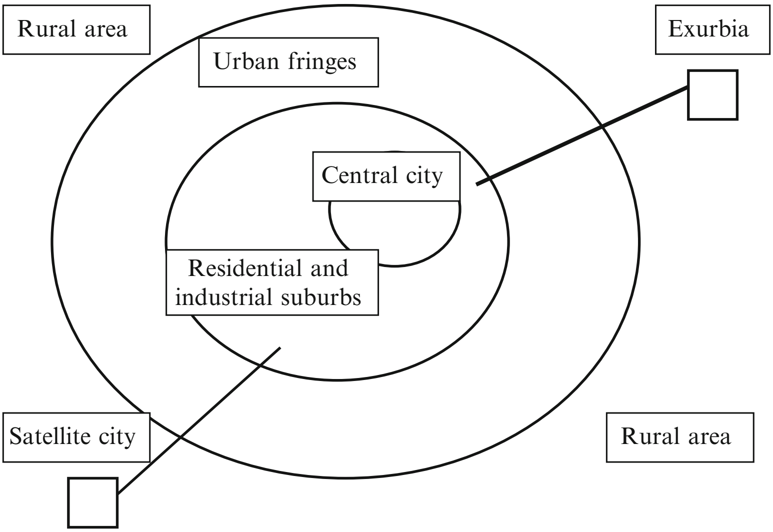


**Fig. 2.5** Principle of city structure

The distance to highly urbanized satellite cities and small urbanized exurbian settlements (towns) is relatively far. A close connection to the city centre and the suburbs, however, may exist due to migration of employees and shopping opportunities.

In contrast to rural areas the diversity of land-use types is much higher in urban environments. In rural areas the diversity is mainly restricted to cropland, pastures and forests, interrupted by farms, villages, water bodies and nature reserves. In urban environments the diversity involves many land-use types that are listed in Table 2.5. Urban land-use types are illustrated in Fig. 2.6a–g.

The use diversity rises at the outskirts and falls in the city centre. It is caused by the diversity of natural and anthropogenic soils on the city periphery in contrast to the city itself, which consists predominantly of man-made soils only. Moreover, the size of different land-use types decreases in the urban environment compared with large-scale cropland, pastures and woodland, leading to a spatial heterogeneity and to structural and material variability in urbanized areas (Fig. 2.7a). The population density increases from the outskirts to the central city apart from the satellite cities, which can be even located in the periphery of the city (Fig. 2.7b) (Burghardt 1994).

The current land-use type has an influence on soil contamination. In Shenzhen (China) with its 12,000,000 inhabitants 79 soil samples (0–20 cm depth) of different types, namely park, road greenbelts, residential areas and municipal parks were collected in the central district. The soils of the urban green space were to some extent polluted by Cu, Zn, and Pb, but seriously polluted by Cd. Spatially, Cu and Zn concentrations were highest in the residential area, and lowest in the municipal parks. The concentrations of Cd and Pb were highest in soils of roadside green space (see Section 3.3.1). Accordingly, a relationship between land-use type and metal content in the uppermost part of the soil has been found. The relationship

**Table 2.5** Land-use types in the urban environment

| Land-use types | | Unsealed area |
|---|---|---|
| Residential area | Exclusive residents | Garden, park |
| | One-family houses | Garden |
| | Semi-detached houses | Garden |
| | Terraced houses | Garden |
| | High rise buildings | Green buffer |
| | Perimeter development | Garden, courtyard |
| | Mixed housing area | Courtyard, Green buffer |
| Industrial and commercial area | Industrial plots, factories | Courtyard |
| | Manufacturing and processing trade area | Courtyard, park |
| | Shopping malls | Mall plantation |
| | Yards | Unsealed portion |
| Special building area | Administrative institutions | Green buffer, park |
| | Universities and research institutions | Green buffer, park |
| | Education institutions | Green buffer, park |
| | Utility and disposal facilities | Green buffer |
| Disposal area | Dumps | Total area |
| | Sludge fields | Total area |
| | Mining heaps | Total area |
| Traffic area | Roads and highways | Roadside green belt |
| | Squares | Square plantation |
| | Car parks | Car park plantation |
| | Railway lines and operating areas | Total area |
| | Airports | Meadows and lawns |
| | Harbours | Harbour plantation |
| Recreational area | Playgrounds | Unsealed portion |
| | Sports fields | Unsealed portion |
| | Allotments | Unsealed portion |
| | Camping sites | Unsealed portion |
| | Cemeteries and churchyards | Unsealed portion |
| Urban agricultural area | Crop areas | Total area |
| | Nurseries and greenhouses | Unsealed portion |
| | Pastures and meadows | Total area |
| | Orchards | Total area |
| | Vineyards | Total area |
| Urban forest | Forest | Total area |
| | Woodland | Total area |
| | Afforestation area | Total area |
| Water bodies | Rivers and canals | Bank |
| | Lakes and ponds | Shore |
| Nature reserves | Peatland and swamps | Total area |
| | Coastlines and beaches | Total area |
| | Heathland | Total area |
| | Rock area | Total area |
| Abandoned land | Residential brownfields | Unsealed portion |
| | Industrial brownfields | Unsealed portion |
| | Traffic brownfields | Unsealed portion |

**Fig. 2.6** (**a**) Typical urban land-use type: high rise buildings with green buffer in the city centre of Shang Hai, China. (**b**) Typical urban land-use type: perimeter development with few green buffers restricted to inner courtyards and roadsides; view from Eiffel tower, Paris, France. (**c**) Typical urban land-use type: blocks of flats with small green areas alongside the roads in Manhattan, New York, USA. (**d**) Typical urban land-use type: mixed housing areas with green buffers and parks partly built in the socialist era in Most, Czech Republic. (**e**) Typical urban land-use type: mixed housing area with green buffer in Mombasa, Kenya. (**f**) Typical urban land-use type: exclusive residents of Italian style in Nanjing, China. (**g**) Typical urban land-use type: allotment in Essen, Ruhr area, Germany

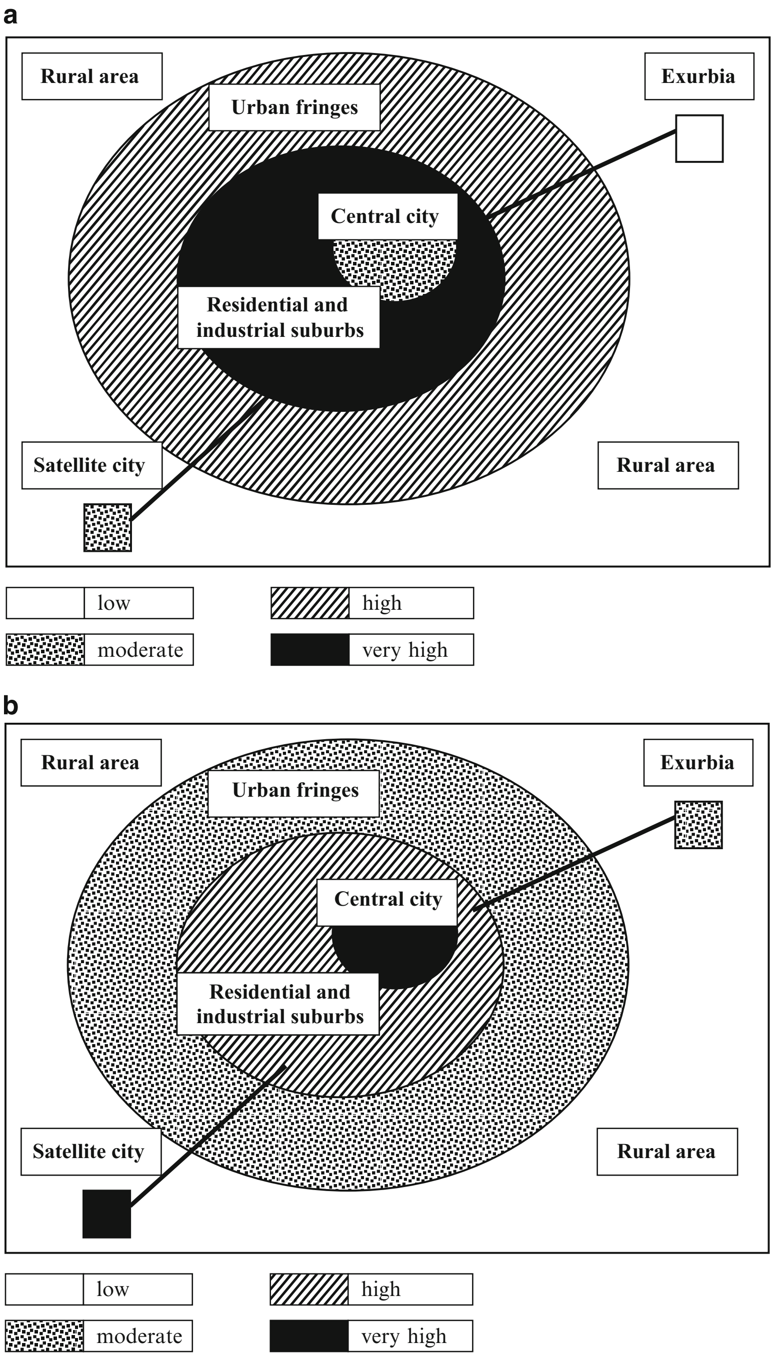


**Fig. 2.7** (**a**) Diagram of the amount of land-use types. (**b**) Diagram of the population density in relation to the city structure

**Table 2.6** Results from pollutant analyses of topsoils (0–5 cm) (mg $kg^{-1}$, μg $kg^{-1}$ for PCB) in Bucharest, Romania, related to different land-use types in city area (Data from Lacatusu et al. 2000)

| | Park soils (n = 13) | | | Vegetable garden soils (suburbs) (n = 9) | | | Street area soils (n = 10) | | |
|---|---|---|---|---|---|---|---|---|---|
| | Mean | Min | Max | Mean | Min | Max | Mean | Min | Max |
| Cd | **0.4** | 0.3 | 0.6 | **2.8** | 1.9 | 3.8 | **2.2** | 1.1 | 4.6 |
| Co | **9** | 6 | 13 | **25** | 19 | 35 | **23** | 15 | 31 |
| Cr | **18** | 11 | 29 | **31** | 23 | 40 | **29** | 18 | 41 |
| Cu | **14** | 8 | 24 | **32** | 21 | 57 | **51** | 27 | 86 |
| Ni | **20** | 11 | 29 | **41** | 28 | 54 | **37** | 14 | 50 |
| Pb | **11** | 5 | 20 | **54** | 19 | 127 | **119** | 27 | 359 |
| Zn | **59** | 36 | 110 | **173** | 126 | 240 | **165** | 70 | 404 |
| $PCB_{Ballschmiter}$ | **4.2** | 1.1 | 9.6 | **2.7** | 1.2 | 5.1 | **29.1** | 1.6 | 110.2 |
| PAH-13 | **8.4** | 1.7 | 24.4 | **18.7** | 14.6 | 24.6 | **15.9** | 2.7 | 45.8 |

Metal extraction method: Hydrochloric acid

revealed clearer tendencies than were discovered between soil properties like texture and organic matter content and pollutants (Lu et al. 2005).

Topsoil (0–5 cm) analyses carried out in Bucharest (capital of Romania) with a population of 2,000,000 included a comparison of distinct land-use types. In the last decade the industry expanded and in spite of modernizing an increase in pollutant emissions generated by the chemical industry, machine factories, power stations and car traffic occurred. In the suburbs new houses with small gardens were established, which received pesticides like organo-chlorinated insecticides. These sites were investigated and compared to soils located in street areas and parks (Table 2.6). With regard to the heavy metals park soils showed lower concentrations than the other sites under consideration. The results from both garden soils and soils nearby streets did not indicate any differences. Polychlorinated Biphenyls (PCB) indicated generally low values except for street soils, while the group of Polycyclic Aromatic Hydrocarbons (PAH, 13 detected congeners) was associated with a relatively high level. The park topsoils seemed exceptionally to be moderately influenced (Lacatusu et al. 2000).

The example of Adelaide (Australia) underlines the pollution differences between urban and rural soils in the same catchment area. Ranges in values of the metropolitan area and the agricultural hinterland were comparable but urban soils resulted several times in much higher Pb and Zn values. Furthermore, the bioavailable Cu, Pb and Zn concentrations analysed by EDTA extraction were three to four times higher in the urban area compared with the rural locality (Table 2.7) (Tiller 1992).

In urban areas a lot of isolated sites with an industrial history indicate extremely high values, termed hot spots. The example of Australian cities is compatible with similar results yielded in big cities worldwide. In Table 2.8 mean and range of heavy metals investigated in parks and home gardens as well as localized concentrations found at industrial sites, dumpsites and areas with accidental spillage are

**Table 2.7** Metal results (mg kg$^{-1}$) from topsoil investigations of Adelaide city soils and soils of the adjacent countryside called Mt Lofty Ranges (Data from Tiller 1992)

| | Adelaide city | | | | Mt Lofty Ranges | | | |
|---|---|---|---|---|---|---|---|---|
| | Aqua regia extraction | | EDTA extraction | | Aqua regia extraction | | EDTA extraction | |
| Element | Mean | Range | Mean | Range | Mean | Range | Mean | Range |
| As | **1.8** | 0.2–16 | **–** | – | **3.9** | nd-8 | **–** | – |
| Cd | **0.4** | 0.1–4.3 | **0.33** | 0.03–5.0 | **nd** | nd | **0.18** | 0.01–0.73 |
| Cr | **11.4** | 2–31 | **–** | – | **27** | 3–110 | **–** | – |
| Cu | **19.7** | 2–138 | **15** | 0.9–118 | **15** | 1–59 | **3.3** | 0.25–19 |
| Pb | **64** | 5–212 | **97** | 5–1,450 | **21** | 2–160 | **12** | 0.4–74 |
| Ni | **14.9** | 2–41 | **–** | – | **9.8** | 2–28 | **–** | – |
| Zn | **80** | 12–421 | **67** | 2–750 | **27** | 4–200 | **6** | 0.3–52 |

nd = not detectable

presented. Thus, the difference between city areas influenced only by atmospheric deposition and water transport and the small-scale sites influenced by industrial factors is obvious. The latter are formally registered under legislation. In Australia most of the registered sites concerned with heavy metals apart from Sydney, where the reverse is true and organic chemicals play the major role (Tiller 1992).

The relation between typical urban land-use type and chemical soil characteristics was systematically analysed in Baltimore (USA) with a population of 800,000. The city is historically an industrial one traversed by several highways and amounting to 31 facilities with chemical atmospheric release, including Cr, Cu, Ni, and Zn. The topsoils were sampled using a grid with 122 plots randomly located. Different land-use types such as commercial and transportation areas, industrial areas, forest, parks and golf courses, residential and institutional areas were considered. With reference to heavy metals no statistically significant differences among land uses were found. It was, however, possible to describe tendencies (Table 2.9). Industrial and residential sites, for instance, tended to show enhanced Cr, Pb and Zn values. Parameters like cobalt, nickel and vanadium did not show any differences. Apparently, at the scale of this sampling intensity there were no significant differences.

Other parameters analysed reacted more clearly with reference to the land-use types. Sodium, for example, showed the statistically significant highest concentrations at transportation sites. Calcium reacted in the same way, because the concrete of transportation routes is a source of alkaline, Ca-rich dust to the adjacent areas (see Section 3.3.1). Potassium indicated very low values for woodland in comparison with other types investigated. In general, nutrient parameters revealed the clearest differences resulting from turf management, e.g. liming and fertilizing. Consequently, unmanaged forest soils showed significantly low potassium and phosphorus values, since they probably received fewer added elements (Pouyat et al. 2007).

With reference to organic pollutants close statistical connections between contamination level and land-use type appears to be problematical. In Seville (Spain) with 700,000 inhabitants 15 Polycyclic Aromatic Hydrocarbons (PAH) of 41 top soils (0–10 cm) under different land use (garden, roadside, riverbank and agricultural allotment) were selected. The results of the sum of 15 PAH in Seville soils

**Table 2.8** Metal concentrations (mg $kg^{-1}$, %) in park and home garden topsoils of the Australian cities Hobart, Melbourne and Sydney (ranges) compared with maximum values found at profiles of industrial sites in different Australian cities (Data from Tiller 1992)

| Location | | As | Cd | Cr | Cu | Hg | Ni | Pb | Sb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| Hobart | **Range** | **2–45** | **0.5–164** | **2–94** | **1–571** | **<0.1–3.9** | **2–130** | **<1–2,200** | **<0.2–2.2** | **18–2.2%** |
| | Hot spot | 112 | 512 | 94 | 675 | 5.6 | 130 | 8,300 | 5.3 | 10.1% |
| Melbourne | **Range** | **<1–8.1** | **–** | **3–43** | **<1–38** | **–** | **<2–50** | **<1–39** | **–** | **<2–42** |
| | Hot spot | 530 | 22 | 10.5 % | 22 % | 26 | 580 | 10.4 % | 470 | 8,100 |
| Sydney | **Range** | **<1–11** | **<0.1–0.7** | **<2–380** | **2–200** | **<0.1–0.3** | **<1–180** | **3–170** | **–** | **2–450** |
| | Hot spot | 40 | 5 | – | 360 | 17 | – | 5,100 | – | 1.2 % |

Extraction method: aqua regia

**Table 2.9** Mean surface soil properties (0–10 cm) for distinct land use classes in Baltimore, USA (Data from Pouyat et al. 2007)

| | Cr, mg kg$^{-1}$ | Co, mg kg$^{-1}$ | Cu, mg kg$^{-1}$ | Ni, mg kg$^{-1}$ | Pb, mg kg$^{-1}$ | V, mg kg$^{-1}$ | Zn, mg kg$^{-1}$ | P, mg kg$^{-1}$ | K, mg kg$^{-1}$ | pH, mg kg$^{-1}$ |
|---|---|---|---|---|---|---|---|---|---|---|
| Transportation/ Commercial area | 50 | 11 | 37 | 20 | 226 | 35 | 154 | 497 | 1,177 | 6.8 |
| Industrial/urban open | 98 | 10 | 55 | 20 | 258 | 29 | 304 | 706 | 924 | 6.3 |
| Unmanaged forest | 53 | 14 | 51 | 20 | 113 | 36 | 103 | 373 | 561 | 5.7 |
| Park/Golf course | 91 | 19 | 39 | 34 | 109 | 35 | 93 | 486 | 1,038 | 6.4 |
| Residential area | 83 | 16 | 43 | 33 | 340 | 40 | 145 | 592 | 924 | 6.0 |
| Institutional area | 33 | 12 | 37 | 18 | 103 | 37 | 86 | 453 | 1,239 | 6.4 |

Sampled soils: 122
Metal extraction method: $HNO_3$ / HCl

were in the range 0.09–4.0 mg $kg^{-1}$, but there was no relationship between the concentration of PAH and the land use (Morilla et al. 2005).

In examples, taking only the topsoil into account, atmospheric deposition might be decisive (see Sections 3.2.2 and 3.3.1). If the subsoil is involved, a dependence of soil contamination on land-use type may play a much more important role. Areas consisting predominantly of man-made soils and additionally influenced by anthropogenically caused immission may exhibit the highest pollutant values in urban land (see Section 3.5.3).

By comparison, research studies in Ibadan (Nigeria) with a population of 5,200,000 covering topsoil analyses of different metropolitan uses resulted in minimal or no indication of anthropogenic metal inputs of 25 for peri-urban untouched soils, while 40 industrial sites, mechanic workshops/motor parks indicated higher concentrations. In 45 sampled roadside and farmland soils and 60 waste-dump site soils the values were higher. For assessment purposes, the results showed average total concentrations of approximately 85 mg Cu $kg^{-1}$, 22 mg Co $kg^{-1}$, 90 mg Cr $kg^{-1}$, 32 mg Ni $kg^{-1}$, 225 mg Pb $kg^{-1}$, 95 mg V $kg^{-1}$, and 526 mg Zn $kg^{-1}$, respectively. This city was altered by intensive urbanization due to rapid population increase. Rural-to-urban migration had resulted in varied human activities and uncoordinated land use planning within the urban catchment. Apart from chromium, this exhibited a concentration similar to the local background concentrations within the granitic catchment area present, all other analysed metals revealed twofold to sixfold enrichment, suggesting anthropogenic heavy metal inputs. The project highlighted impressively the impact on the environment with regard to urbanization and associated anthropogenic activities as well as the lack of adequate infrastructural and land use plans in most urban centres of less developed countries (Bigalke et al. 2005).

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# Chapter 3
# Causes of Soil Contamination in the Urban Environment

**Abstract** This chapter gives information about all kinds of contaminant sources of soils in urban environments. The description is mainly focused on heavy metals but also a number of organic pollutants, e.g. petroleum hydrocarbons, PAH and PCB, are of concern. Urban-to-rural gradients, depth gradients and relations to urban and industrial land uses are discussed in association with dust deposition. Different typical urban land uses such as parks and gardens in a number of examples worldwide are introduced. Special attention is paid to linear soil contamination with metals, petroleum hydrocarbons and de-icing salts along roadsides. Furthermore, soil contamination of floodplains in longitudinal and vertical direction located in urbanized areas is presented. Due to fertilizing with problematical mineral compounds and application of sewage sludge, wastewater and pesticides, particularly soils used for horticultural purposes in the urban environment potentially receive contaminants which are also integrated in the presentation of possible soil contamination sources. However, the chapter deals predominantly with soil contamination in relation to urban influences. Accordingly, reasons and extent of contamination of derelict land (industrial plots used in former times), anthropogenic deposits such as heaps and dumps as well as accident sites are taken into account. Many examples in different countries are given in order to illustrate the situation in detail. Finally, the chapter will provide a helpful tool to identify and separate the distinct contamination patterns described.

**Keywords** Anthropogenic deposit • Derelict land • Dust deposition • Floodplain • Roadside soil • Sewage sludge

## 3.1 Overview

Figure 3.1 presents the different contaminant sources influencing soil quality. Non-site related sources can be separated from site related ones. The first group involves extensive and linear contamination patterns. Extensive contamination sources can be caused by the geologically determined concentration and dust deposition.

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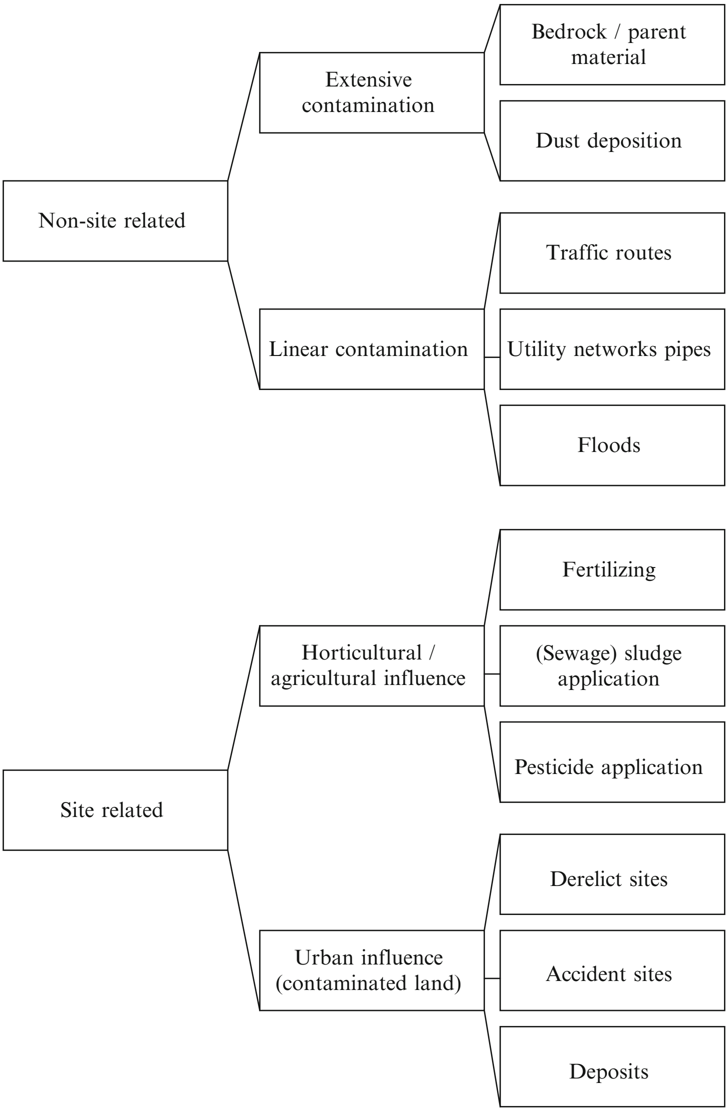



**Fig. 3.1** Diagram of contaminant sources in the urban environment

The linear sources include immission along traffic routes and utility networks pipes as well as flood occurrences in alluvial floodplains. Each contamination source is not strictly limited to defined local areas; it covers larger areas independent of the boundaries of the owner's property and land-use types. The second group is related to usually clearly limited sites because the contaminants have been used over an area (horticultural and agricultural influence) or it is generally site-specific such as contaminated derelict sites, heaps, etc.

## 3.2 Extensive Contamination

### *3.2.1 Bedrock and Parent Material Concentration*

One important source of trace elements in soils is the parent material from which the soil is derived. In spite of the fact that urban soils have undergone many anthropogenic changes such as cut and fill, bedrock and parent material concentration play an important role with reference to the contamination level, because this material can be excavated, transported and mixed on site and consequently it can be present within the whole soil profile afterwards.

Apart from the considered, neighbouring areas can be influenced by weathered material, since it can be transported by wind and water activity. Compared with urban areas, however, it is assumed that the levels of parent material are much lower than values expected in urban areas.

Metals and metalloids and other trace elements vary greatly in natural abundance in different geological materials (Table 3.1). For example, basalt-derived soils in low rainfall climates that have not undergone much leaching are invariably high in several metals such as Co, Cr, Cu, Ni, and V compared to other rock types such as granites or shale, but these metals have often been translocated to the subsoil (Reimann and de Caritat 1998).

Large areas of the world are covered in geologically recent (Holocene) alluvial or windblown sediments or glacial morainic materials. Loess and morainic deposits are widespread in northern North America, Northern Europe and Northern Asia. These consist of transported soils which often have a mixed geological origin and have undergone relatively little weathering as they tend to be of post-glacial age and hence are younger than 12,000–16,000 years. All the great mountain belts and hilly areas of the world are affected by continuing geological erosion exposing freshly weathered soil material on slopes and colluvial mantles.

Elsewhere, large areas of the world, particularly in the continents of Africa, South America and Australia, consist of old Tertiary Age land surfaces covered by a layer of unconsolidated, non-cemented, weathered material, including rock fragments, mineral grains and all other superficial deposits, that rests on unaltered, solid bedrock. Its lower limit is the weathering front (Meuser and van de Graaff 2010).

Weathering, leaching, pH and internal (within soil profile) translocation affect also distribution and mobility of the chemical elements that are recognised as contaminants, regardless whether they were native or have been added by mankind. In the great majority of cases where unusually high concentrations of heavy metals have been found, their solubility and mobility are extremely low so that ecological impacts are negligible. In some cases, however, such as the high Ni in soils formed on serpentine rock (Table 3.1), the vegetation has evolved to cope with the higher availability of Ni (Meuser and van de Graaff 2010).

Reimann and de Caritat (1998) warn that locally elevated concentrations of a specified element in a rock type will have an influence on all media that interact with that rock, such as soils, groundwater, vegetation and wind blown dust.

**Table 3.1** Average total metal concentration (mg kg$^{-1}$) of different parent materials (Data collected from different sources (Krauskopf 1967; Rose et al. 1979; Wedepohl 1984))

| | As | Cd | Co | Cr | Cu | Hg | Ni | Pb | Sb | Se | V | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Serpentinite | 1 | 0.1 | 110 | 2,980 | 42 | <0.01 | 2,000 | 14 | 0.1 | 0.1 | 40 | 58 |
| Basalt | 1.5 | 0.1 | 35 | 200 | 90 | 0.01 | 150 | 3 | 0.2 | 0.1 | 250 | 100 |
| Granite | 1.5 | 0.1 | 1 | 4 | 13 | 0.08 | 1 | 24 | 0.2 | 0.1 | 72 | 52 |
| Limestone | 1 | <0.1 | 0.1 | 11 | 6 | 0.16 | 7 | 6 | 0.3 | <0.1 | 45 | 20 |
| Sandstone | 1 | <0.1 | 0.3 | 35 | 30 | 0.29 | 9 | 10 | <0.1 | <0.1 | 20 | 30 |
| Shale | 13 | 0.2 | 19 | 90 | 39 | 0.18 | 68 | 23 | 1.5 | 0.5 | 130 | 120 |
| Loess | 7 | 0.2 | 9 | 30 | 15 | 0.02 | 18 | 45 | – | – | 64 | 25 |
| Sand | 1 | 0.1 | 1 | 2 | 2 | <0.01 | 5 | 10 | – | – | 3 | 11 |
| Glacial till | 8 | 0.3 | 7 | 35 | 15 | 0.04 | 18 | 20 | – | – | 29 | 40 |

For some trace elements similar results can be expected, taking the background values of topsoils related to very large areas like countries into consideration. Background values consist of both contamination sources the geological percentage and the ubiquitous atmospheric deposition. Systematically investigated topsoil concentrations of several elements from England and Wales showed geometric means of 38 mg Cr $kg^{-1}$, 18 mg Cu $kg^{-1}$, and 21 mg Ni $kg^{-1}$ confirming the low background values. On the contrary, there appeared to be a higher concentration of other elements than most of the parent materials listed in Table 3.1. The background values in England and Wales were 0.9 mg $kg^{-1}$ for Cd, 48 mg $kg^{-1}$ for Pb, and 85 mg $kg^{-1}$ for Zn (Thornton 1991) (see Section 3.2.2).

The majority of cases of gross soil contamination with organic pollutants are the result of industrial land use such as petrol stations, fuel and oil depots, etc., where tanks were leaking or spills occurred. However, river deltas are the natural environment where buried organic materials are often transformed into petroleum hydrocarbons so that low concentrations of these compounds and their derivatives can occur naturally in the soil. Another example mentioned refers to detectable subsoil concentration of Polycyclic Aromatic Hydrocarbons ($PAH_{EPA}$) ranging from 68 to 122 µg $kg^{-1}$ in a depth of 4 m in Polish peatland that were likely synthesized biologically (Malawska et al. 2006). However, as far as organic contaminants are concerned, the naturally occurring values do not tend to be much higher than the detection limit.

### 3.2.2 Dust Deposition

Extensive dust deposition is mainly caused by industrial emission. It is deposited in dry conditions and as suspended particulate matter. The size of particulate matter varies. The fraction of less than 10 µm designated as respirable particulate matter is most dangerous for human health. A portion of cement dust and fly ash may exceed that value, but there are some kinds of particulate matter with a general size distribution of less than 10 µm like asbestos dust and smoke derived from oil-fired power stations. It should be borne in mind that the smaller the suspended matter is the higher the contaminant concentration is due to the enhanced sorption capacity.

Contaminated particulate matter can be transported from outdoors into rooms. There, a dust accumulation occurs, particularly if windows remain opened during daytime. Of interest is the comparison between garden soil and house dust concentration. In England it has been found that the Cd, Cu, Pb and Zn values were larger in house dust than in the associated gardens (Thornton 1991).

Dust formation in urban areas may play an important role, in particular in arid and semi-arid regions, where dry conditions dominates, but also in humid climates dust is of importance for soil formation. For instance, in the city of Hanover (Germany) with a population of 520,000, an enormous accumulation of 5–8 cm within 50 years was observed (Burghardt and Höke 2005). When dust development and deposition occurred without any filter technique systems, the deposited layer


**Fig. 3.2** Dust deposition of approx. 50 cm from a coal processing factory onto an adjacent slope in Halle, Germany

can reach enormous thickness as observed in vicinity of a coal processing factory in Halle (Germany) with its 230,000 inhabitants (Fig. 3.2).

One important feature of urban areas is the often exposed land surface you can never discover in woodland and pasture and over long periods percentages of bare soils are private gardens and allotments in wintertime, playing fields, cemeteries, demolition and building sites, derelict and disused, mostly industrial land, waste heaps, railway embankments, and storage sites, where permanent dust deposition occurs (Thornton 1991). In urban and industrial areas sealed sites are influenced as well, since dust may easily penetrate into gaps between pavement stones, cobbles as well as railway embankments, constantly filling them up. It is supposed that dust will be laterally transported on the pavement stones and ultimately concentrated in gaps downslope (Burghardt and Höke 2005).

It is logical to expect that urban soils show higher contamination levels than the rural areas because of their proximity to a number of potential pollution sources. Big cities like New York (USA) with 23,200,000 inhabitants are affected by several contamination sources, for instance simultaneous industrial emission, impact of traffic, deposits of technogenic substrates, etc. Consequently, a decline in e.g. Cu, Ni and Pb concentrations was found with increasing distance from the city centre (Manhattan) into the rural district outside of the city. While in Manhattan Pb topsoil values of more than 130 mg kg$^{-1}$ were measured, at a distance of 50–60 km the values decreased to about 40 mg kg$^{-1}$, and at a distance of 120–130 km to about 30 mg kg$^{-1}$ (Pierzynski et al. 2005).

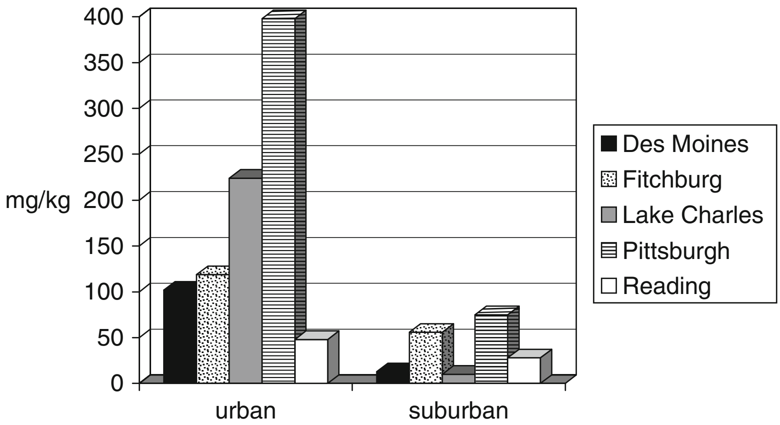



**Fig. 3.3** Lead concentration (mg kg$^{-1}$) in lawns (*upper horizon*) of suburban and urban areas in five US towns (Data from Carey et al. 1979, cited in Craul 1992)

The urban-to-rural gradient has frequently been found in developed countries of the northern hemisphere. Figure 3.3 presents the lead concentrations of five relatively small towns in the United States. The contamination level of the urban lawn is comparably high, as would be expected by the vicinity and exposure to high levels of air pollution in urban environments. The air pollution is related to metallic aerosols from heavy industry as well as combustion of fossil fuel. The investigations referred to lawns close to houses and in parks. The high level has not to be restricted to the upper horizons and forest floors. The activity of earthworms and ants (bioturbation) may play a role in the long-term mixing of the humic topsoil and the mineral subsoil, causing translocation of contaminants like Pb (Craul 1992) (see Section 6.3).

A city – suburb gradient has been confirmed by the soil investigations of the upper 5 cm in Marrakech (Morocco) with 1,200,000 inhabitants (El Khalil et al. 2008). They collected material from nine sites according to a gradient from suburban (No. 1) to urban zones (No. 9) (Fig. 3.4a–c). It is obvious that the Cd, Cu, Ni and Zn values tend to increase the shorter the distance to the city centre is. However, other factors as well as the expected dust deposition close to the city influence the situation. With increasing distance to the historic city centre the anthropogenic disturbance of the soil profiles showed distinct fingerprints as well. The technogenic fraction in the upper soil layer reaches 14% at site No. 9, indicating the huge disturbance. The coarse technogenic fraction revealed similar values at a distance of approximately 500 m from historic centre. Behind this distance the percentage ranged between 1% and 2%. Because of their relatively high contamination level the findings may contribute to the soil pollution significantly (see Section 4.3). Both the factors dust deposition and the presence of technogenic substrates overlaps each other with reference to the topsoil contamination.

In general, dust concentration in industrial areas tends to be much higher than in residential and rural areas. In particular, in regions with factories that have a relatively

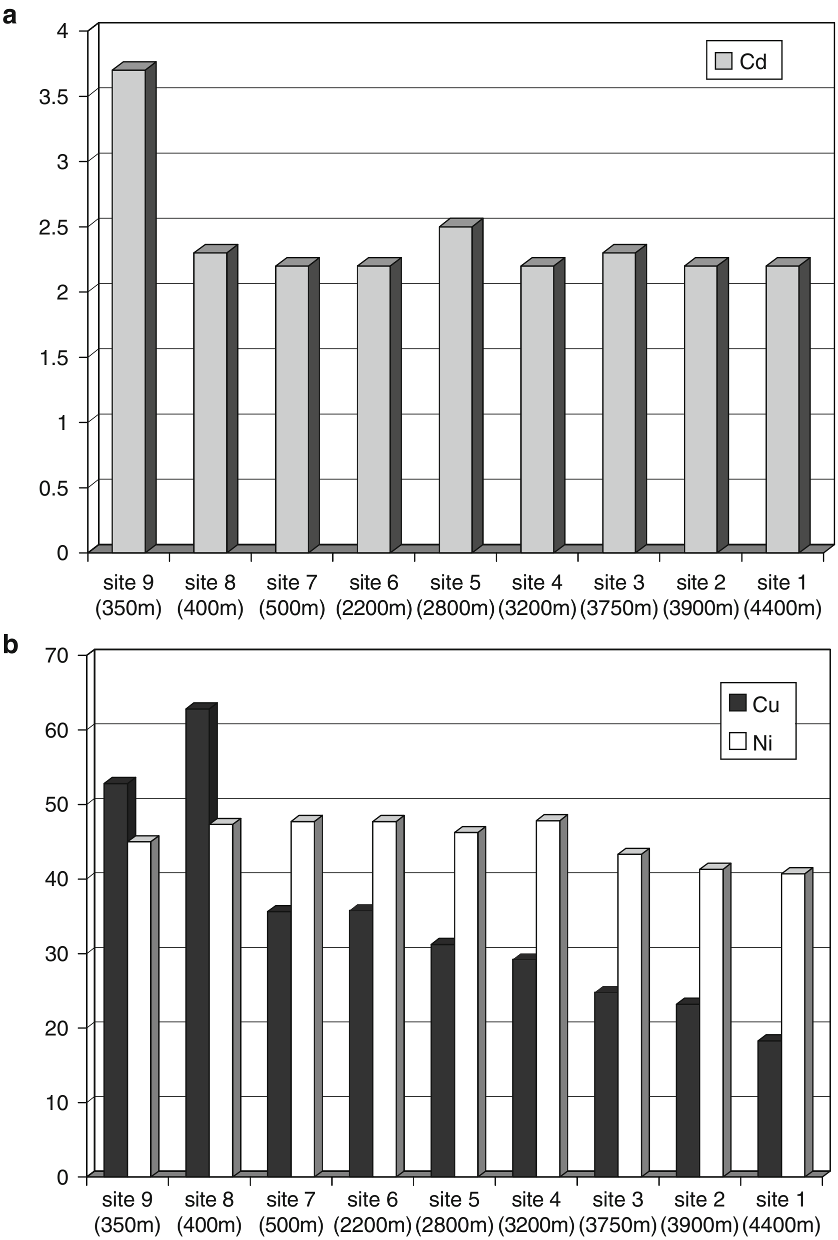



**Fig. 3.4** (**a**) Total Cd concentration (mg $kg^{-1}$), (**b**) total Cu and Ni concentration (mg $kg^{-1}$)

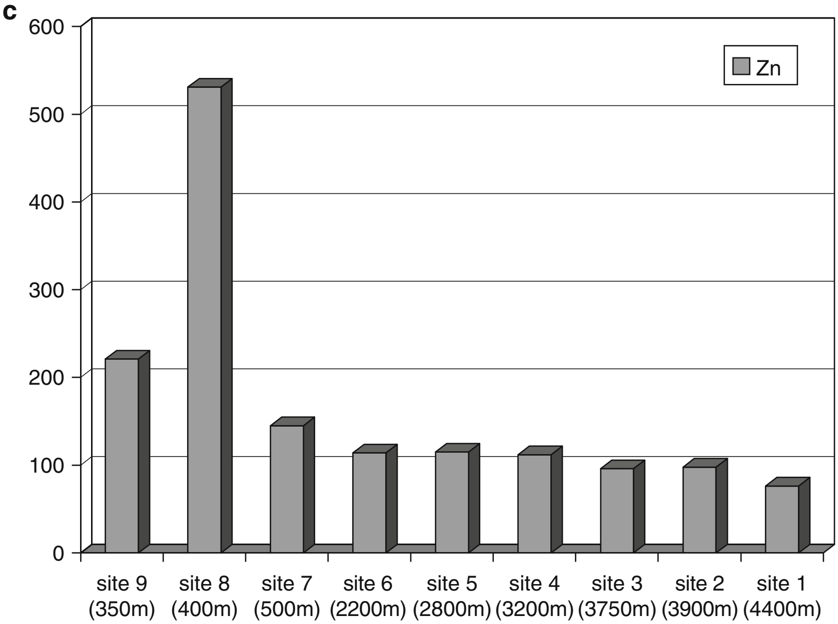



**Fig. 3.4** (continued) (**c**) total Zn concentration (mg kg$^{-1}$) in the upper part of urban soils in Marrakech, Morocco, depending on the distance to the historic city centre (Data from El Khalil et al. 2008. Sampled depth: 0–5 cm. Metal extraction method: aqua regia)

low number of air pollution filter systems the differences between the areas are enormous. For instance, the emission of suspended particulate matter ranged between 360 and 500 µg m$^{-3}$ in industrial catchments of several Indian cities, whilst in residential and rural areas the values varied from 140 to 200 µg m$^{-3}$ only. In relation to the respirable particulate matter with a diameter less than 10 µm, the results were 120–150 µg m$^{-3}$ for industrial areas and 60–100 µg m$^{-3}$ for residential and rural areas (CPCB 2004).

If some industrial complexes with very high emissions such as heavy metal works are present, the soil contamination is going to reach extremely high values. As seen in Fig. 3.5a and b, the non-ferrous metallurgy plot in Pirdop (Bulgaria) with a population of 8,000 influenced the soil properties not only in the immediate proximity of the industrial plot. Apart from lead, the elements Cu and Zn revealed the same tendencies of decreasing values with increasing soil depth and with increasing distance from the source. Moreover, it is obvious that there is a dependence on the wind direction, since the concentrations may be enhanced predominantly in the main wind direction. Very high concentrations of heavy metals accumulated especially in the upper portion of the humic topsoils (Penin and Tschernev 1997).

The tendencies described associated with the heavy metal gradients are basically applicable to organic pollutants as well. Near the town of Strazske (Slovakia)

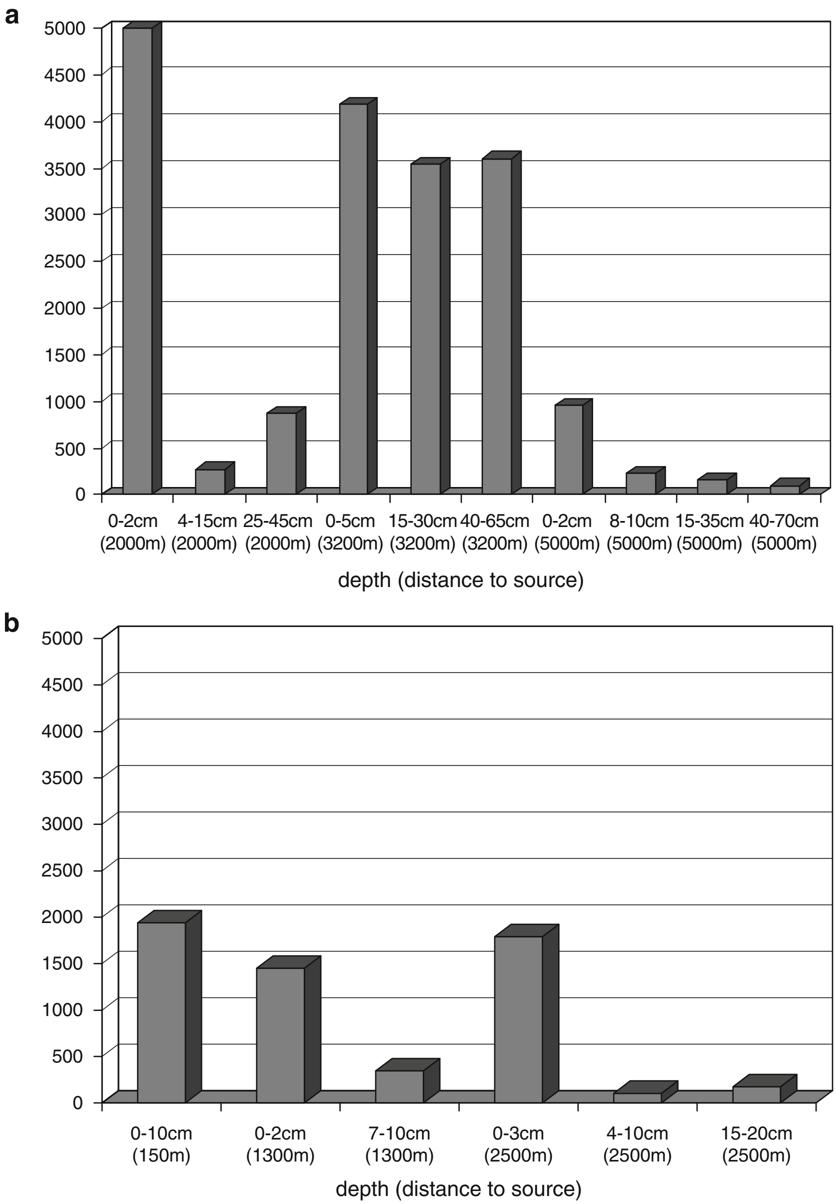



**Fig. 3.5** (**a**) Dispersion of lead (mg $kg^{-1}$) in relation to sampled soil depth (cm) and distance to emitting industry (m) in Zlatitza-Pirdop valley, Bulgaria (concentration in main wind direction). (**b**) Dispersion of lead (mg $kg^{-1}$) in relation to sampled soil depth (cm) and distance to emitting industry (m) in Zlatitza-Pirdop valley, Bulgaria (concentration in weak wind direction) (Data from Penin and Tschernev 1997)

with 5,000 inhabitants which is dominated by a chemical factory that produced technical Polychlorinated Biphenyls (PCB) mixtures between 1959 and 1984, samples were taken on an adjacent mountain slope influenced by the intensive dust emission and nearby deposition. As shown in Table 3.2, the accumulation of PCB (and Polycyclic Aromatic Hydrocarbons) in the organic layer exceeded the results from the mineral subsoil (depth gradient) and higher values tended to be found at the lower position of the neighbouring slope in direct contact with the emission source than at the central and upper position. In contrast to the heavy metals, the distance gradient seemed obviously to be less significant (Wilcke et al. 2003).

Industrial and mining processes influence increasingly the inventory of contaminants in the upper parts of the soils in the course of time. Trace elements usually in the earth's crust in very small quantities reach high concentrations that would never have been found in the absence of industrial development. For instance, in Wyoming and Idaho, USA, dust-inducing surface mining of coal and phosphorus led to an increase of selenium in the environment due to the exposure of Se containing overburden (Pierzynski et al. 2005).

If soils of an urban environment are compared with heavy metal mining soils, an appreciable difference can be noticed. Thornton (1991) analysed topsoils of special mining villages and the British capital London, with a population of 13,200,000, with reference to all locations investigated in England and Wales (Table 3.3). In general, it was stated that the results from London are much higher than from all locations investigated as everybody would expect, but surprisingly, the concentrations of the mining villages exceeded the London results considerably, apart from cadmium. In other words, the urban impact apparently seems to be less significant than the mining impact.

With reference to urban environments, in the two Norwegian cities Bergen, with 230,000 inhabitants, and Trondheim, with 160,000 inhabitants, sensitive uses like gardens, parks, kindergartens and playgrounds were checked in order to ascertain

**Table 3.2** Ranges of PAH and PCB concentrations in soil horizons of lower, central and upper slope position in the vicinity of a chemical factory dealing with technical PCB mixtures in Strazske, Slovakia (Data from Wilcke et al. 2003)

| Slope position | Horizon | Σ PAH[a] (mg $kg^{-1}$) | Σ PCB[b] (μg $kg^{-1}$) |
|---|---|---|---|
| Low (265–280 m asl, near factory, n = 3) | Organic layers | 1.0–6.8 | 6.6–95.5 |
| | Humic topsoil | 0.3–0.8 | 1.6–2.1 |
| | Subsoil | 0.05 | 0.1 |
| Central (395–405 m asl, n=3) | Organic layers | 1.3–2.5 | 7.2–16.0 |
| | Humic topsoil | 0.4–0.5 | 2.1–4.2 |
| | Subsoil | <0.02 | 0.1–0.3 |
| Upper (535–540 m asl, n = 3) | Organic layers | 1.7–6.2 | 6.7–23.1 |
| | Humic topsoil | 1.3–1.4 | 4.7–7.7 |
| | Subsoil | nd | nd |

nd = not determined
[a]Sum of 21 congeners
[b]Sum of 14 congeners

**Table 3.3** Cd, Pb and Zn concentrations (geometric means, mg $kg^{-1}$) of different land-use types in London boroughs and some mining villages compared to all study locations of England and Wales (Data from Thornton 1991)

| | London | Mining village (Shipham-Cd, Zn, Derbyshire-Pb) | All studied locations (less hotspots) |
|---|---|---|---|
| Cadmium | | | |
| Garden soil (depth 0–5 cm) | 1.3 (n = 579) | 100 (n = 522) | 1.2 (n = 4,128) |
| Vegetable plot soil (depth 0–15 cm) | 1.4 (n = 29) | 89 (n = 228) | 1.2 (n = 193) |
| Public garden soil (depth 0–5 cm) | 1.0 (n = 35) | no data | 1.2 (n = 221) |
| Lead | | | |
| Garden soil (depth 0–5 cm) | 654 (n = 578) | 5,610 (n = 89) | 266 (n = 4,126) |
| Vegetable plot soil (depth 0–15 cm) | 571 (n = 29) | 8,730 (n = 5) | 270 (n = 193) |
| Public garden soil (depth 0–5 cm) | 294 (n = 35) | 3,030 (n = 5) | 185 (n = 221) |
| Zinc | | | |
| Garden soil (depth 0–5 cm) | 424 (n = 579) | 9,340 (n = 519) | 278 (n = 4,127) |
| Vegetable plot soil (depth 0–15 cm) | 522 (n = 29) | 8,750 (n = 215) | 321 (n = 193) |
| Public garden soil (depth 0–5 cm) | 183 (n = 35) | no data | 180 (n = 221) |

Cd and Zn in Shipham: arithmetic mean
Metal extraction method: aqua regia

the influence of dust deposition. Consequently, the upper 2 cm of the 661 (Bergen) and 631 (Trondheim) soil samples were taken into consideration. Bergen's most important economic sectors are trade, shipping, the maritime industry and the public service. Trondheim has got a large number of companies producing and processing agricultural goods. For both cities, the main pollution sources discovered were in building maintenance (especially painting), industrial waste treatment, energy production, car traffic and the use of Cu-Cr-As impregnated wood. Much attention was paid to the latter source, because sand-pits in playgrounds and kindergartens were impregnated by copper, chromium and arsenic. Table 3.4 provides information about the soil concentrations of the two Norwegian municipalities. In detail, some areas were considered in order to confirm close correlations between some emitting factors and topsoil pollution nearby. The metals As, Cd, Cu, Pb, Se and Zn were more highly concentrated in the harbour area, where shipment of copper ore previously took place. Mercury was enhanced in proximity to a crematorium and a hospital incinerator. Near main roads there were accumulations of Hg, Pb and Zn (see Section 3.3.1). In general, some elements like Cd, Hg, Pb and Zn revealed increasing concentrations in the older parts of the city (Ottesen et al. 2000a, 2000b).

Similar results relating to a depth of 0–3 cm have been found in the capital of Norway, Oslo, with 570,000 inhabitants. A lot of industry is located in the city,

**Table 3.4** Content of different pollutants (mg kg$^{-1}$, µg kg$^{-1}$ for PCB, ng kg$^{-1}$ TE for PCDD/F) in the uppermost layer (0–2 cm) in soils of Bergen and Trondheim, Norway (Data from Ottesen et al. 2000a, b)

| | Bergen | | | Trondheim | | |
|---|---|---|---|---|---|---|
| Parameter | Median | Minimum | Maximum | Median | Minimum | Maximum |
| As | **3** | 0.5 | 38 | **3** | 0.5 | 83 |
| Cd | **0.2** | <0.1 | 1.5 | **0.2** | <0.1 | 11.3 |
| Cr | **18** | 0.5 | 215 | **69** | 7.9 | 199 |
| Cu | **29** | 4 | 2,850 | **35** | 1.7 | 706 |
| Hg | **0.1** | <0.1 | 2.9 | **0.1** | <0.1 | 4.5 |
| Ni | **13** | 1 | 310 | **45** | 6 | 231 |
| Pb | **38** | 3 | 5,780 | **35** | 9 | 976 |
| Zn | **85** | 8 | 998 | **98** | 7 | 3,420 |
| $PAH_{EPA}$ | **2.1** | <0.1 | 81 | **nd** | nd | nd |
| Benzo(a)pyrene | **0.2** | <0.1 | 9.9 | **nd** | nd | nd |
| $PCB_{sum\ 7}$ | **10** | <1 | 1,900 | **nd** | nd | nd |
| PCDD/F | **3.9** | 0.4 | 11.7 | **1.5** | 0.5 | 17.8 |

nd = not detected
Sampled sites: 661 (Bergen) and 631 (Trondheim) (parks, gardens, kindergartens, playgrounds)
Metal extraction method: aqua regia

particularly in the North-Eastern part, and in the past mining of sulphides was widespread. The mean values of metals investigated in the other Norwegian cities were as follows: arsenic 4.5 mg kg$^{-1}$, cadmium 0.4 mg kg$^{-1}$, chromium 33 mg kg$^{-1}$, copper 32 mg kg$^{-1}$, mercury 0.1 mg kg$^{-1}$, lead 56 mg kg$^{-1}$, and zinc 160 mg kg$^{-1}$. Partly, they indicated slightly higher concentrations than in Bergen and Trondheim. Furthermore, some elements associated with industrial emissions were taken into consideration such as barium (mean value 131 mg kg$^{-1}$), beryllium (mean value 5.5 mg kg$^{-1}$), cobalt (mean value 10 mg kg$^{-1}$) and vanadium (mean value 51 mg kg$^{-1}$) (Tijhuis and Brattli 2000).

In vertical direction the effect of dust deposition was visible in anthropogenically unaltered soils of three parks in the capital of Bulgaria, Sofia, with 1,100,000 inhabitants with old oak tree plantations (Table 3.5). For cadmium, copper, lead and zinc the concentrations decreased with increasing soil depth, mostly gradually. In the uppermost horizons the values were the highest ones, indicating heavy metal accumulation associated with litter enrichment. Arsenic shown in the table, the humus and total nitrogen content revealed parallel tendencies. Hence, the urban influence can be detected, if soils are analysed that do not demonstrate anthropogenic disturbance like excavation and deposition of technogenic substrates (Doichinova 2000).

In urban environments, where industrial and traffic emissions occur simultaneously, it is hard to draw a dividing line. In Szczecin (Poland), with a population of 510,000, the influence of dust deposition on the soil surface of public parks was investigated (Table 3.6). In the first instance, the heavy metal concentrations of the upper horizons depended on the proximity to intensively used traffic routes. Soils

**Table 3.5** Chemical characteristics of natural-like soils in parks with old oak plantation in Sofia, Bulgaria (Data from Doichinova 2000)

| Depth (cm) | Humus (%) | Total nitrogen (%) | pH | Cd (mg kg$^{-1}$) | Cu | Pb | Zn |
|---|---|---|---|---|---|---|---|
| Boris garden | | | | | | | |
| 0–10 | 5.8 | 0.18 | 6.5 | 0.33 | 33.2 | 34.3 | 37.9 |
| 10–30 | 3.3 | 0.07 | 5.8 | 0.16 | 24.2 | 20.1 | 30.1 |
| 30–53 | 2.4 | 0.06 | 5.7 | 0.16 | 30.9 | 19.4 | 47.9 |
| 53–73 | 1.6 | 0.05 | 6.2 | 0.03 | 14.5 | 7.8 | 18.6 |
| Southern park | | | | | | | |
| 0–6 | 6.4 | 0.14 | 5.3 | 0.11 | 32.6 | 31.7 | 39.9 |
| 6–34 | 3.1 | 0.10 | 4.3 | 0.06 | 27.5 | 26.7 | 35.5 |
| 34–60 | 1.6 | 0.05 | 5.4 | 0.06 | 27.7 | 12.9 | 27.7 |
| 60–80 | 1.1 | 0.04 | 5.2 | 0.05 | 23.9 | 10.5 | 20.2 |
| Western park | | | | | | | |
| 0–10 | 5.1 | 0.16 | 5.9 | 0.14 | 65.7 | 30.9 | 81.0 |
| 10–30 | 2.5 | 0.13 | 4.6 | 0.11 | 71.8 | 27.8 | 85.8 |
| 30–60 | 1.8 | 0.12 | 5.7 | 0.21 | 64.3 | 22.7 | 61.9 |
| 60–73 | 0.9 | 0.10 | 5.9 | 0.11 | 59.5 | 22.0 | 51.9 |
| 73–83 | 0.3 | 0.04 | 6.2 | na | na | na | na |

na = not analysed
Metal extraction method: aqua regia (total amounts)

**Table 3.6** Heavy metal concentrations (mg kg$^{-1}$) in soils of urban parks in Szczecin, Poland (Data from Niedzwiecki et al. 2000)

| Site | Distance from main roads (m) | Depth (cm) | Cd (mg kg$^{-1}$) | Cu | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|
| Slowackiego street | 2–5 | 0–5 | 1.8 | 28.5 | 8.4 | 140 | 398 |
| | | 5–20 | 1.8 | 20.4 | 8.3 | 94 | 280 |
| Dendrological garden | 5–50 | 0–5 | 0.7 | 20.6 | 10.9 | 48 | 73 |
| | | 5–20 | 0.6 | 18.7 | 11.4 | 50 | 71 |
| Zeromskiego park | 5–50 | 0–5 | 0.6 | 23.6 | 9.0 | 61 | 85 |
| | | 5–20 | 0.6 | 20.9 | 8.8 | 54 | 60 |
| J. Kasprowicz park | 5–50 | 0–5 | 2.5 | 32.4 | 13.6 | 111 | 263 |
| | | 5–20 | 2.0 | 17.0 | 9.4 | 60 | 169 |
| J. Kasprowicz park | 150–200 | 0–5 | 0.5 | 13.8 | 8.2 | 48 | 63 |
| | | 5–20 | 0.4 | 11.0 | 4.2 | 45 | 44 |

Metal extraction method: concentrated $HNO_3 + HClO_4$

**Table 3.7** Heavy metal content (mg $kg^{-1}$) in dust fallout of different sampling sites in Szczecin, Poland in 1998 (Data from Niedzwiecki et al. 2000)

| Sampling site | Cd | Cu | Ni | Pb | Zn |
|---|---|---|---|---|---|
| Gas station area | 4.0 | 161 | 30 | 272 | 1,135 |
| Machinery for municipal waste treatment | 7.2 | 214 | 38 | 1,029 | 3,389 |
| Metal and repair shop in harbour area | 31.0 | 864 | 352 | 763 | 3,709 |
| Warehouse near grain silo in harbour area | 13.6 | 60 | 23 | 179 | 389 |
| Residential area | 3.1 | 139 | 31 | 952 | 2,970 |
| Playground | 1.0 | 50 | 19 | 214 | 486 |

41 samples of dust fallout in 1998
Metal extraction method: concentrated $HNO_3$ + $HClO_4$

sampled relatively close to the streets with heavy traffic exhibited higher concentrations than soils with more distance to the roads (see Section 3.3.1). On the other hand, the influence of vehicle traffic can obviously be overlapped by other industrial emissions, since the results of three parks with sampling distances between 5 and 50 m differ considerably. Collected fallout in the same city showed impressive differences (Table 3.7). There was a tendency for industry branches normally responsible for metal emissions such as metal and repair shops and treatment facilities for municipal waste to indicate higher metal concentrations in dust fallout compared to less problematical sites like residential areas and playgrounds located in parks with a minimal distance to roads of 20–25 m. Therefore, it can be assumed that local industrial emissions may alter the heavy metal contents abruptly. The typical gradient of decreasing concentration with increasing distance from the road can be decisively disturbed due to local dust deposition of some industrial complexes (Niedzwiecki et al. 2000).

It was also possible to discover a spatial differentiation in the capital of Korea, Seoul, with 9,700,000 inhabitants, although both approx. 50,000 factories and approx. 1.6 Million vehicles are present. In spite of elevated copper level in the whole city (average concentration in 0–10 cm depth: 84 mg $kg^{-1}$) based on a grid with 64 sampling points (Table 3.8), a copper accumulation was analyzed in the south-western part of the city called Kuro, where industrialization was very high and many brass and bronze alloy factories are present. For production purposes copper is principally used. Accordingly, in Kuro the copper concentration in dust was enhanced as well (Chon et al. 1995). Obviously, the exact location of dust fallout sampling may play a significant role. The dust concentrations in Seoul based on a static grid pattern were much lower than results from the Polish city of Szczecin, where the sampling points were closely associated with the polluters.

Agglomeration areas tend to have an enhanced topsoil contamination, since several air-borne factors may impact soil quality simultaneously. Multivariate geostatistical analysis of topsoils in Beijing (capital of China) with a population of 12,200,000, confirmed the agglomeration effect. There 773 samples were collected, considering the depth 0–20 cm only and covering different city uses all over Beijing. Anthropogenic factors generally caused metal concentration to increase, in particular in the city centre. It has been found that the main sources of pollution

**Table 3.8** Heavy metal concentration of soils (0–10 cm) and dust in Seoul, Korea (Data from Chon et al. 1995

| | Soil | | Dust | |
|---|---|---|---|---|
| | Arithmetic mean | Range | Arithmetic mean | Range |
| Cd | **3.1** | 1.0–4.4 | **3.0** | 1.9–3.2 |
| Cu | **84** | 11–471 | **101** | 35–294 |
| Pb | **240** | 93–1,636 | **245** | 118–965 |
| Zn | **271** | 55–596 | **296** | 98–958 |

64 sampled garden soils, 64 sampled roadside dust collectors
Metal extraction method: nitric-perchloric acid + HCl

**Table 3.9** Average concentration of metals (mg $kg^{-1}$) in three Chinese agglomeration areas (rural soils in periphery, city soils) (Data from Zheng et al. 2008)

| | Range (city area) | Mean value (city area) | Mean value (rural area) | Mean value (city area) | Mean value (rural area) | Mean value (city area) | Mean value (rural area) |
|---|---|---|---|---|---|---|---|
| Element | Beijing | | | Nanjing | | Hong Kong | |
| As | 0.1–25.6 | 8.4 | 7.8 | – | – | 10.7 | 16.5 |
| Cd | <0.01–0.97 | 0.15 | 0.12 | – | – | 0.74 | 0.94 |
| Cr | 7–228 | 35.6 | 29.8 | 84.7 | 41.9 | – | – |
| Cu | 2–282 | 23.7 | 18.7 | 66.1 | 25.4 | 9.1 | 16.1 |
| Ni | 2.8–169 | 27.8 | 26.8 | – | – | – | – |
| Pb | 5–117 | 28.6 | 22.6 | 107.3 | 17.5 | 89.9 | 40.6 |
| Zn | 22–400 | 65.6 | 57.5 | 162.6 | 75.2 | 58.8 | 51.0 |

773 sampled sites, depths 0–20 cm (Beijing)
Metal extraction method: $HNO_3 + H_2O_2$

result from vehicle exhaust and smelters. Generally, the differences between the background values of the rural Beijing area and the city are relatively low, if the mean values are taken into account (Table 3.9). However, the maximum values usually linked to specific pollutant sources indicate partly very high values. Furthermore, the values from Beijing seem to reveal a lower level in comparison with other results of Chinese cities. For example, the urban influence showed clearer tendencies in Nanjing with 5,300,000 inhabitants and Hong Kong with 7,300,000 inhabitants (Zheng et al. 2008).

The contamination level of topsoils in urban environments depends obviously on many parallel factors. Besides the question of air-borne dust deposition, concentration of parent material and the land-use type appear to determine the contamination level, additionally leading to a complicated interpretation of topsoil contamination. In this context three European cities have been investigated in detail: the city of Torino (Italy) with a population of 1,700,000 and with a long industrial history, mainly car manufacturing as well as metallurgical industries; Seville (Spain) with 700,000 inhabitants and with intensive industry dealing with fertilizer and building material production, pottery, ship-building and processing of agricultural goods; Ljubljana (capital of Slovenia) with a population of 260,000 and with some industrial plots and central heating stations, but particularly heavy traffic resulting from daily migration of employees from the suburbs into the city area. Parks, riverbanks, ornamental gardens and roadsides with a total number of more than 800 samples were considered. Results are presented in Table 3.10 (Biasioli et al. 2007).

It can be concluded that all three towns show a high level of Pb and Zn due to the car traffic as a predominant source of pollution (see Section 3.3.1). Furthermore, the copper level reveals enhanced values identifying the parameters Cu, Pb, and Zn as typical contaminant parameters in an urban and industrial landscape. On the contrary, chromium and nickel are of less significance. Apart form this general point of view both Cr and Ni indicate high concentrations in the Torino catchments, especially in the alluvial soils of the region. But this result is well-known for the Torino area associated with lithogenic origin, the Cr and Ni containing upland serpentinites (see Section 3.2.1), which are very common in the alluvial deposits. Subsequently, both elements have been strongly correlated (r = 0.91). With reference to all three cities Cr and Ni demonstrated close correlations, whilst Cu, Pb and Zn created another cluster with high correlation coefficients.

Biasioli et al. 2007 found that the Cu, Pb and Zn values were influenced by the land-use type (see Section 2.4) with the general order riverbank < parks < roadsides < ornamental gardens. In particular, the young age of the riverbanks explained their relatively low level. The permanent anthropogenic disturbance of sites such as parks and ornamental gardens connected with material fill and mixing led to the enhanced concentration of these uses. Moreover, the mixing procedures are responsible for the missing depth gradient between 0–10 and 10–20 cm.

Industrial dust is a carrier of many hazardous substances, which accumulate in topsoils in the course of time. Every source connected with iron metallurgy, fossil fuel combustion, cement production and traffic emits a considerable amount of magnetic particles and connected trace elements. Accordingly, in urban areas an enhanced level of magnetic susceptibility compared with rural areas appears,

**Table 3.10** Heavy metal concentration (mg kg$^{-1}$) in parks, riverbanks, ornamental gardens and roadsides of three European cities (depth 0–20 cm) (Data from Biasiolo et al. 2007)

| | Cr | Cu | Ni | Pb | Zn |
|---|---|---|---|---|---|
| Ljubljana | | | | | |
| Mean | **34** | **39** | **26** | **86** | **145** |
| Min-max | 10–186 | 14–135 | 14–45 | 10–387 | 56–581 |
| Seville | | | | | |
| Mean | **34** | **53** | **29** | **118** | **101** |
| Min-max | 11–101 | 9–365 | 16–62 | 15–977 | 21–443 |
| Torino | | | | | |
| Mean | **172** | **89** | **187** | **156** | **173** |
| Min-max | 64–930 | 15–430 | 77–830 | 14–1,440 | 53–880 |

130 (Ljubljana), 154 (Seville), 122 (Torino) sampling sites (open spaces, riverbanks, ornamental gardens, roadsides)
Metal extraction method: aqua regia

because there is a strong correlation between magnetic susceptibility and heavy metal content. In the uppermost horizons of soils in areas of increased industrial dust and fly ash immissions as a result of fossil fuel combustion and mineralogical transformation of iron forms into ferrimagnetic iron oxides such as magnetite and maghaemite, provoking detectable magnetism in soils. The enhanced magnetism in topsoils of areas with heavy industry has been found in the surroundings of iron-works and power plants in Polish cities like Katowic with 310,000 inhabitants, Krakow with 800,000 inhabitants and Nova Huta with 200,000 inhabitants (Magiera and Strzyszcz 2000).

For the same reason, soils of about 30 parks were investigated in the capital of the Czech Republic, Prague, with a population of 1,200,000. Magnetic susceptibility was compared with concentrations of some elements like arsenic, cobalt, nickel, lead and zinc. Generally, high values of magnetic reaction coincided with areas indicating high metal concentration. However, it was not possible to determine a strong correlation, since the distribution of deposited dust was irregular due to strong local sources of pollution and to the distinct spectrum of metals analysed (Kapicka et al. 2000).

## 3.3 Linear Contamination

### *3.3.1 Traffic Routes*

#### 3.3.1.1 Roadland

In cities dust derived from traffic sources may play a significant role independent of the developing status of the town investigated, because in city centres heavy traffic can generally be supposed to be present. Consequently, heavy metal deposition onto soil surface occurs, increasing topsoil contamination. Arsenic observed in some city centres in Bangladesh, trace elements showed enhanced levels. For instance, in Dhaka with a population of 10,000,000, lead alongside roads ranged from 96 to 212 mg $kg^{-1}$ and zinc from 346 to 365 mg $kg^{-1}$ in topsoils (Sattar 2000).

There are a lot of contaminant sources relating to traffic influence. Car exhaust particularly from diesel fuel produces benzene and Polycyclic Aromatic Hydrocarbons (PAH). In former times gasoline lead was emitted. This source was strongly reduced after the introduction of unleaded gasoline. On the other hand, new possible contaminants based on currently used car technology such as platinum and rhodium arose. Both trace elements derive from catalytic filter systems that are common in manufactured cars. Along German motorways an increase in platinum and rhodium in topsoils was found. The values varied between 10 and 69 μg $kg^{-1}$ for Pt and 1–20 μg $kg^{-1}$ for Rh. The background values in the same catchments, however, indicated 5 μg Pt $kg^{-1}$ and 0.4 μg Rh $kg^{-1}$ only. Apart from car exhaust, other contaminant sources should be taken into account (Table 3.11).

**Table 3.11** Contaminant sources along roadsides

| Contamination source | Parameter |
|---|---|
| Car exhaust | $NO_x$, benzene, PAH |
| | CO, $SO_2$, Pb, Pt, Rh, phenols |
| Residues from tyre and brake wear | Heavy metals (Cr, Cu, Ni) |
| Products of corrosion | Heavy metals (Cd, Cu, Zn) |
| Oil drip lost | Hydrocarbons |
| | Halogenated hydrocarbons |
| Residues from weathered and overused road surface | PAH (asphalt), heavy metals (concrete) |
| Road management | De-icing salts |
| | Detergents |

PAH = polycyclic aromatic hydrocarbons

In European cities it was possible to find different impacts. Heavy metal concentrations were measured in the upper 0–20 cm soil layers in Sofia, Bulgaria, taking the gradient urban – suburban – rural into consideration. Locations close to the highways were mainly sampled. In urban areas copper varied from 25 to 70 mg $kg^{-1}$, in suburban areas from 14 to 25 mg $kg^{-1}$ and in the surrounding countryside from 23 to 53 mg $kg^{-1}$. Related to zinc the results were 31–99 mg $kg^{-1}$ (urban), 29–48 mg $kg^{-1}$ (suburban) and 36–131 mg $kg^{-1}$ (rural), to lead 24–32 mg $kg^{-1}$ (urban), 18–35 mg $kg^{-1}$ (suburban) and 21–58 mg $kg^{-1}$ (rural). The relatively high amounts in the periphery of Sofia were connected with the chosen sampling sites near roads (Doichinova and Zhyanski 2005).

In general, the concentration of heavy metals decreases with increasing distance to the traffic and with increasing soil depth. This context is illustrated for Cd, Pb and Zn in Table 3.12 (Craul 1992). It is estimated that about 10% of the emitted Pb is deposited within the first 100 m. The problem of lead contamination of roadside soils is ongoing in spite of the lead reduction in petrol, since Pb is relatively immobile and consequently it will remain in topsoils over many decades (Thornton 1991).

The impact of traffic routes on adjacent soils was also interpreted in relation to intensively used motorways. In Warsaw Province different soils were sampled along the highways Warsaw – Krakow and Warsaw – Gdansk, Poland, at several distances from the road. Cadmium did not show any clear tendencies but lead and zinc reacted visibly (Table 3.13). The concentrations decreased with increasing soil depth and, except for zinc between 10 and 20 m, increasing distance to the emitting road. At a distance of 50 m it was still possible to detect increased values. Adjacent allotment soils exhibited same tendencies. At a distance of 30 m the topsoil values ranged between 21 and 176 mg $kg^{-1}$ and at a distance of 100 m between 29 and 60 mg $kg^{-1}$ Pb. The results for Zn were 90–1,290 mg $kg^{-1}$ (30 m) and 88–392 mg $kg^{-1}$ (100 m). Consequently, it must be assumed that road emissions like car exhaust fumes of main traffic arteries may accelerate soil contamination in adjacent sites as far as a distance of up to 100 m (Maciejewska and Kwiatkowska 2000).

Two bypass motorways with three lanes each in Nantes (France) with 800,000 inhabitants were checked in relation to heavy metals and organic pollutants.

**Table 3.12** Ranges of Cd, Pb and Zn in soils (mg $kg^{-1}$) with varying distances to roads, sampled in different roadside locations in Baltimore, Washington D.C., Platte City and Cincinnati, USA (Data from Lagerwerff and Specht 1970, cited in Craul 1992)

| | Soil depth (cm) | | |
|---|---|---|---|
| Distance (m) | 0–5 | 5–10 | 10–15 |
| Cadmium | | | |
| About 2.4 | 0.90–1.82 | 0.44–0.76 | 0.28–0.54 |
| About 4.9 | 0.40–1.51 | 0.38–0.70 | 0.26–0.61 |
| About 9.6 | 0.22–1.02 | 0.18–0.51 | 0.12–0.05 |
| Lead | | | |
| About 2.4 | 150–522 | 20–460 | 11–416 |
| About 4.9 | 101–378 | 14–260 | 8–104 |
| About 9.6 | 55–164 | 10–108 | 6–69 |
| Zinc | | | |
| About 2.4 | 54–172 | 24–94 | 11–72 |
| About 4.9 | 60–110 | 16–48 | 10–42 |
| About 9.6 | 15–54 | 11–46 | 8–42 |

**Table 3.13** Lead and zinc average content (mg $kg^{-1}$) along highways in Warsaw Province related to road distance (Data from Treblinska, cited in Maciejewska and Kwiatkowska 2000)

| | Pb | | | Zn | | |
|---|---|---|---|---|---|---|
| | Distance | | | Distance | | |
| Highway from Warsaw to | 10 m | 20 m | 50 m | 10 m | 20 m | 50 m |
| Gdansk | 58 | 47 | 23 | 78 | 169 | 37 |
| Krakow | 78 | 62 | 57 | 135 | 247 | 117 |
| Katowice | 89 | 70 | 53 | 116 | 115 | 117 |

6 sampled surface soils each
Metal extraction method: $HNO_3 + H_2SO_4 + HCl$

The metal dispersion agreed with the results yielded in the research studies mentioned before. Higher Pb and Zn values, particularly at a distance of 0.5 m, were due to the use of leaded gasoline in the past and galvanized steel crash barriers. Total hydrocarbons and PAH were also involved. The first motorway of concern was exposed to a daily traffic volume of about 24,000 vehicles and consisted of porous asphalt surfaces. The second one measured a daily traffic volume of approximately 21,000 cars and consisted of conventional asphalt. The Total Petroleum Hydrocarbons (TPH) varied from 111 to 385 mg $kg^{-1}$ at a distance of 0.5 m and from 1.0 to 6.8 mg $kg^{-1}$ at a distance of 5 and 25 m from the first road, respectively. Accordingly, the second motorway revealed 262–908 mg $kg^{-1}$ at 0.5 m and < 0.1–7.9 mg $kg^{-1}$ at distances far away. In relation to different soil depths measured (0–2, 2–10 and 10–30 cm), the enhanced hydrocarbon level continued to a depth of 30 cm due to migration downwards. Although the road surfaces contained asphalt, the PAH results were negligible. PAH of the first motorway always remained below detection limits, the other motorway reacted to a small degree with respect to fluoranthene only. The French study concluded that the organic pollutant level decreased rapidly with increasing distance from the road (Legret and Pagotto 2000).

It is difficult to estimate the exact distance of deposited dust, since some characteristics may influence the contaminant dispersion, especially the speed of vehicles and the presence or absence of roadside green and barriers like sound-insulating walls. It can be expected that the dispersion increases with increasing average vehicle speed, because higher wind speed caused by fast-moving cars and particularly buses and trucks can transport particulate matter far away. Moreover, missing barriers accelerate dust dispersion.

Contaminated areas along roadsides are predominantly important in relation to environmental impacts. Hay harvesting for animal feed or berries harvesting for private consumption are typical problems associated with roadside soils which are heavily polluted (Craul 1992).

In the vicinity of roads the broad drainage of runoff and its infiltration into the road bank with a percolation through planted soils turned out to be the best common treatment. This is the standard treatment in most countries for road runoff even on motorways with very high traffic density. So, even in the countryside roadside soils serving as a filter medium are modified by permanent infiltration of road runoff. As a result of the retention processes in the soil contaminants are enriched in the upper layers of the soil and road embankment close to the roads. When removing this material from the road site an adequate means of disposal has to be found.

The material used for road construction is of interest with respect to the potential contamination level (see Section 6.1.1). In La Teste (France) with 20,000 inhabitants and Le Mans with 140,000 inhabitants roads containing municipal solid waste incinerator bottom ash as sub-base were examined. Both urban roads were intensively used, the first one by 30–40 trucks a day, the second one by 12,000 vehicles per day involving bus traffic. The road construction enabled good deflection for flexible structures and a high bearing capacity. It consisted of a bituminous top layer (thicknesses 4–17 cm), unbounded graded material (thicknesses 8–15 cm), waste incinerator ash (thicknesses 16–71 cm) and natural sandy soil below. Influenced by weathering processes a long term percolation of heavy metals was supposed (see Section 4.1.2) but the leaching potential and accordingly the contamination of the underlying sand were relatively low (Table 3.14). However, it should be noted, that copper, lead and zinc tended to percolate to the layer below the load-bearing layer containing ash in La Teste and some elements in the Le Mans test trial seemed to indicate chromium, copper and zinc percolation into the underlying layers, too (Francois 2000).

Another point for consideration related to road land is the application of de-icing salts. Commonly de-icing salts consist of sodium chloride or calcium chloride. They are able to be mixed with mineral particles like sand, gravel and crystalline bottom ash.

The general disadvantages of de-icing salt application are:

- Replacement of cations by sodium, disturbance of the exchangeable cation capacity
- Stimulation of heavy metals desorption potential, leading to raised mobile metal concentration
- Dispersion of soil colloids, leading to filled small pores on the surface in a line with reduced water infiltration
- Changes of the osmotic potential, leading to altered wilting points of plants

**Table 3.14** Heavy metal concentration (mg $kg^{-1}$) and pH values in substrates and soils underneath constructed roads in La Teste and Le Mans, France (Data from Francois 2000)

| | | | Cd | Cr | Cu | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|
| Depth (cm) | Substrate | pH | mg $kg^{-1}$ | | | | | |
| La Teste, trench 1 | | | | | | | | |
| 12–32 | Ash | 9.0 | 2.0 | 170 | 549 | 44 | 1,013 | 2,261 |
| 32–37 | Sand | 8.6 | 0.1 | 6 | 17 | 2 | 55 | 122 |
| 37–42 | Sand | 7.1 | 0.1 | 4 | 2 | 1 | 9 | 9 |
| 42–52 | Sand | 7.2 | <0.1 | 6 | 1 | 1 | 7 | 5 |
| La Teste, trench 2 | | | | | | | | |
| 35–85 | Ash | 8.9 | 4.3 | 215 | 1,238 | 67 | 1,341 | 4,120 |
| 85–90 | Sand | 8.9 | <0.1 | 3 | 2 | 1 | 13 | 21 |
| 90–95 | Sand | 8.9 | <0.1 | 3 | 2 | <1 | 7 | 10 |
| 95–105 | Sand | 9.0 | 0.1 | 4 | 6 | 2 | 29 | 48 |
| Le Mans, trench 1 | | | | | | | | |
| 70–90 | Ash | 8.2 | 40.3 | 362 | 1,585 | 106 | 1,359 | 3,671 |
| 90–100 | Silty sand | 7.5 | 0.5 | 121 | 17 | 27 | 33 | 82 |
| 100–120 | Silty sand | 7.4 | <0.1 | 105 | 15 | 22 | 46 | 59 |
| Le Mans, trench 2 | | | | | | | | |
| 35–60 | Ash | 8.3 | 15.9 | 403 | 940 | 85 | 881 | 2,881 |
| 60–70 | Silty sand | 7.9 | 0.3 | 43 | 16 | 8 | 32 | 52 |
| 70–80 | Silty sand | 7.5 | 0.1 | 38 | 10 | 9 | 26 | 50 |
| 80–100 | Silty sand | 6.4 | <0.1 | 95 | 9 | 30 | 26 | 87 |

Metal extraction method: Hydrofluoric and perchloric acid

- Damages to the plant tissue by salt spray
- Leachate of soluble chloride into groundwater or laterally into surface water bodies
- Corrosion of concrete, metals, asphalt

The Electrical Conductivity of roadside soils in six U.S. cities revealed average values between 0.57 and 4.04 dS $m^{-1}$, while the highest detectable value reached 8.83 dS $m^{-1}$ (Craul 1992).

Cities located in colder climate regions are commonly exposed to de-icing salt application. In the capital of Russia, Moscow, with 10,500,000 inhabitants, mixtures of sand and small amounts of salt were used until 1993. Because of littered highways and communication lines the sand percentage was reduced. Nowadays up to 5 kg $km^{-2}$ of pure salt (NaCl) are applied in wintertime together with a small amount of sand and pea gravel. Several boulevards and highways in the centre of Moscow were taken into consideration in order to estimate the impact of de-icing salts on adjacent soils and vegetation. Trees along the roads such as *Acer negundo, Tilia cordate* and *Populus balsamifera* increasingly suffered from osmotic and salt shock. Sodium was leached out from the upper layer slowly and continuously and chloride was transported downwards quickly. In autumn toxic concentrations of sodium and chloride were observed at a depth of 20–30 cm. Even at a depth of 100 cm it was possible to detect toxic concentration. Moderate to high salinization was found within a depth of 1 m, inhibiting growth and development of trees. Hence, the complete root zone was apparently affected. Moreover, sodium led to increased pH values in the root zone, indicating a slightly alkaline reaction and the exchangeable percentage of sodium (usually much lower than 1%) reached 4.2–13.9% in some profiles under investigation (Chernousenko et al. 2000).

Soils of lawns alongside roads in Lodz (Poland) with a population of 750,000 confirmed the enhanced salt concentrations in urbanized areas. A salt concentration between 230 and 9,550 mg $kg^{-1}$ was found and the percentage of NaCl ranged between 7 and 319 mg $kg^{-1}$. In podsolic forest soils of Lodz, however, the sum of salts lay below 130 and of NaCl below 13 mg $kg^{-1}$. The raised values were even detected to a depth of 150 cm, resulting from salt leaching. Both sodium chloride and sodium sulphate were always present in deeper sections of the profiles. The conditions of salinity in soils near streets contributed to the increased concentrations of chlorine and sodium in the foliage of analyzed street trees such as *Acer platanoides* and *Tilia euchlora* (Czarnowska et al. 2000).

Chemical conditions of different greenbelts along roads were considered in Saint-Petersburg (Russia) with a population of 4,500,000. They include greenbelt between road and sidewalk, complex greenbelts combining different plantations and tree plantations. A lot of trees indicated growth deterioration and visible damages such as chlorosis, necrosis, complete leaf yellowing and premature falling of leaves. The main reason might be the application of de-icing salt-sand mixtures in winter. Results between 41.7 and 128.0 mg $kg^{-1}$ chloride as well as 15.5 and 67.3 mg $kg^{-1}$ sodium were found. The highest values were obtained at a boulevard with intensive traffic, where the trees planted (*Tilia cordata*) indicated deteriorated growth. It was not possible for other chemical properties to cause damages to plants. In the upper 25 cm the soil revealed a slightly alkaline reaction, a base saturation of nearly 100% and humus contents between 3.4% and 8.3%. The total nitrogen content ranged from 0.16% to 0.41%. In most of the soils analyzed the mobile nutrient supply was assessed to be middle to high (P), low to middle (K) and low (N). In conclusion, the restricted plant growth in a line with visible features did not accord with the nutrient parameters (Yurieva 2000).

#### 3.3.1.2 Railway Embankment

Different materials are used for the construction of railway embankments, so that the contamination level may depend on the kind of construction material used. Below tracks and sleepers, which are often conditioned by tar painting (see Fig. 3.15d), the embankment is built of natural stones like basalt or technogenic substrates like metal works slag. The subsoil below contains gravel and sand, occasionally concrete and styropene, and is underlaid by plastic sheets. In order to stabilize it some binding agents are usually applied. Under the construction the piled soil forming a dam is mechanically compacted and chemically stabilized.

A railway embankment at a ruined railway station that had remained unused for 40 years in the capital of Germany, Berlin with its 3,400,000 inhabitants was studied in detail. The parent material of the banks consisted mainly of basalt and occasionally construction debris was found. Between the coarse fraction fine material derived from the accumulation of dust, ash, corroded particles and organic matter was discovered. In the area, which had been unused for a long time, humus accumulation occurred, leading to the formation of an initial A horizon with a maximum thickness of 11 cm and the establishment of herbaceous vegetation (Blume 1986).

In Duisburg (Germany) with 490,000 inhabitants a shunting area, 95 ha in size and used between 1866 and 1986, was investigated in detail. The skeleton content ranged from 42% to 100%, the pH value from 6.5 to 7.1, in the absence of calcium carbonate the value for track gravel was about 4.5. The carbon content varied in the upper part between 0.5% and 48% and in the lower part results between 0.1 and 7.6 were found. The high carbon content, particularly typical for the areas between the tracks, was due to the ash deposits of earlier times, when steam engines were used and the ashes were swept away out of the engines (Fig. 3.6). Consequently, except for slag, the technogenic components ash and coke were found to a great extent. In turn, the ashes (see Section 4.1.2) were responsible for extremely high (1,030–1,670 mg $kg^{-1}$) total phosphorus concentration, while in principle the nutrient status (N, P, K) of the railway embankment was relatively low. It was possible to observe comparable results with regard to the pollutants. Between the tracks, where fine earth together with ash and coke residues were deposited over a long period of time, the contaminant results were as follows: As 1.8–60 mg $kg^{-1}$, Cd < 0.1–8.8 mg $kg^{-1}$, Cu 47–830 mg $kg^{-1}$, Pb 3–963 mg $kg^{-1}$, Zn 19–1,740 mg $kg^{-1}$, Polycyclic Aromatic Hydrocarbons ($PAH_{EPA}$) < 2–1,410 mg $kg^{-1}$, and cyanides <0.1–5.2 mg $kg^{-1}$, respectively (Hiller and Meuser 1998).

**Fig. 3.6** Abandoned shunting area with dismantled tracks (*gravelly surface*) and ash accumulation between the former track routes (*vegetated area*) in Duisburg, Germany

### 3.3.2 Utility Networks Pipes

In urban areas, especially underneath road surfaces, there are many pipes for infrastructure purposes (Fig. 3.7). They include wastewater pipes, water supply pipes, hot steam pipelines, telecommunications networks, electricity cables, gas pipelines, etc. They were constructed in underground trenches combined with excavation and backfilling procedures. Underground leakages of e.g. wastewater pipes and gas pipelines causing dispersion of soluble contaminants or reductive gases can be a source of soil contamination. In the case of leakages different kinds of damage are possible. For instance, corrosion, swelling effects of loamy or clayey soils, traffic influence or subsidence present in mining areas (see Section 6.1.6) can initiate losses of wastewater or gas.

Damaged wastewater pipes cause spill of polluted liquids migrating downwards to the aquifer. Typical compounds which cause concern are ammonia, boron, sulphate, chloride, phosphorus and heavy metals. Furthermore, the anaerobic wastewater may reduce the redox potential, hence reductomorphic features become visible. Heat emission of hot steam pipes accelerate the soil temperature leading to enhanced evapotranspiration, increase of thermophilic microbes and higher root growth in roadside green belts. Damaged gas pipelines may change the soil vapour conditions, leading to oxygen consumption and creation of reductive gases. Reductomorphological features appear (see Section 5.4.2), leading to a black or dark grey soil colour caused by pyrite formation, to enhance heavy metal mobility, to an increase of methanotrophic bacteria and to root damage, if roadside shrubs and trees are present. Purple root tips are a typical feature of reductive gases in root zones and, with reference to shallow

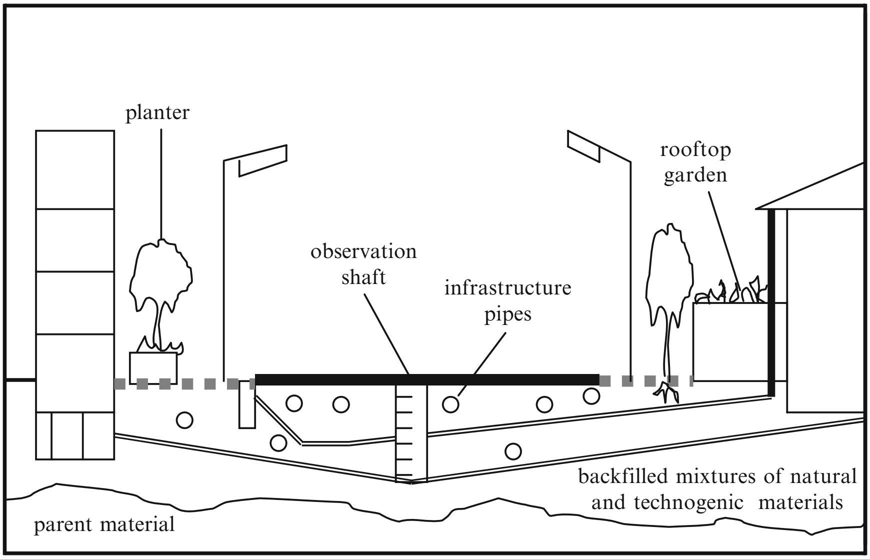



**Fig. 3.7** Urban soil used for infrastructure purposes (Modified from Pietsch and Kamieth, 1991)

groundwater tables, gas bubbles may have an impact on groundwater flow. The most important problem, however, may arise, if corroded gas pipelines volatilize gases like methane into the subsoil. Between 5 and 15 vol% and in the presence of oxygen the gas emission might lead to the danger of an explosion.

### *3.3.3 Floods in Alluvial Floodplains*

A high percentage of city agglomerations are located alongside rivers. For instance, in Europe the big, industrially influenced cities of Basel (Switzerland), Strasbourg (France), Ludwigshafen/Mannheim, Mainz/Wiesbaden, Cologne, Düsseldorf, Duisburg (Germany), as well as Nijmwegen and Rotterdam (The Netherlands) have been threaded along the Rhine river like a chain of pearls. Another impressive example is the Yangtze River in China, where several densely populated cities between Shanghai and Nanjing within a distance of approximately 300 km are located. There were historical reasons for locating cities in the vicinity of long rivers because rivers had to be used for goods transportation and fertile alluvial soils and river marshes for food production purposes. Locations enabling a favourable traffic situation and fertile soils in wide river valleys were suitable for early establishment of settlements.

Alluvial soils recognizable by yearly high groundwater level fluctuations, interactions between stream water and groundwater and periodical flood occurrences indicate a strong impact due to the city proximity:

- Discharges from industry and wastewater plants
- Surface water runoff from sealed sites
- Deflation and erosion from contaminated areas with bare or less vegetated soils
- Ship traffic, in particular ship accidents
- Dust deposition resulting from air pollution

The process of soil contamination in alluvial floodplains depends on several factors:

- Frequency and duration of flood occurrences
- Landscape topography
- Aquifer hydrology
- Soil properties

In general, there may be a tendency for subhydric usually loamy and humus-enriched sediments to accumulate contaminants, followed by alluvial loam, terrace sand and ultimately terrace gravel. In soils the adsorption potential of contaminants increases with increasing content of silt, clay, humus and iron oxides. Coarser material is deposited near the river channel and finer material settles down in the calmer floodwater, leading to higher adsorption potential and consequently to higher heavy metal content. This effect, however, can be disturbed by farmers' or gardeners' activities and bioturbation.

The dispersion of contaminants related to longitudinal direction and horizontal transverse axis should be explained in detail using the example of the river Strouma in the Pernik district (140,000 inhabitants), Bulgaria, southwest of the capital Sofia (Meuser et al. 2008; Meuser and Härtling 2008). There are a lot of natural resources in the district, especially brown coal (lignite) and limestone (suitable for metallurgy). The Pernik Coal Basin is the largest coal-production centre in Bulgaria and it spreads along both banks of the river Strouma. At the beginning of the twentieth century the industrial history began with the first discovery and simplified mining of the valuable lignite coal. After about 1955 the industrial development increased rapidly and other branches of industry were established, leading to the construction of steelworks, power stations, cement industry plots and glassworks (Fig. 3.8). After 1990 Bulgaria experienced considerable social, economic and ecological changes during the political transformation from a communist country to a democratic European state.

In the course of time the alluvial meadow soils were disturbed and changed anthropogenically. Near the stream water a lot of anthropogenic earthworks were deposited and the groundwater level was lowered during the construction of a reservoir called Pchelina. River straightening led to deep erosion of the river bed and former meanders were removed downstream. Furthermore, the river Strouma has served as a wastewater canal, since most of the industrial companies have discharged their poisonous effluents into the stream. The inevitable contamination of the Strouma floodplains may not only result from discharges but also from dust deposition of the strongly industrialized Pernik region (Fig. 3.9a and b).

In the lengthwise direction the lead concentrations revealed relatively low (background) values, whereas after entering the industrialized zone (steelworks) the lead values increased considerably. The concentration level did not change any more in the city area. It indicated a slight rise at site in the city centre. Behind the city the values seemed to reduce gradually.

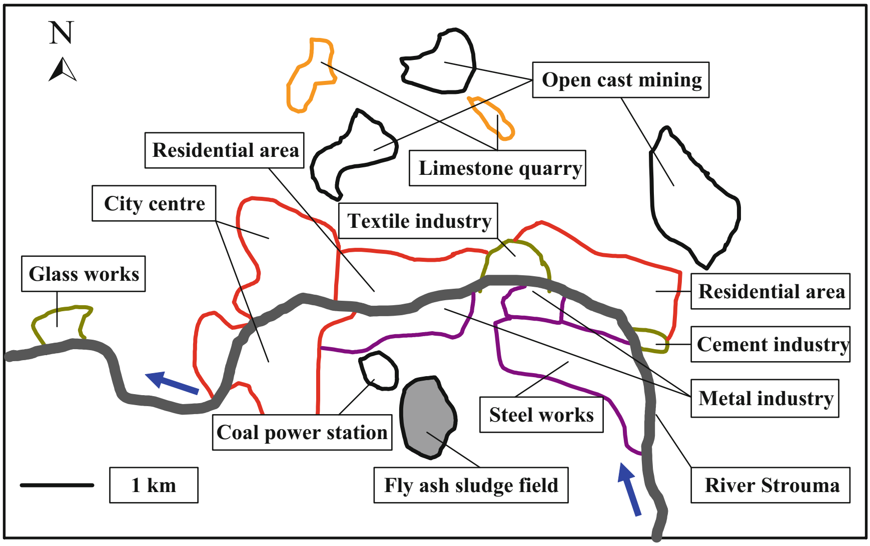



**Fig. 3.8** Heavily industrialized area of the city of Pernik, Bulgaria (Illustration based on a satellite image)

**Fig. 3.9** (**a**) Straightened and sealed river bed of the Strouma in the city of Pernik, Bulgaria influenced by discharge pipes and atmospheric deposition of an adjacent steelworks. (**b**) River bed of the Strouma in the city of Pernik, Bulgaria influenced by waste deposits

The effects of the industrial influence are clearer with regard to the sediment Pb concentrations (Fig. 3.10). The first investigated points 0–2 had low concentrations, completely changed from site 3. It was usually possible to detect very high numbers at sites 3 and 4, followed by decreasing values. After a second interruption in the city area the contaminant level was reduced in the lengthwise direction. Thus, an enormous influence of the steelworks was observable. Other elements typical for

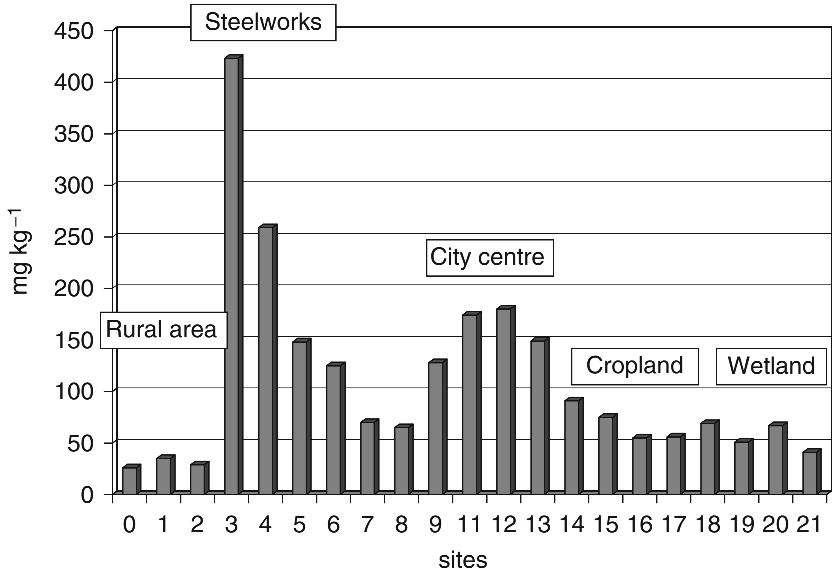



**Fig. 3.10** Sediment concentrations of lead (mg kg$^{-1}$) of the river Strouma, Bulgaria related to potential contaminant sources (Data from Meuser and Härtling 2008). Metal extraction method: aqua regia

the branches of industry in the Pernik region (Cd, Cr, Cu, Zn) reacted in the same manner, while metals such as Co, Ni, and Sn and some organic pollutants like the Polycyclic Aromatic Hydrocarbons (PAH) did not confirm the direct heavy industry influence (Table 3.15).

In spite of the local branches of industry in Pernik that are related to production processes emitting phenols, the phenol concentrations ranged between 0.5 and 2.8 mg kg$^{-1}$ only. It is supposed that the organic group well-known for its rapid biodegradation potential tended towards a relatively quick biological reduction, taking the period of approximately 15 years between the break-down of industry and the investigation period into account. Consequently, well-biodegradable parameters are not good indicators for the reporting of industrial history (see Section 2.2).

Metal concentrations of alluvial soils can be estimated as witnesses of the past. The German river Ruhr, for instance, had a long history of heavy industry since the Industrial Revolution. After the 1970s, however, most of the branches of industry trading with iron, coal and steel collapsed as a result of economic changes. Accordingly, the wastewater discharges were reduced and younger sedimented layers in the periodically flooded catchments revealed considerably lower pollutant values. Figure 3.11 presents two sites of the Ruhr river floodplain. The first one (Nos. 1 and 2) shows contaminant concentrations with increasing depth, just like the technogenic carbon content, particularly indicating coal and coal product origins (see Section 6.2.3). Thus, the deeper layers sedimented in the main industrial period of time had a higher concentration than the layers sedimented later. It should be taken into consideration that the explained relations can be influenced by soil properties

**Table 3.15** Sediment (depth 0–15 cm) pollutant concentrations (mg $kg^{-1}$) of the river Strouma, Bulgaria (Data from Meuser et al. 2008)

| No. | Cd mg $kg^{-1}$ | Co mg $kg^{-1}$ | Cr mg $kg^{-1}$ | Cu mg $kg^{-1}$ | Ni mg $kg^{-1}$ | Pb mg $kg^{-1}$ | Sn mg $kg^{-1}$ | Zn mg $kg^{-1}$ | $\Sigma$ $PAH_{EPA}$ mg $kg^{-1}$ | $\Sigma$ Phenols mg $kg^{-1}$ |
|---|---|---|---|---|---|---|---|---|---|---|
| S 0 | 0.7 | 8 | 25 | 72 | 23 | 26 | 5.5 | 90 | nd | 1.0 |
| S 1 | 0.6 | 17 | 29 | 62 | 31 | 35 | 6.6 | 104 | nd | 1.0 |
| S 2 | 0.6 | 12 | 29 | 52 | 26 | 29 | 6.3 | 86 | nd | 0.5 |
| S 3 | 12.8 | 13 | 135 | 168 | 51 | 423 | 18.1 | 1,750 | 1.7 | 1.4 |
| S 4 | 8.3 | 13 | 107 | 236 | 64 | 259 | 20.3 | 1,040 | 6.1 | 2.6 |
| S 5 | 5.3 | 11 | 66 | 173 | 46 | 148 | 13.4 | 624 | 4.5 | 1.4 |
| S 6 | 3.7 | 11 | 64 | 118 | 42 | 125 | 10.7 | 460 | 3.5 | 1.5 |
| S 7 | 2.0 | 10 | 46 | 75 | 31 | 70 | 7.0 | 234 | 1.8 | 1.0 |
| S 8 | 1.5 | 12 | 57 | 93 | 42 | 65 | 8.6 | 192 | 6.9 | 0.6 |
| S 9 | 4.3 | 11 | 65 | 149 | 47 | 128 | 10.7 | 582 | 2.3 | 1.9 |
| S 10 | no sampling opportunity (concrete) | | | | | | | | | |
| S 11 | 6.5 | 12 | 76 | 195 | 55 | 174 | 11.6 | 856 | 2.3 | 2.8 |
| S 12 | 2.1 | 12 | 131 | 99 | 53 | 180 | 7.9 | 388 | 5.0 | 1.3 |
| S 13 | 3.4 | 10 | 88 | 129 | 45 | 149 | 8.3 | 530 | 1.8 | 1.5 |
| S 14 | 2.4 | 10 | 48 | 111 | 41 | 91 | 7.4 | 395 | 2.2 | 1.6 |
| S 15 | 2.0 | 10 | 44 | 98 | 40 | 75 | 5.0 | 319 | 2.2 | 1.4 |
| S 16 | 1.4 | 9 | 39 | 76 | 35 | 55 | 4.7 | 201 | 2.5 | 0.6 |
| S 17 | 1.0 | 12 | 38 | 73 | 44 | 56 | 4.2 | 145 | nd | 0.7 |
| S 18 | 2.0 | 8 | 41 | 100 | 35 | 69 | 6.6 | 394 | 0.9 | 2.0 |
| S 19 | 1.5 | 10 | 43 | 80 | 36 | 51 | 5.6 | 266 | 1.4 | 1.5 |
| S 20 | 2.3 | 9 | 47 | 107 | 41 | 67 | 5.7 | 460 | 0.9 | 1.9 |
| S 21 | 1.2 | 11 | 40 | 70 | 39 | 41 | 4.7 | 213 | 1.3 | 0.8 |

nd = not detectable
Metal extraction method: aqua regia

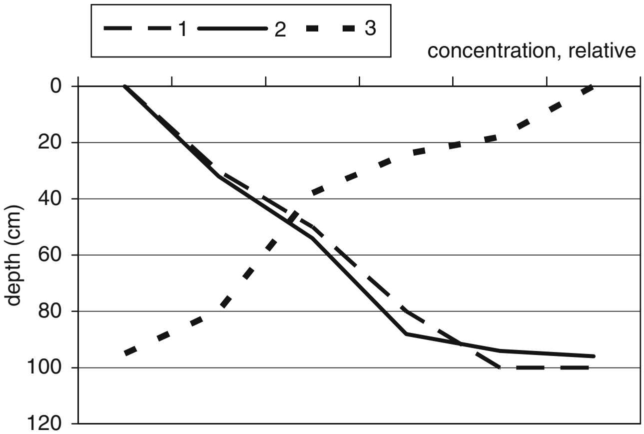



**Fig. 3.11** Schematic heavy metal profile of alluvial floodplain soils in the Ruhr area, Germany (Data from Meuser et al. 1996). Line 1: heavy metal concentration, sandy soil, catchment area with previous industrial use. Line 2: same site, technogenic carbon content (see Section 4.2.2). Line 3: heavy metal concentration, alluvial loam (0–40 cm) over terrace sand (>40 cm), catchment area with current industrial use. Metal extraction method: aqua regia

like texture. The other site (No. 3) revealed generally higher concentrations in the upper part of the profile, since alluvial loam with high adsorption potential overlaid terrace sand (Meuser et al. 1996).

The historical dimension of alluvial soil pollution had been discovered during the soil investigations in the Bulgarian catchments as well. As a consequence of the transformation, areas with an economic monostructure related to the industrial/military complex suffered particularly severe economic, social and ecological setbacks. During the transformation, a collapse of the economic activities occurred in this previously prospering area within a few years. The branches of industry collapsed completely, apart from the coal power station and a few other industrial complexes, creating a lot of highly contaminated brownfields. After the political changes a substantial decrease in the percentage of the processing industry (metal processing industry, mining industry) took place due to the decrease in the production of large enterprises.

The river Strouma flows into a water reservoir called Lake Pchelina just behind sampling point No. 21 (Fig. 3.10). The sediments in Lake Pchelina were fairly homogeneous sedimentologically. Unsupported $^{210}$Pb activities declined with depth in all cores (Fig. 3.12). However, the activities fluctuated considerably and did not decline exponentially ($^{210}$Pb half life is ~22 years). The bottom of the cores cannot be older than 1970 (construction of the reservoir) and correspondingly, background $^{210}$Pb levels were not achieved. This suggests that sediment mixing or slumping of very old sediments cannot have occurred. Similarly, the presence of the distinct $^{137}$Cs peaks suggests undisturbed sedimentation. These $^{137}$Cs peaks, that represent the 1986 Chernobyl accident, occurred at 24–26 cm (core 2), 22–24 cm (core 3) and 40–42 cm (core 4) core depth (see Section 3.5.2). A conservative estimate of 1.0 cm $a^{-1}$ was made for the sedimentation rate for cores 2 and 3 between 1970 and

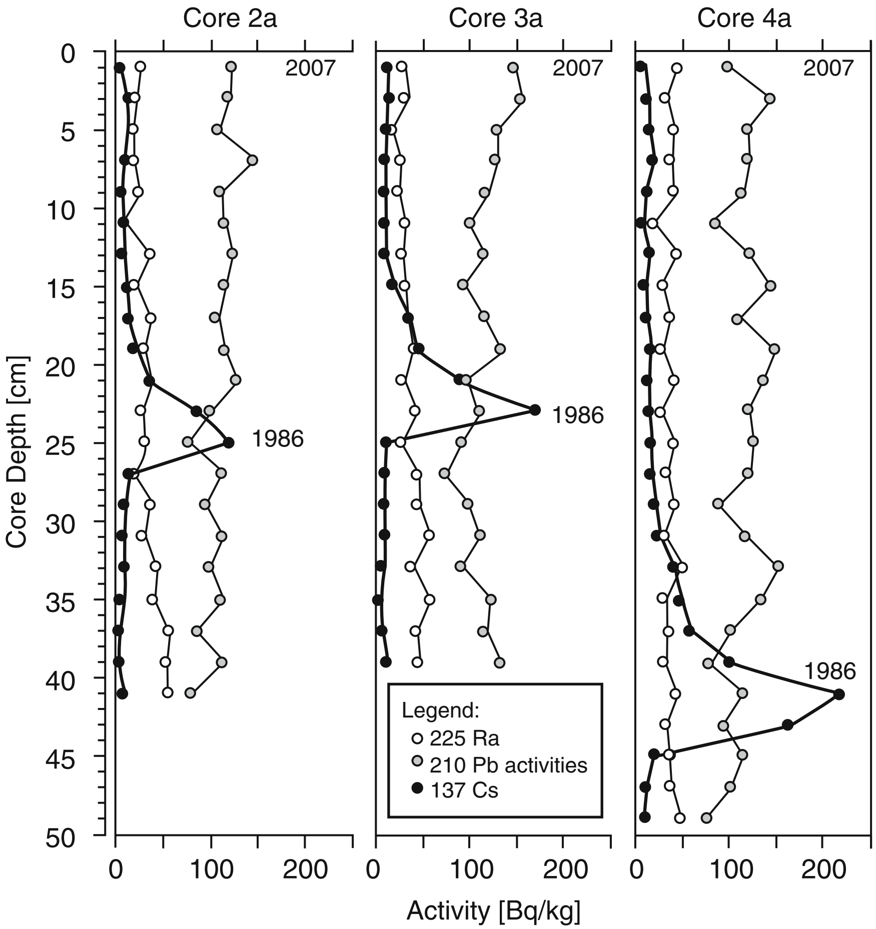



**Fig. 3.12** Total $^{210}Pb$, $^{226}Ra$ (representing background) and $^{137}Cs$ profiles (Bq kg$^{-1}$) of cores 2b, 3b and 4b from Pchelina Reservoir, Bulgaria (Data from Meuser and Härtling 2008)

1986. There was no indication of coarse materials or grading from slumps or turbidity currents during the initial years. Between 1986 and 2006 the sedimentation rates in cores 2 and 3 reached 1.2–1.3 cm a$^{-1}$ (uncorrected), while core 4 showed much higher rates (2.1 cm a$^{-1}$). The higher sedimentation rates at core 4 can be explained by the greater proximity to the inflow of the Strouma river.

The results of heavy metal extraction matched each other perfectly as the concentrations of most metals run parallel in all three cores, indicating similar post-sedimentary conditions (Fig. 3.13). Most of the metals showed a marked peak between 1986 and 1990 and a sharp decline thereafter. This tendency can best be observed in core 4, which is closer to the inflow of the river Strouma. Thus, we can assume that the environmental history of the Pernik area was reflected in the sediments of Lake Pchelina (Meuser et al. 2008).

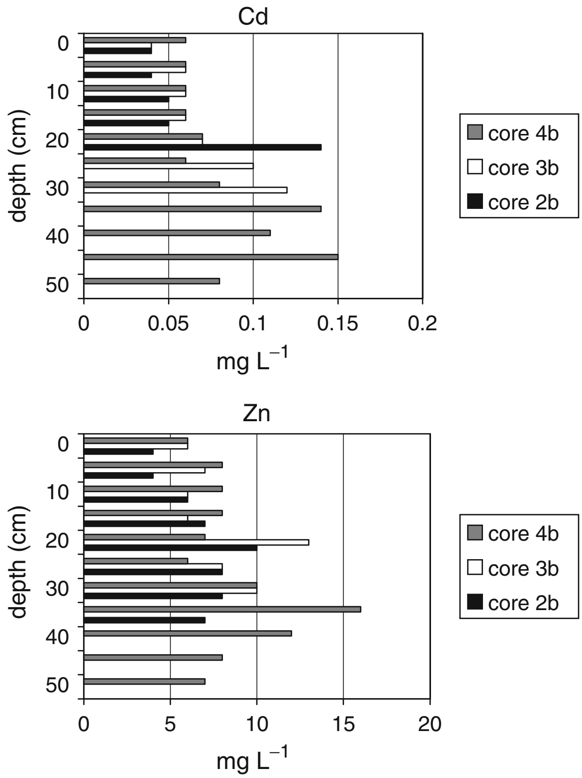



**Fig. 3.13** Concentrations (mg L$^{-1}$) of the trace metals cadmium and zinc of cores 2b–4b from Lake Pchelina, Bulgaria (Data from Meuser and Härtling 2008). Depth of 1986 line: core 2b: 25 cm, core 3b: 23 cm, core 4b: 42 cm. Metal extraction method: aqua regia

The historically explainable sediment contamination depends mainly upon the period of industry time. In spite of the huge urbanization process in China the sediments of the big rivers still seem to be moderately contaminated. Table 3.16 supplies information about the heavy metal concentration of the Yangtze River and the Huaihe River in Anhui Province, the longest and third longest river in China respectively, bordering different agglomeration areas. The latter, which is rather straight and wide and is not directly connected with industrial outlets and sewage disposal sites, was sampled at 18 locations. The sediment pH ranged from 7.9 to 8.9 and the organic matter content revealed results between 0.3% and 1.4%. The contaminant potential was relatively low and enhanced values related to some large amounts of solid waste and coal preparation plants (Yan et al. 2007).

In long-term industrialized catchments sediments and alluvial soils can also indicate reduced values with reference to possible thinning factors. In the first instance, this phenomenon is recognizable in estuarine areas, where mixing of uncontaminated marine with contaminated fluvial sediments occurs. The salt water penetration must exceed the landward transport of marine sediments. Subsequently, differences in

**Table 3.16** Heavy metal concentration (mg kg$^{-1}$) in river sediments of two Chinese streams, compared with background values of Anhui Province (Data from Yan et al. 2007)

| | Cd | Cr | Cu | Pb | Zn |
|---|---|---|---|---|---|
| Huaihe River | 0.17 | 56 | 22 | 20 | 70 |
| Yangtze River | 0.18 | 63 | 35 | 53 | 90 |
| Background values Anhui Province | 0.08 | 67 | 20 | 26 | 56 |

18 sampled sites (Huaihe River)
Metal extraction method: aqua regia

concentration of metals are visible, creating a considerable gradient. Good examples confirming that situation are the rivers Rhine and Meuse as part of the estuary in the Rotterdam catchments (The Netherlands) with a population of 590,000. As illustrated in Table 3.17, it was possible to locate the limit of salt penetration in the Holland Diep. From there the sediments consisted of a mixture of marine and fluvial origin. Rhine and Meuse components were mixed in a 2:1 ratio and homogenized. In the Rhine-Meuse estuary a dominant landward transport of marine sediments occurred, leading to a lowering of heavy metal concentrations. The tendencies led ultimately to a decrease in metal concentration downstream (Salomons and Mook 1977).

Research was carried out in garden areas where vegetables had also been grown in Dar es Salaam (Tanzania) with a population of 3,100,000. In this city four major industrial areas including workshops and garages were located. The solid waste collection and disposal infrastructure of the city area proved to be inadequate. Three rivers, namely the Msimbazi, Sinza and Ubungo, pass through the city and are exposed to rarely pre-treated industrial and household discharges. In their floodplains many vegetable garden plots were present. Results from the heavy metals analyzes of the topsoils are presented in Table 3.18. There was a tendency for areas close to the polluted rivers and nearby municipal dumpsites to indicate higher concentrations in comparison with open land. According to the soil results, the areas with higher metal values tended to show an enhanced uptake in vegetables as well (Luilo 2000).

Contrary to compounds which are well biodegradable, biologically more resistant organic pollutants like the Polycyclic Aromatic Hydrocarbons (PAH) seem to be more suitable for interpretation purposes of alluvial floodplains enriched with pollutants. Pies et al. (2007) sampled 22 sites along a distance of 169 km on the river Mosel and a further 20 km along the river Saar in addition to PAH analysis. The Saar river, an important tributary of the Mosel, flows through Lorraine (France) and Saarland (Germany), two regions with an intensive coal mining industry in the past. Both rivers have frequently burst their banks and visible coal remnants have permanently been spread onto the floodplain. Generally, human activities including the disposal of wastewater effluents took place. The investigations presented the historical point of view in floodplain terrains. Though the discharge of mine water decreased for several decades, an enhanced PAH level was recognizable due to previous mining activities. The measured values ranged from 0.1 to 81.5 mg kg$^{-1}$ $PAH_{EPA}$ and particularly the Saar river showed noticeably high values. However, it should be noted that the PAH parameters of this study included the 16 congeners of the US-EPA list and three additional PAH substances. Coal fragments combined

**Table 3.17** Mean metal concentrations (mg kg$^{-1}$) in various sediments of the Rhine-Meuse estuary and adjacent marine areas (Data from Salomons and Mook 1977)

| | | Cd | Cr | Cu | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|
| | Meuse (n = 14) | 28 | 216 | 160 | 44 | 382 | 1,516 |
| | Rhine (n = 20) | 14 | 642 | 294 | 54 | 533 | 2,420 |
| | Holland Diep 1 (n = 10) | 17 | 391 | 192 | 58 | 385 | 2,012 |
| | Holland Diep 2 (n = 10) | 17 | 383 | 185 | 59 | 354 | 1,875 |
| | Holland Diep 3 (n = 10) | 12 | 288 | 138 | 56 | 329 | 1,488 |
| | Haringvliet (n = 20) | 5 | 224 | 99 | 31 | 213 | 870 |
| | Grevelingen (n = 19) | 1.7 | 142 | 48 | 25 | 124 | 393 |
| | Zwarte Polder (n = 15) | 0.6 | 100 | 29 | 22 | 74 | 212 |
| | Vlaamse Banken (sea bottom) (n = 22) | 0.6 | 92 | 26 | 19 | 75 | 190 |
| Seaward direction | | | | | | | |

Metal extraction method: $HNO_3 + H_2SO_4 + HClO_4$ and $HNO_3$ (Cd, Pb)

**Table 3.18** Heavy metal concentration (mg kg$^{-1}$) in topsoils from various locations in Dar es Salaam, Tanzania (Data from different sources, cited in Luilo 2000)

| Site | Structure | Cd | Cu | Cr | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|
| | | mg kg$^{-1}$ | | | | | |
| Kiwalani | Open land | 0.5 | 27 | 4 | 5 | 12 | 109 |
| Ukonga | Open land | 0.8 | 28 | 4 | 6 | 12 | 101 |
| Sinza | Close to polluted river | 2.4 | 28 | 32 | 14 | 78 | 344 |
| Tabata | Old municipal dumpsite | 5.3 | 26 | 53 | 7 | 82 | 60 |

Metal extraction method: total concentration

**Table 3.19** $PAH_{EPA}$ content (mg kg$^{-1}$) of floodplain sites in different locations around the world (Data summarized and listed by Pies et al 2007, added by Meuser and Härtling 2008)

| Site | n | Range (mg kg$^{-1}$) |
|---|---|---|
| Alluvial soil, Rhine river, Germany | 18 | 0.02–3.6 |
| Sediments, Gironde estuary, France | 31 | 0.0–4.9 |
| Sediments, Yangtze estuary, China | 14 | 0.3–5.5 |
| Alluvial soil, Strouma river, Bulgaria | 20 | nd–6.9 |

nd = not detectable

with typical remnants of coking plant processes like coke and tar can be a source of PAH, but also a sink because of their high sorption capacity. Therefore, locations associated with coal mines and their related processing industries are often PAH polluted (Pies et al. 2007). Compared with floodplains in heavily industrialized regions, in which the coal industry is not predominant, the Saar and Mosel floodplains revealed relatively high PAH concentrations (Table 3.19).

## 3.4 Horticultural and Agricultural Influence

### *3.4.1 Fertilizing*

Because urban soils often show a lack of organic matter, the application of organic manures is recommended. In this way both humus application and nutrient supply can be well managed (Craul 1992). Table 3.20 gives information about the distinct organic materials usually applied to urban parks, gardens, etc.

The most commonly used organic material is compost derived from well-aerated piles, in which thermophilic composting takes place. In Central Europe (Austria, Germany and The Netherlands) 50–80% of the collected biodegradable municipal waste is composted. In Southern Europe (Greece, Italy, Spain) the figure is less than 15% and in Scandinavian countries and The United Kingdom it is between 5% and 25%. Besides, in some European countries household home composting is very common. In Austria, Germany and Luxembourg 25–40% of the organic garbage is composted in people's own gardens but in most European countries (e.g. France, Italy, Spain, United Kingdom), however, house composting was not very common until 2003. In some European countries there is an increasing tendency towards home composting in the current decade (Williams 2005).

Municipal waste composting technology starts with shredding and homogenisation. Afterwards compost piles enable rapid breakdown of organic matter, destruction of weed seeds and pathogens and water reduction. The decomposition process reaches temperatures between 50°C and 75°C, leading to a carbon loss of up to 75%. In relation to the initially piled material approximately 55% are degraded, 5% form part of the sieve residues and about 40% remains as compost material. Mineral nutrients are normally conserved. Disturbing artefacts have to be removed from the end product (Williams 2005).

**Table 3.20** Organic materials usually applied to urban open spaces (Data from own sources and Craul 1992)

| Material | Colour | pH | C/N ratio | Durability | Typical area to be applied |
|---|---|---|---|---|---|
| Hay | Brown to greyish brown | 5.5 | About 20:1 | Only one growing season | Urban agriculture, garden, park |
| Peat moss | Dark brown | 2.5–3.5 | Variable | Persistence for several seasons | Garden, cemetery |
| Straw | Yellow | 5.6–7.1 | 48–150:1 | Only one to two growing seasons | Urban agriculture, garden, park |
| Woodchips | White, yellow, grey | 4.1–6.0 | About 600:1 | 5–15 years | Park |
| Sawdust | Yellow | 3.5–5.5 | 130–930:1 | 5–10 years | Park |
| Compost | Dark brown | 6.0–7.0 | Less than 25:1 | 6–12 months | Urban agriculture, garden, park, cemetery |

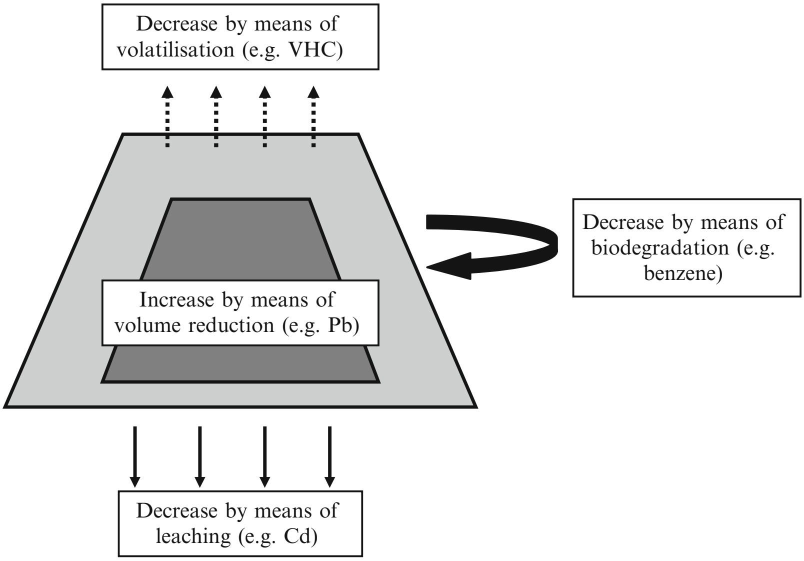



**Fig. 3.14** Contaminant changes during compost piling

The concentrations of potentially biodegradable organic pollutants decrease. Inorganic contaminants, however, are not destroyed during composting processes. In Fig. 3.14 the development in compost piles is illustrated.

Subsequently, in spite of some metabolized organic pollutants, compost products can be problematical depending on the origin of the organic material to be composted. Results of analysed composts derived from public green zones in the city areas of Essen with 580,000 inhabitants and Berlin exhibited enhanced contaminant values. In particular, compost consisting of litter collected at the side of streets in autumn to prevent traffic hazards, appeared to contain relatively high metal values (Table 3.21). For instance, benzo(a)pyrene emitted by diesel vehicles reached concentrations up to 2.7 mg kg$^{-1}$. In tendency, the results from Kolkata (India) with 15,400,000 inhabitants showed higher values than compost in German cities. Enhanced values were also visible in bark mulch often used in ornamental beds in wintertime to protect vegetation from frost. The metal accumulation is mainly caused by dust deposition onto plants, which are composted later on.

Generally, municipal solid waste compost exhibits the highest pollutant values. Average values based on analyses in Europe, North America, Australia and Asia (Table 3.22) were 4.5 mg Cd kg$^{-1}$, 162 mg Cu kg$^{-1}$, 318 mg Pb kg$^{-1}$ and 542 mg Zn kg$^{-1}$. Results from bio waste showed lower concentrations. The differences were marginal between bio waste compost and green waste compost (Williams 2005).

While in industrialized countries municipal solid waste is incinerated or land filled, municipal garbage of less developed and developing countries are used for soil fertilizing purposes. Because of a lack of modern waste handling and treatment opportunities there is a trend to compost solid waste on-site after removal of disturbing inorganic materials like glass and metals. Most of the waste is compostable if one thinks of paper,

**Table 3.21** Contaminant ranges (mg $kg^{-1}$) of composts in two German cities and one Indian city (Data from de Haan and van der Zee 1993 and unpublished results)

| Location | Material | As | Cd | Cu | Hg | Pb | Zn | Benzo(a)pyrene |
|---|---|---|---|---|---|---|---|---|
| Essen, Berlin (Germany) | Compost | 1–9 | 0.3–2.4 | 16–220 | 0.1–1.0 | 37–170 | 113700 | 0.1–2.7 |
| | Bark mulch | 5–25 | 0.1–2.6 | 2–32 | 0.2–2.0 | 7–23 | 19–500 | na |
| Kolkata (India) | Compost | – | 0.7–2.0 | 25–300 | – | 65–200 | 75–900 | na |

na = not analysed
36 (Germany) and 4 (India) analysed compost pits
Metal extraction method: aqua regia

**Table 3.22** Average total heavy metal concentration of compost from different waste sources (mg $kg^{-1}$) (Data from Hogg et al. 2002, cited in Williams 2005)

| Component | Municipal solid waste compost | Organic waste compost (bio waste) | Green waste compost |
|---|---|---|---|
| Cadmium | 4.5 | 0.9 | 1.4 |
| Chromium | 122.0 | 28.5 | 45.6 |
| Copper | 161.8 | 95.9 | 50.8 |
| Lead | 318.1 | 85.5 | 87.3 |
| Mercury | 1.6 | 0.6 | 0.5 |
| Nickel | 59.8 | 23.8 | 22.4 |
| Zinc | 541.5 | 288.5 | 186.4 |

food residues, tree leaves, etc. The nutrient content of the treated product, however, is rather low (Brady and Weil 2008). Composted waste will likely be increasingly applied to urban gardens and open green spaces in future, surely causing a new problem.

Additionally, attention should be paid to the mineral fertilizers as well. Cadmium and other problematical elements exist in materials that were used for fertilizer purposes. It is well-known that rock phosphates can be a carrier for unwanted cadmium. Continuous application to areas of urban agriculture and horticultural sites over a long period of time may enhance the Cd concentration irrespective of other typical anthropogenic sources of contamination in urban landscapes. Amendments of mineral and organic fertilizers in garden soils are significant with reference to potential contamination:

- Superphosphate, triplephosphate and rock phosphate contain increased cadmium concentration depending on the origin of exploration. In particular, phosphates derived from some African countries (e.g. Morocco, Senegal, Togo) are problematical (Osterhuis et al. 2000). The average values of rock phosphates were 7.8 mg $kg^{-1}$ and a content of 110–130 mg $kg^{-1}$ was even noted for triple superphosphate in fertilizers applied (Boysen 1992; Pierzynski et al. 2005).
- Apart from cadmium the metal chromium reveals accelerated concentrations in superphosphate and triplephosphate (average value: 273 mg $kg^{-1}$). This element is accumulated in basis slag produced in ironworks and frequently applied to cropland after technical conditioning. The average concentration is 1,759 mg $kg^{-1}$ (Boysen 1992).
- Some problematical concentrations are present in manures used in organic farming and occasionally used in areas for horticultural purposes like allotments and gardens, namely stone dust derived from basalt and serpentinite (Cr, Ni) (see Section 3.2.1), leather flour (Cr), and swine liquid manure (Cu) (Boysen 1992).

## *3.4.2 Application of Sewage Sludge and Wastewater*

The often intensively cultivated soils indicate high nutrient capacity and enhanced humus content. One reason for the nutrient accumulation is the application of sewage sludge that contains 2.0% N, 0.3% P and 45% organic matter and has been widely used on urban gardens and for land rehabilitation purposes. Sewage sludge applied to anthropogenic materials and soils improved not only the total nitrogen content but also the nitrogen mineralization by stimulation of the microbial activity. By means of sewage sludge addition the mineralized nitrogen increased significantly. After 16 weeks stockpiled topsoil revealed 392–938 mg N $kg^{-1}$, undisturbed topsoil 636–1,023 mg N $kg^{-1}$, mine spoil (pH > 6) 274–617 mg N $kg^{-1}$ and sand 40–273 mg N $kg^{-1}$, respectively (Pulford 1991).

The supply of nitrogen, phosphorus and sulphur leads to an improved nutrient balance which plants require in times of unfavourable soil and weather conditions. Related to the output of a cropping farm it was possible to provide about 60% of the necessary N and more than 100% of the P required by application of sewage sludge. In the case of compost it was possible to find even higher percentages (N: approx. 120%, P: approx. 140%) (Düring and Gäth 2002). Moreover, the application of sewage sludge improves physical soil characteristics like the water holding capacity, contributing to favourable root growth.

But continuous fertilizing with sewage sludge on agricultural and garden soils shows negative impacts as well. Continued application of sewage sludge can accumulate heavy metal concentration in soils, although the concentration of the sludge is not always very high (Craul 1992). Although the heavy metal concentration decreased for the last 2 decades, problematical values had to be taken into consideration. Consequently, continuous application resulted in aqua regia heavy metal enrichment. In particular, the application in former times, when the contamination level was even higher, caused an accumulation of heavy metals (Düring and Gäth 2002).

Very high values of up to 60 mg $kg^{-1}$ arsenic, 60 mg $kg^{-1}$ cadmium, 2,000 mg $kg^{-1}$ chromium, 1,400 mg $kg^{-1}$ copper, 240 mg $kg^{-1}$ lead, 385 mg $kg^{-1}$ nickel and 3,000 mg $kg^{-1}$ zinc were reported by Bridges (1991) for soils receiving sewage sludge over a long period of 74 years in Leicester (United Kingdom) with a population of 280,000. At other places garden soils received sewage sludge from septic tanks to a great extent.

The aqua regia concentration, however, is not of importance for mobility and bioavailability. Because of the organic character of the sewage sludge complexing agents like EDTA (ethylenediaminetetraacetic acid) are a better means for analysis purposes. Nevertheless, with increasing metal concentration, it was to be expected that the plant uptake would become higher. In this context, cadmium seemed to be the primary element in terms of food chain contamination (see Section 7.3.3). The uptake depended upon the species grown and, to be more precise, upon the part of the plant to be consumed by humans. Metal uptake was the highest in leafy plants and, in relation to cereals, higher in cereal leaves than in cereal grains. In general, an accumulation of heavy metals can be expected as a result of permanent or at least frequent application of sewage sludge, since in any case metal uptake is much smaller than input and thus residual accumulation (Düring and Gäth 2002).

A study dealing with long-lasting amendment of sewage sludge polluted by heavy metals usually detected a higher percentage of mobile fractions at three sites in Egypt called El-Gabal Asfar, Abou-Rawash and Helwan. The results showed that the total, bioavailable, mobile and non-mobile contents of Cd, Cu, Pb and Zn in soils increased as the irrigation period increased. Based on the sequential extraction method (see Section 6.2.1), the different chemical fractions revealed the prevalence of immobile pools, since they followed the order: organic > residual > oxides > carbonates > exchangeable. In spite of high movement of sewage effluent-burden heavy metals downward it was possible to analyze the soil profile. Irrespective of their predominant immobile fractions high availability of the four metals to plant was found. Thus, the concentrations of Cd, Cu, Zn and partly Pb in plants increased with the increase in the sewage sludge application rate (Badawy et al. 2005).

Although some organic contaminants have decreased in sewage sludge due to decreasing usage in industrial processes (e.g. Polychlorinated Biphenyls – PCB, Polychlorinated Dibenzodioxins and Dibenzofurans – PCDD/F), enhanced concentrations are currently present, because distinct organic pollutants enter the wastewater system and accordingly can be found in the sludge later on. Moreover, some pollutants like toluene may serve as a carrier for other organic pollutants like Polycyclic Aromatic Hydrocarbons (PAH), mobilizing the hydrophobic organic pollutants decisively. Investigations in Poland revealed the dependence of the location where the sludge was produced. In relation to the PAH in sewage sludge from rural regions contamination was negligible, whereas it was not possible to use sludge from industrial catchments in agricultural areas. Apart from the very important PAH group, toxic organic pollutants such as PCB and PCDD/F may play an additional detrimental role. On the other hand, the bioavailability of long-chain organic pollutants indicated low intensity and thus uptake by plants was usually not of concern.

With reference to the complete food chain there was a tendency for organic pollutants, however, to enter into the animal food chain of grazing sheep after application of sewage sludge to pasture. The longer the pollutants remained in the soil (aging process), the better the opportunity to biodegrade the substances. For this reason, application of sewage sludge that took place a long time ago sometimes did not result in high detected contaminant values (Düring and Gäth 2002).

Composting of the sludge may improve chemical properties and particularly the biodegradation of organic pollutants can definitely be considered. Therefore, composting of sewage sludge will play a bigger role in future. However, the sludge is usually stabilized with lime, causing increased pH values up to 11. Subsequently, the sludge seemed to be inappropriate for application purposes in gardens or other sensitive uses (Craul 1992).

In the near future the amount of sewage sludge may increase considerably, since in developing countries more and more municipal sewage treatment plants will be constructed. In China, for example, about 300 treatment plants exist today, producing approximately 300,000 t of dried sludge. But only less than 30–40 % of the sewage is treated at all (in 2005). The sludge is generally costly to handle and a final disposal in landfills is not planned. For this reason, the recycling of sludge in agriculture is going to increase, although it is well-known that the sludge produced in the cities of South China has high amounts of copper and zinc (1,500–5,000 mg $kg^{-1}$ each) (Wu et al. 2000).

The pollution problem cannot be reduced satisfactorily by composting and stabilizing. For a long time public authorities have begun to establish limit values in order to control the long-term pollutant accumulation, if sewage sludge was applied

**Table 3.23** Quality standards for maximum concentrations of contaminants in sewage sludges applied to soils in different countries (mg $kg^{-1}$) (Data collected by Düring and Gäth 2002)

| Pollutant | USA ceiling concentration | Russia | Germany | France | European Union (proposed) |
|---|---|---|---|---|---|
| As | 75 | – | – | – | – |
| Cd | 85 | 30 | 10 | 20 | 10 |
| Cr | 3,000 | 1,200 | 900 | 1,000 | 1,000 |
| Cu | 4,300 | 1,500 | 800 | 1,000 | 1,000 |
| Hg | 57 | 15 | 8 | 10 | 10 |
| Ni | 420 | 400 | 200 | 200 | 300 |
| Pb | 840 | 1,000 | 900 | 800 | 750 |
| Se | 100 | – | – | – | – |
| Zn | 7,500 | 4,000 | 2,500 | 3,000 | 2,500 |
| PAH[a] | – | – | – | – | 6 |
| PCB[b] | – | – | 1.2 | – | 0.8 |
| PCDD/F[c] | 300 | – | 100 | – | 100 |

Standards for metals based on total concentration
[a]Sum of ten congeners
[b]Sum of six congeners
[c]ng TE $kg^{-1}$

**Table 3.24** Quality standards for maximum concentrations of contaminants in soils to which sewage sludge will be applied (mg $kg^{-1}$) (Data collected by Düring and Gäth 2002)

| Pollutant | USA (calculated) | Germany (pH > 6) | France | European union (proposed, pH > 5) |
|---|---|---|---|---|
| Cd | 20 | 1.5 | 2 | 0.5–1.5 |
| Cr | 1,500 | 100 | 150 | 30–100 |
| Cu | 750 | 60 | 100 | 20–100 |
| Hg | 8 | 1 | 1 | 0.1–1 |
| Ni | 210 | 50 | 50 | 15–70 |
| Pb | 150 | 100 | 100 | 70–100 |
| Zn | 1,400 | 200 | 300 | 60–200 |

Standards for metals based on total concentration

to agricultural and horticultural sites in urban and rural environments. So-called quality standards were developed in a number of countries. Tables 3.23 and 3.24 give information about standards put in use in different states and soon in the European Union. In conclusion, the U.S. standards allow the highest thresholds, while in Germany relatively low values have been established. Organic pollutants are mostly not listed and the parameters of concern differ between the acts and ordinances. In the case of the proposed European Ordinance additional pollutants such as Linear Alkylbenzene Sulphonates (LAS) and Di-2-(ethyl-hexyl)-phtalate (DEHP) should be taken into consideration. In relation to heavy metals some ordinances include mobility determining parameters like pH value. Most of the regulations refer to both the sewage sludge and the soil to which the sludge will be applied.

It should be mentioned that the thresholds put in use nowadays will not prevent pollutant accumulation in soils continuously treated by sewage sludge in the past (Düring and Gäth 2002).

A special but probably widespread type of soil contamination occurs when either treated hospital wastewater or sludge from hospital and municipal sewage treatments plants is discharged on the soil. Besides soils, groundwater, surface water and sediments of rivers and lakes can be affected. Kümmerer (2004) reviewed the range of pharmaceuticals that are excreted by patients and unused medicines that are disposed of in wastewater in hospitals. Typical parameters are tetracyclines with concentrations between 150 and 900 μg $kg^{-1}$ in agricultural soils influenced by liquid manure (Thiele-Brunn 2003). To this should be added the personal care products like shampoos and soaps that likewise find their way into the wastewater and other segments of the environment.

Most pharmaceutical drugs are resistant to partial or complete mineralization through biodegradation. In general, pharmaceutical drugs are very poorly soluble in water and have high partition and adsorption coefficients. These properties contribute to strong adsorption to organic matter and the potential for bioaccumulation. Only if there are large pores and preferential flow paths antibiotics can move rapidly. The properties also make them unavailable for microbial degradation (aerobic and anaerobic biodegradation). They are very strongly bound to sludge in wastewater treatment facilities and to soil. Antibacterial antibiotics may potentially inhibit bacteria present. The epimers and metabolites of various antibiotics can react in soil quite differently to the original substances. Mobility and transport of antibiotics in the soil depends obviously on their water solubility so that those that have low solubility are strongly retarded (Klaschka et al. 2004).

Apart from sewage sludge in the urban environment industrial wastewater derived from industrial complexes in and around cities was frequently used for agricultural and horticultural purposes, particularly in less developed countries. A study focused on the investigation of heavy metal contamination of soils, plants and waters in rice fields influenced by industrial wastewater irrigation was carried out in Dhaka, Bangladesh. Raised levels of heavy metals were found in surface paddy soils with a range of 0.84–15.3 mg Cd $kg^{-1}$, 12–289 mg Cu $kg^{-1}$, 33–1,210 mg Pb $kg^{-1}$ and 128–7,600 mg Zn $kg^{-1}$. These high levels of metal concentrations were mainly due to the effluent of polluted wastewaters from the industry. In the rice fields close to the city, rice stalks, leaves and grain grown on those contaminated soils contained significant amounts of the metals, although the degree and extent of levels can vary depending upon the period of rice growth. Relatively high values were found under the oxidizing conditions rather than the reducing ones. This may be due to the difference in the metal uptake process during the growth of the plants (Hossain and Khondaker 2005).

Moreover, irrigation with industrial wastewater may alter a lot of chemical properties with beneficial and detrimental impacts. For example, soils irrigated with industrial effluent of ceramic, paper and starch factories revealed increased values of pH from 7.8 to 10.2 and electrical conductivity from 4.7 to 8.6 dS/m, impacting plant growth negatively (conducted in a greenhouse study in Ramadan and Ras Sudr

Research Station, Egypt). On the other hand, results showed that the irrigation with wastewaters increased soil organic matter from 0.7% to 1.2%. The maximum increases in total N and available P and K were found in soils irrigated by starch effluent. These increases amounted to 74%, 39% and 65% for total N, available P and K respectively. It should be taken into account that paper effluent resulted in remarkable increases of diethylenetriaminepentaacetic acid (DTPA) extractable Cd, Co, Cu, Ni, Pb and Zn in soils. In the case of Cu and Zn the increases reached 486% and 316%. In conclusion, the study indicated that particularly the effluents of the paper and ceramic industries should not be used directly as a source of irrigation (Eid et al. 2005).

### *3.4.3 Pesticide Application*

Pesticides are used in all parts of the world and more than 600 chemicals in about 50,000 formulations are applied. In agricultural areas pesticide application is widespread, but also in urban areas the chemicals are used. Pesticides consist of several categories.

The most important group is the herbicides preventing the growth of weeds in agriculture, but also in urban land-use types such as lawns and golf courses. They include triazines, substituted ureas, and carbamates. Furthermore, some roundups are applied which kill non-selectively almost all plants. Most of the herbicides are relatively well-biodegradable and low in mammalian toxicity.

Two further important categories involve the insecticides applied to kill and control insects that live on plants and in buildings and the fungicides that protect plants from diseases and buildings from fungi dispersion in rooms and cellars. Insecticides include chlorinated hydrocarbons extensively used worldwide till the 1970s and nowadays applied in less developed countries to a significant extent. Low biodegradability combined with persistence and high toxicity for birds, fish, mammals and humans are the reasons for restricted application in a lot of countries. Moreover, organophosphates which are toxic for humans and carbamates and which are less toxic for humans and easily biodegradable, are frequently sold. Fungicides such as thiocarbamates and triazines are also used in urban areas, where fruit and vegetables are grown (Brady and Weil 2008).

In urban environments rodenticides may play an important role, since they are related to mice and rats living in and around buildings. Other categories like bactericides and acaricides are mainly associated with agricultural use, apart from some uses on ornamental plants (Pierzynski et al. 2005).

Pesticides show different behaviours with regard to the environment. They can volatilize into the atmosphere. Furthermore, they can be absorbed preferentially by organic matter content, secondly by clay and oxides. Some herbicides such as diquat and paraquat show relatively high mobility, i.e. they move downwards to aquifer, particularly in soils indicating high pH values. Pesticides are capable of degradation by micro-organisms or by chemical decomposition. On the other hand, as mentioned before, they can be incorporated into the food chain. Unfortunately, in hilly regions surface runoff and possible following wash into water bodies, should be taken into account, since pesticides are applied aboveground and subsequently accumulated in the upper portion of the soil profiles. Some pesticides are taken up by plants to perform the intended function.

Irrespective of the metabolism in plant tissue, a harmful situation will exist, if pesticide residues remain in plants like vegetables and fruit which are eaten by man.

Pesticides are either imported from topsoils transported from outside the urban area or they are results of direct pesticide application in urban areas. In the first case pesticides are residues from topsoils of former agricultural and horticultural land, in the second one property owners, lawn care firms and public authorities are in charge of the application. It has been found that core city soils normally have higher pesticide concentration than those in the suburban areas. But it should be taken into account that drift from adjacent agricultural sites can influence urban land like residential areas. In forests only 25% of the aerially applied chemicals reach the tree foliage and approximately 30% reach the soil (Brady and Weil 2008).

If pesticides not capable of quick biodegradation reach the soil surface, the food chain movement up can take place. Earthworms ingest the contaminated soil, afterwards birds and fish consume earthworms and the transport to humans consuming fish for example is feasible. Accumulation can ultimately build up to lethal concentrations.

Especially herbicide-laden dust can be very harmful to gardens and other residential surfaces. Garden owners have frequently to suffer plant damage in their garden. Distorted growth of leaves and stems indicating twisting, bending and curling, visible symptoms such as interveinal chlorosis, off-colour foliage and necrosis and finally plant death were reported to be typical results from herbicide drift into ornamental gardens (Craul 1992).

Gardeners, however, are not only victims of pesticide application – in a way they are sources of contamination as well. In the United States an annual application of pesticides in agriculture there was supposed to reach 0.56 million Mg in order to control weeds, insects and diseases. But an estimated 25,000 Mg of pesticides are used annually in a non-agricultural application, primarily in the urban environment, to control pests in lawns, flowerbeds, golf courses, urban forest parks and along waterways, utilities networks and ultimately rail embankments (Pierzynski et al. 2005).

The use in urban areas led to contamination of soils and groundwater, as investigations in the USA showed. In cities herbicides such as Diuron, Bromacil, 2,4-D and Simazin were frequently detected in soils and aquifers. For instance, in 1.4% of detections wells Simazin was found with a maximum value of 1.3 μg $L^{-1}$. There is no doubt that in agriculturally used areas much higher concentrations had to be analysed, e.g. atrazine was detected in 38% of the detection wells with a maximal concentration of 3.8 μg $L^{-1}$. Nevertheless, the pesticides problem is of importance in urban environments as well (Brady and Weil 2008).

## 3.5 Urban Influence

### *3.5.1 Derelict Land*

Abandoned industrial sites are commonly harmful to wildlife and they may cause damages to building structures. Such areas termed derelict land exhibit detrimental features. The areas show complex topography due to holes and piles in direct proximity and sometimes subsidence is present. A high percentage of bare unvegetated soil is visible, leading to erosion and deflation. The most important problem is the presence of toxic solid and fluid substances. Finally, the sites make the landscape unsightly since they consist of ruined and damaged buildings as well as removed or damaged infrastructure. In such an environment one can expect a general depressed social state with absenteeism of owners, poor housing, etc. (Craul 1992). Some examples for derelict land are illustrated in Fig. 3.15a–d.

The inventory of contaminants depends on the kind of industry present in the past. There are a number of reasons leading to enhanced contaminant values in soils:

- Careless handling and storage of hazardous substances over longer periods of time
- Simplified production technology tending to damages and uncontrolled leakages
- Construction activities during processing time, causing excavation, transport and backfill of possible contaminated soil materials on-site
- Bomb attacks during wars
- Accident disasters

In Tables 3.25–3.27 industry branches and their potential discharges of contaminants are summarized. In heavily industrialized areas a number of abandoned derelict sites can exist in direct proximity, as shown in Pernik (Bulgaria) with a


**Fig. 3.15** (**a**) Example for an urban derelict land: ruined former furniture industry plot nearby Sofia, Bulgaria. (**b**) Example for an urban derelict land: ruined former foundry area in Szczecin, Poland. (**c**) Example for an urban derelict land: ruined former metal works in Dortmund, Germany. (**d**) Example for an urban derelict land: damaged railway area with tar conditioned sleepers in Dortmund, Germany

**Table 3.25** Potentially emitted industry-specific contaminants (metals) (Data from different sources)

| Branch | As | Be | Br | Cd | Co | Cr | Cu | Hg | Ni | Pb | Sb | V | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Coal mine | | | | | | | | | | | | | |
| Coking plant | X | | | | | X | | | | X | | | X |
| Ore mine | | | | | | | | | | | | | |
| Heavy metal mine | | | | X | | X | X | | | X | | | X |
| Salt mine | | | | | | | | | | | | | |
| Crude oil extraction | | | | | | | | | | | | | |
| Oil refining | X | | | | X | X | X | | X | X | | X | |
| Brickworks | | | | | | | | | | | | X | X |
| Cement works | X | | | X | | X | | | X | X | | | |
| Glassworks | X | | | X | X | X | X | | X | X | X | | X |
| Steelworks | X | | X | X | X | X | | X | X | X | | X | X |
| Aluminium works | | X | | | | X | | | | | | X | |
| Heavy metal works | X | | | X | X | X | X | X | X | X | X | | X |
| Foundry | X | | | X | | | X | X | X | X | X | X | X |
| Metal processing works | | | | X | | X | X | | X | X | X | X | X |
| Car manufacturing | | | X | X | | X | X | | | | | | X |
| Engineering works | | | | X | | X | X | X | X | | | | |
| Shipyard | | | | | | | X | | | X | | | |
| Basic chemicals industry | X | X | X | X | X | X | X | X | X | X | X | | X |
| Foodstuffs industry | | | X | | | | | | | | | | |
| Textile industry | | | X | | | X | X | | | | | | X |
| Leather industry | | | X | | | X | | X | | | | | |
| Dye-works | X | | | X | X | X | X | X | | X | X | | X |
| Fertilizer production | X | X | | | | | | | | | | | |
| Pesticide production | X | | X | | X | | X | X | | X | | | X |
| Battery production | X | | | X | | X | X | X | X | X | X | | X |
| Timber industry | X | | | | X | X | X | X | X | | | | X |
| Printing works | X | | | | | X | X | | | X | | X | X |
| Explosive production | X | | | | | X | X | | | X | X | | |

**Table 3.26** Potentially emitted industry-specific contaminants (acids, anions, organic pollutants) (Data from different sources)

| Branch | $H_2SO_4$ | $HNO_3$ | HCl | $SO_4^{2-}$ | $Cl^-$ | $F^-$ | $CN^-$ | TPH | BTEX | Phenols | PAH |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Coal mine | | | | | | | | X | | | |
| Coking plant | X | | | X | X | | X | | X | X | X |
| Ore mine | | | X | | | | X | | | | |
| Heavy metal mine | | | X | | | | X | | | X | |
| Salt mine | | | | | X | | | | | | |
| Crude oil extraction | | | | | | | | X | | | |
| Oil refining | X | | | X | X | | | X | X | X | X |
| Brickworks | | | | | | X | | | | | |
| Cement works | | | | X | | X | | | | | |
| Glassworks | | | | | | X | X | | X | | |
| Steelworks | X | | X | | X | X | X | X | | X | |
| Aluminium works | | | X | | | X | | | | | |
| Heavy metal works | | X | X | | | X | | | | | |
| Foundry | | | X | X | | | | | | X | |
| Metal processing works | X | | X | X | | X | X | | X | | |
| Car manufacturing | | | | | | X | X | | | | |
| Engineering works | X | | | | | X | X | X | X | | |
| Shipyard | | | | | | | | | | | |
| Basic chemicals industry | X | X | X | X | | X | X | | X | X | X |
| Foodstuffs industry | X | X | X | | X | X | | | | | |
| Textile industry | X | | X | X | X | | | X | | X | |
| Leather industry | X | | X | X | X | X | | | | X | X |
| Dye-works | | | X | X | | X | | X | X | X | X |
| Fertilizer production | X | X | | X | X | | | | | | |
| Pesticide production | | | | | | X | | | X | X | X |
| Battery production | X | | | | X | X | | | | | |
| Timber industry | | | X | | | X | | X | X | X | X |
| Printing works | | | X | | | | | | X | | |
| Explosive production | X | X | | | | | | | | | |

TPH = total petroleum hydrocarbons; BTEX = benzene, toluene, ethyl benzene, xylene; PAH = polycyclic aromatic hydrocarbons

**Table 3.27** Potentially emitted industry-specific contaminants (organic pollutants) (Data from different sources)

| Branch | Volatile Hydrocarbons | HCB | PCP | PCB | PCDD/F | Pyridine | Nitro-aromates | Organic acids |
|---|---|---|---|---|---|---|---|---|
| Coal mine | | | | | | | | |
| Coking plant | | | | | | X | | |
| Ore mine | | | | | | | | |
| Heavy metal mine | | | | | | | | |
| Salt mine | | | | | | | | |
| Crude oil extraction | | | | | | | | |
| Oil refining | X | | | X | X | X | | |
| Brickworks | | | | | | | | |
| Cement works | | | | | | | | |
| Glassworks | | | | | | | | |
| Steelworks | | | | | | | | |
| Aluminium works | | | | | | | | |
| Heavy metal works | | | | | | | | X |
| Foundry | | | | | | | | X |
| Metal processing works | X | | | | | | | X |
| Car manufacturing | X | | | | | | | X |
| Engineering works | X | | | X | X | | | X |
| Shipyard | | | | | | | | |
| Basic chemicals industry | X | | X | X | X | X | X | X |
| Battery production | | | | X | | | | |
| Foodstuffs industry | X | | | | | | | X |
| Textile industry | X | | X | | | X | | X |
| Leather industry | X | | | | | | | X |
| Fertilizer production | | | | | | | | |
| Pesticide production | X | X | X | | X | X | X | X |
| Explosive production | | | | | | | X | X |
| Dye-works | X | | X | X | | | X | X |
| Printing works | X | | | | | | | |
| Timber industry | X | | X | X | X | X | | X |

HCB = hexachlorobenzene; PCP = pentachlorophenol; PCB = polychlorinated Biphenyls; PCDD/F = polychlorinated dibenzodioxins and -furans population of 130,000 (see Section 3.3.3). In industrial areas industry-specific quantity and dispersion of some contaminants are expected. It should be noted that the contaminants can potentially reach sensitive media like soil, groundwater and plants. If former abandoned industrial plots are investigated, a detailed soil survey and contaminant analysis is necessary aimed at verification of the contaminant parameters mentioned in Tables 3.25–3.27.

In developed countries the public authorities have begun to register contaminated land. Some examples should be presented in order to give an overview of the current situation of contaminated land involving both old derelict installations and deposits (see Section 3.5.3). In Denmark, for instance, approximately 30,000 potentially contaminated sites were registered in 2000. Most of the sites were waste dumpsites, followed by contaminated soils of petrol stations (Fig. 3.16) (DEPA 2001). In Swedish statistics relating to derelict land the most important branches contaminating soils were named as the wood and paper industries as well as the metal-working industry (Fig. 3.17) (Prokop et al. 2000). The comparison of the Scandinavian countries shows the state of industry, because in Denmark important polluters were petrol stations that released mobile oil derivates into the near-surface groundwater present all over the country. In Sweden, which is wooded to large extent, the wood and paper industry played the most important role, as was to be expected.

In Romania, which was heavily industrialized in an environmentally unfriendly way during the socialist era, the metallurgical industry was mainly associated with contaminated sites. In total, in 2007 about 402,000 ha were assumed to be contaminated and of these 33,000 excessively and 19,000 severely contaminated. Approximately 224,000 ha were only slightly contaminated, most of them as a result of wind erosion from derelict land into neighbouring areas. The detailed division is shown in Fig. 3.18 (MEDD 2008).

Outside Europe the registration and publication of contaminated land is not very common. For instance with regard to different land-use types in the year 2003 in Japan most of the contaminated sites (about only 320 in number) belonged to continuously producing and abandoned factories, whilst residential areas fell below 50 sites in total (MOE 2006).

The aim of the following paragraphs is to introduce some branches of industry by way of example, namely an iron and steel production company, a battery plant, a mercury production plant, a metal recycling yard and a chemical industry area.

Investigations of an iron and steel production site occupying approximately 300 ha and closed in 1991 were conducted in Bagnoli near Napoli (Italy) with a population of 1,000,000. The soils were considerably disturbed and contained materials used in production processes such as iron pellets, iron minerals and coke as well as waste material such as slag. A number of soil profiles were taken into account. Typically, one profile chosen revealed anthropogenic layers to a depth of 72 cm underlaid by natural material. The pH(KCl) values ranged from 6.9 to 8.1 in the deposited layers. While the upper three layers indicated few technogenic fragments only, the fourth one was enriched by technogenic components like slag. This layer had the highest Cu (more than 900 mg $kg^{-1}$) and Pb (more than 650 mg $kg^{-1}$) values.

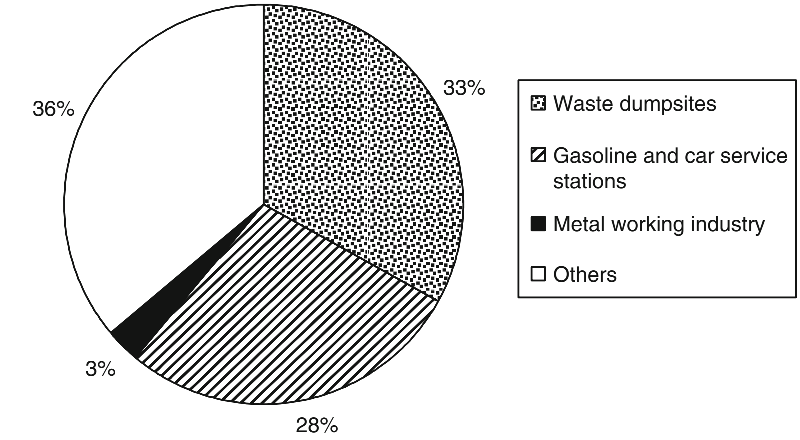



**Fig. 3.16** Polluter sources responsible for registered soil contamination in Denmark (Data from DEPA 2001)

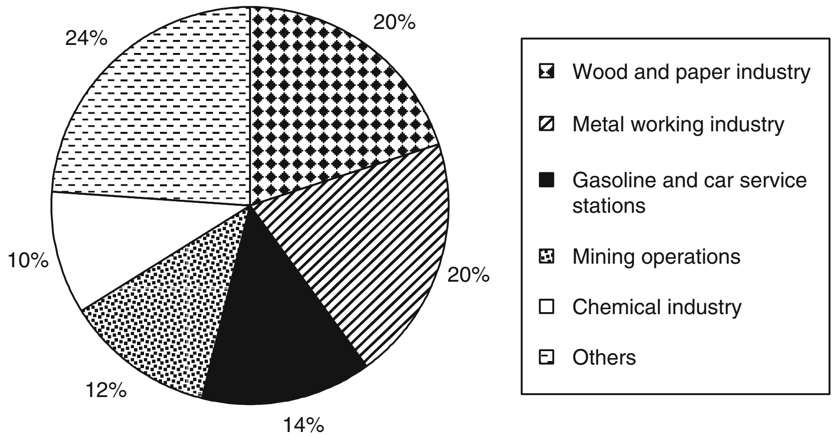



**Fig. 3.17** Polluter sources responsible for registered soil contamination in Sweden (Data from Prokop et al. 2000)

Cobalt and zinc did not strongly correlate to the technogenic ingredients but the elements were generally concentrated within the deposited layers. Chromium and nickel, parameters closely linked to steelworks, exhibited high concentrations in the upper dust influenced horizons (0–28 cm) and reached maximum values higher than 2,400 (Cr) and 950 (Ni) mg $kg^{-1}$ (see Sections 3.3.3 and 4.3).

Based on sequential extraction the exchangeable percentage was found to be very low and most of the metal quantity was bound by iron and manganese oxides and concentrated in the residual fraction as well. However, it was possible for Fe-rich small particles, which adsorb heavy metals to a great extent, to migrate through the sandy and gravely deposited soil profiles in colloidal form (Adamo et al. 2000).

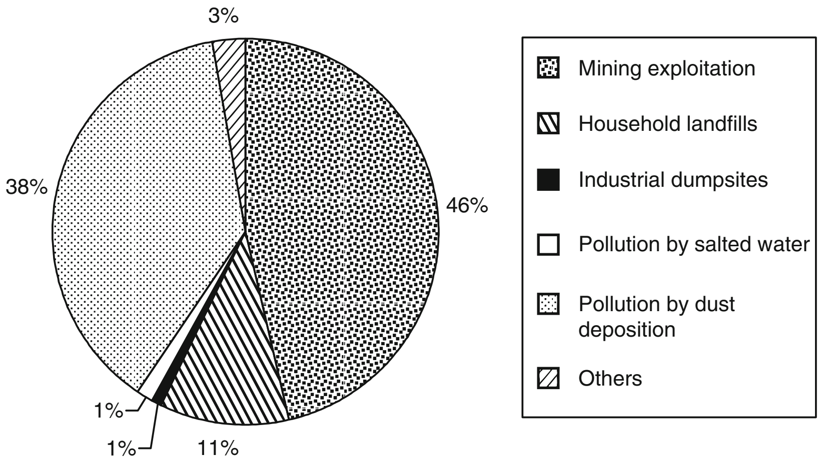



**Fig. 3.18** Reasons for severely and excessively contaminated registered sites in Romania in 2007 (Data from MEED 2008)

**Table 3.28** Arithmetic means of heavy metals (mg $kg^{-1}$) in soils adjacent to a battery plant related to directions from emitter in Piastow, Poland (Data from Apolinarska and Grzebisz 2000)

| | West | | | East | | |
|---|---|---|---|---|---|---|
| | Main wind direction | | | Counter current main wind direction | | |
| Depth (cm) | Cd | Pb | Zn | Cd | Pb | Zn |
| 0–30 | 2.0 | 199 | 156 | 1.9 | 132 | 79 |
| 31–60 | 1.4 | 96 | 102 | 1.9 | 98 | 73 |
| 61–90 | 0.6 | 62 | 79 | 0.3 | 42 | 33 |

Metal extraction method: $HNO_3$

Locally contaminated soils are usually closely connected with specific production processes potentially caused by two different sources. In Piastow with 20,000 inhabitants, located south of Warsaw (Poland), an abandoned battery plant producing from 1925 to 1999 was investigated. This kind of industry is well-known for its emission of lead (up to 0.6 t $year^{-1}$ until 1997) and zinc as well as to a lesser degree barium, cadmium, copper, nickel and strontium. The topsoils (0–30 cm) in the main Western wind direction showed high Cd, Pb and Zn values in contrast to the results yielded in the Eastern direction due to chimney stacks (Table 3.28). This was additionally underlined by the distance to the plant, since adjacent topsoils were more highly contaminated than topsoils in areas far away. In the subsoil enhanced concentrations were discovered. At a depth of 31–60 cm, for instance, in an Eastern direction, relatively high values for Cd and Zn were analyzed. Because dust deposition cannot explain these results, deposited processing waste as a second source might be the reason in the sandy subsoils (Apolinarska and Grzebisz 2000).

Mercury is not one of the typical elements usually found in the urban environment. Especially in dry climatic conditions this is explainable by Hg volatilization, which may reduce Hg concentration relatively rapidly. Furthermore, some technogenic substrates such as ashes and slag might indicate low mercury levels resulting from combustion processes and subsequently enhanced gaseous removal (see Section 4.3). However, so-called Hg hot spots have occasionally been found in urban and industrial areas. For instance, Hg contamination of soils close to an abandoned Hg production plant was investigated in La Manjoya in the vicinity of Oviedo (Spain) with 400,000 inhabitants. There, extremely high concentrations of Hg (up to 30 g $kg^{-1}$) were found in soils located next to the discharge area of the wastewater produced during the washing procedures at the production plant. Elemental Hg could even be visually identified in the soils where the wastewater used to run downstream. Away from this runoff trail, Hg concentrations decreased to levels ranging between 1 and 5 g $kg^{-1}$. Mercury concentrations were also high in the surroundings of the production plant, with values above 1 g $kg^{-1}$ at a depth of 1 m. Mercury contamination of soils away from the wastewater discharge area could be attributed to spills from Hg containers and Hg volatilisation in the production process with subsequent condensation in cooler areas of the production plant and in the surrounding forest stands (Macias et al. 2005).

In a city in Southern Finland (location not published) a metal recycling yard in production for more than 20 years was included in environmental studies. There, junk metals were mechanically cut and lead batteries were broken and smelted. Most metals revealed considerable amounts when sampled representatively at 50 locations and subsequently analyzed by aqua regia and $NH_4$ acetate indicating the available portion. In particular, copper and lead showed extremely high values. Some metal concentrations of both extraction liquids correlated clearly, such as copper ($r = 0.97$), lead ($r = 0.99$), and zinc ($r = 0.88$). The median and maximum values are listed in Table 3.29. Apparently, in the presence of a very high concentration performed by total extraction a problematical available portion has to be supposed. The potential leachability was confirmed by the comparison of two observation wells upstream and downstream. The latter one exhibited very high concentrations for most of the elements under question (see Section 7.3.3) (Lintinen 2000).

The potential leachability of metals in soils with very high metal concentration was also found in an area composed of several chemical industries in Estarreja

**Table 3.29** Metal concentration (mg $kg^{-1}$) of 75 soil samples and results from two groundwater observation wells (μg $L^{-1}$) in a metal recycling yard in Southern Finland (Data from Lintinen 2000)

| | Aqua regia | | $NH_4$ acetate | | | |
|---|---|---|---|---|---|---|
| | Mean | Maximum | Mean | Maximum | Well upstream | Well downstream |
| As | **18** | 22 | **1.2** | 1.7 | 0.1 | 12.6 |
| Cd | **3.8** | 12 | **2.0** | 19.8 | <0.1 | 6.4 |
| Cr | **114** | 5,050 | **0.3** | 1.3 | <0.2 | <0.2 |
| Cu | **452** | 11,000 | **75.0** | 2,690.0 | 0.7 | 8.8 |
| Ni | **68** | 2,000 | **1.2** | 9.2 | 0.7 | 16.5 |
| Pb | **1,156** | 27,000 | **654.0** | 19,900.0 | 0.1 | 11.7 |
| Zn | **184** | 6,320 | **39.2** | 725.0 | 5.9 | 472.0 |

75 sampled soils to a depth of maximum 100 cm

(Portugal) with a population of 8,000. Thirty-four topsoil values pointed out extreme total concentrations such as arsenic (mean value 3,296 mg $kg^{-1}$/maximum value 52,050 mg $kg^{-1}$), mercury (115.8 /1,800 mg $kg^{-1}$), selenium (8.1/97.6 mg $kg^{-1}$) and zinc (466/3,989 mg $kg^{-1}$). Based on the $NH_4$ acetate method, the mobile percentage accounted for less than 2% for As and Hg, 6.9% for Se and 5.2% for Zn. Obviously, the percentage can be assessed as sufficient, because forage plants (grasses) sampled in the surrounding area of the industrial complex indicated metal accumulation. Results (mean and range in mg $kg^{-1}$ DM) from shoots were 3.7 (0.1–23.5) for As, 0.2 (<0.1–1.9) for Hg, 0.6 (0.1–3.9) for Se and 91 (25–444) for Zn. Although cadmium exhibited relatively low values (soil total concentration: 1.4 (<0.1–11.4) mg $kg^{-1}$), the $NH_4$ acetate extractable percentage was 21.6% and subsequently it was possible to recognize a slight accumulation in green shoots (Inacio et al. 2000).

Areas used for military purposes may reveal general high disturbance and often soil contamination based on the former utilization. Military structures such as emplacements of firing positions, shelters, bunkers, heliports, airports and target areas were built predominantly causing man-made soils. Some sites like shooting target areas were seriously damaged by explosions. Movements of tanks and heavy transporters altered the soil visibly, overlaying procedures occured in order to make shell splinters disappear. Finally, natural soils and vegetation cover were damaged or even virtually destroyed. Apart from the physical damage, possible pollution by heavy metals, oil products and other chemical contaminants can be recognized, in particular in machinery centres and barracks.

In the Eastern European countries soil contamination caused by the former Soviet Army and discovered just after departure of the Army meant considerable problems related to soil rehabilitation and re-use for civil purposes. In the Czech Republic about 50 seriously contaminated sites were identified, indicating high levels of toxic substances such as petroleum hydrocarbons and chlorinated hydrocarbons. In Estonia it was reported that leakages from fuel oil storage vessels and pipelines used by the Soviets caused thousands of tons of black oil to leak and flow from railway tanks and storage vessels without any control. In Hungary approximately 150 sites were found, in the first instance Soviet barracks and training grounds, that were highly contaminated and needed even urgent remediation. In Poland it is estimated that 70,000 ha occupied by the Army are contaminated with petroleun hydrocarbons due to oil spillages and with hazardous toxic substances due to illegal waste dumping and storage of toxic materials (Duffield et al. 2000).

The kind of long-term industrial utilization may decide on the soil composition and subsequently soil contamination. This context is related to both the civil industrial sector and the military sector. One impressive example explaining the contamination level derived from materials used over a long period of time is the situation of shooting ranges, as investigated several times. The deposition of bullets and pellets in military and recreational shooting ranges led to an accumulation of, for instance, Pb. Although this heavy metal is supposed to be less water-soluble and mobile in soil, the metallic core of the bullets may be quickly oxidized and physically weathered, releasing Pb in great amounts. Apart from preferential flow, soils with low pH values can initiate rapid percolation of Pb-laden soil water downward. In the region of

**Table 3.30** Chemical properties of a civil and military shooting range (woodland, 50 m behind the target area) near Lake Maggiore, Switzerland (Data from Knechtenhofer et al. 2003)

| Depth (cm) | pH ($CaCl_2$) | C content (%) | Pb (mg $kg^{-1}$) | Sb | Cu | Ni | Zn |
|---|---|---|---|---|---|---|---|
| Organic litter | 3.2 | 33.1 | 12,533 | 676 | 149 | 44 | 87 |
| 0–10 | 3.6 | 14.8 | 2,908 | 30 | 96 | 39 | 65 |
| 10–50 | 4.2 | 8.9 | 85 | 1.9 | 12 | 35 | 57 |
| 50–60 | 4.7 | 2.7 | 24 | <0.4 | 10 | 51 | 67 |
| 60–70 | 4.8 | 1.3 | 22 | <0.4 | 13 | 64 | 59 |

Metal determination: X-ray fluorescens spectrometry

Lake Maggiore, Switzerland, normally well-known for its tourism arrangements and consequently densely populated in the summer season, shooting ranges used over a long period of time were investigated, indicating the immense concentration of lead in topsoils due to the binding capacity of Pb to soil organic matter as well Fe and Mn oxides (Table 3.30). Taking the low pH value of the site presented into account, however, it has to be expected that Pb movement downward may occur in future. Man-made substrates deposited onto the soil surface can create a danger potential for groundwater and human health in future with reference to pedogenetic processes (see Section 6.3) and particularly the heavy metal content reaches large quantities, as given in this example. Moreover, antimony and copper that are released together with lead by corrosion of the bullets may show comparable transport mechanisms (Knechtenhofer et al. 2003).

### *3.5.2 Accident Sites*

In human history there have been some spectacular accidents contaminating environmental media including soil to a great extent and these must be kept in mind. In 1976 in a chemical manufacturing plant in Seveso with 20,000 inhabitants located close to Milan (Italy) an enormous exposure of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and other Polychlorinated Dibenzodioxins and Dibenzofurans occurred. The company produced Tri-chloro-phenol used as a basis for synthesis of herbicides. An anomalous pressure was responsible for an exothermic reaction of the Tri-chloro-phenol, resulting in the breakdown of the chemical reactor and production of unwanted TCDD at 250°C, which was released into the atmosphere. Six tonnes of toxic material were distributed by the explosion over approximately 18 $km^2$. Dependent on the wind direction a linear path in a south-east direction with a length of about 7 km occurred. In topsoils in the area close to the disaster site (zone A) with 736 residents more than 50 µg $m^{-2}$ TCDD were accumulated and in a second zone B around with 4,700 residents between 5 and 50 µg $m^{-2}$ were measured. The boundaries of the zones followed the boundaries of the different townships and hence did not represent the real extension of the TCDD deposit. TCDD is an artificial extremely toxic compound indicating values below 0.04 μg $m^{-2}$ in soils ubiquitously influenced only. Children and adults had to be hospitalised because they were suffering from skin lesions and chloracne and countless animals had to be slaughtered and burnt. Later on a number of illnesses were observed. To avoid a continuous threat to human health the soil was excavated and disposed of in a legal manner. Otherwise a biological breakdown would have needed a long period of time. TCDD concentrations beyond 20 cm in depth were not detected because contaminant mobility was low and subsequently leaching was hardly observed (Ratti and Belli 2009).

Fortunately, the Seveso disaster led to an immediate response action. In less developed and developing countries such direct reaction did not occur in a number of soil contaminating accidents. In Bhopal (India) with a population of 2,100,000 a terrible industrial disaster happened in 1984, causing the death of about 16,000 people after a plant producing pesticide released poisonous gases which flooded into the city of Bhopal. Soil and groundwater well analyses showed very high concentrations of some heavy metals like mercury as well as volatile organic chlorines such as chloroform and chlorinated benzene. Twenty-one years after the tragedy the abandoned, fenced-in plant site remained heavily contaminated and regular clean-up measures did not occur. Remediation time would need 4 years and the cost would be more than $30 million. Stockpiled material, ruined buildings and surrounding soils, particularly in the upper 60 cm, are continuously polluted. Open evaporation ponds, metal boxes with pesticide residues and dumpsites enable direct contact with children playing and livestock grazing (India Together 2004).

Spectacular accidents can result in widespread topsoil contamination. Apart from the $^{137}Cs$ fallout from nuclear weapon tests in the 1950s until 1970s, in Central Europe the Chernobyl disaster in 1986 must be designated the primary source of contamination with radioactive substances. Without doubt the accumulations of the radioactive substances are normally not combined with industrial emissions or deposits of technogenic mixtures. In Chernobyl (Ukraine) two explosions caused by mistakes in the handling of the reactor occurred and the ensuing fire was extinguished within 5 h but it resulted in emission of uranium dioxide together with radioactive nuclides such as $^{137}Cs$, $^{134}Cs$, $^{131}I$, $^{132}I$ and $^{140}Te$ into the atmosphere. In Chernobyl only $^{137}Cs$ with a half-life-time of 30.1 years was emitted in contrast to $^{134}Cs$ (half-life-time 2.2 years), which derived from both bomb fallout and the Chernobyl accident. Radioactive material were spread all over Europe and carried by wind over very long distances. Countries that received high amounts of radioactive fallout were in the first instance Ukraine, particularly the evacuation zone, where 114,000 people had to be evacuated, the European part of Russia and Belarus. These countries received more than 40 kBq $m^{-2}$ in a total area between 38,000 and 60,000 $km^2$. Secondly, Austria, Finland, Norway and Sweden received more than 40 kBq $m^{-2}$ in a total area ranging from 11,000 to 24,000 $km^2$. In other countries such as the Czech Republic, Germany, Greece, Italy, Poland, Romania, Switzerland and United Kingdom the area receiving more than 40 kBq $m^{-2}$ was reduced to less than 1,300 $km^2$ each (Kvasnikova et al. 2000; Ratti and Belli 2009).

Consequently, the atmospheric deposition mainly highlighted the contamination level in soils. It has been found that the highest concentrations were located in the upper part of the soil, because the mobility of caesium is supposed to be low. Thus, it was possible to find the highest values in humus layers of forested areas, while underneath the values often were below detection limits. Especially, if the humus layers were thick, the migration below ground usually seemed to be extraordinarily low. Investigations in the southern part of Germany revealed that the $^{137}Cs$ percentage of the humus layers ranged between 44.3% and 91.1%, the following 10 cm downward covered the residual caesium content almost completely (Kruse-Irmer and Giani 2003).

However, a certain soil-to-plant transfer took place. The average $^{134}Cs$ transfer coefficient plant (grasses species of the forested area)/soil reached values of 13–45 related to the humus layers and 3–28 related to peaty soils, while $^{137}Cs$ indicated much lower coefficients <8. The results demonstrated the high bioavailability of radioactive caesium in humus layers and peaty soils even decades after deposition. Obviously, adsorption and storage in the edaphon combined with the low leachate rate are responsible for the bioavailability. Radiocaesium seemed to be quickly mobilized and available for a short cut element cycle (Kruse-Irmer and Giani 2003).

Besides, $^{137}Cs$ migration also occurred in a different way. In Penza (Russia), a city with 530,000 inhabitants and 1,100 km from Chernobyl, the variations of caesium were observed 10 years after the disaster with reference to a hilly landscape, where soil erosion has to be taken into consideration. A flat area without erosion had values of 5.2 kBq $m^{-2}$, an adjacent area influenced by erosion 7.4 kBq $m^{-2}$ and an accumulation zone down slope even 10.4 kBq $m^{-2}$. The results reflected the horizontal transport of contaminated soil material. In the vertical direction it has been found that in unploughed soils more than 70% of $^{137}Cs$ concentration is restricted to the upper 5 cm and more than 90% to the upper 20 cm (Kvasnikova et al. 2000).

The public pays a lot of attention to the disasters described above. But there is another side to urban soils contaminated by accidents. This consists of more simple cases in which traffic and industrial processes accidents happen almost daily. Truck and railway crashes, leakages from long-distance pipelines, ship accidents and mistakes during storage and processing operations in industry plants may cause release of toxic substances, which can frequently only be discovered to a small extent. A few countries publish statistics about these occurrences. In Germany the statistics showed the soil contamination potential due to the high percentage of released liquids (Table 3.31) (Federal Statistical Office of Germany 2008).

### 3.5.3 Deposits and Fills

Deposited soils can be subdivided into distinct kinds of deposits depending on their derivation (Fig. 3.19). In the landscape, a deposit consisting of mining heaps, waste landfills, dumps and linear structures such as dikes, dams and sound-insolating earthen walls are easily recognisable. In Fig. 3.20a–c some examples of deposited sites are shown. The fill materials, however, result from excavations or refilling of former holes. Consequently, they do not have to be recognisably similar to the materials in their natural source areas. Fills are associated with former usually moist natural depressions

**Table 3.31** Registered accidents with hazardous substances in Germany in 2005 (Data from Federal Statistical Office of Germany 2008)

| | Total accidents | Released volume ($m^3$) | Not recovered volume ($m^3$) | Not recovered volume (%) |
|---|---|---|---|---|
| *Handling accidents* | | | | |
| Storage plants | 545 | 6,242 | 3,368 | 54 |
| Trans-shipment | 80 | 46 | 21 | 47 |
| Production and treatment plants | 76 | 803 | 248 | 31 |
| Miscellaneous | 89 | 1,672 | 1,577 | 94 |
| *Transport accidents* | | | | |
| Vehicles | 1,302 | 534 | 167 | 31 |
| Railway | 21 | 8 | 2 | 25 |
| Shipping | 49 | 78 | 62 | 80 |
| Long-distance pipelines | 5 | 129 | 129 | 100 |

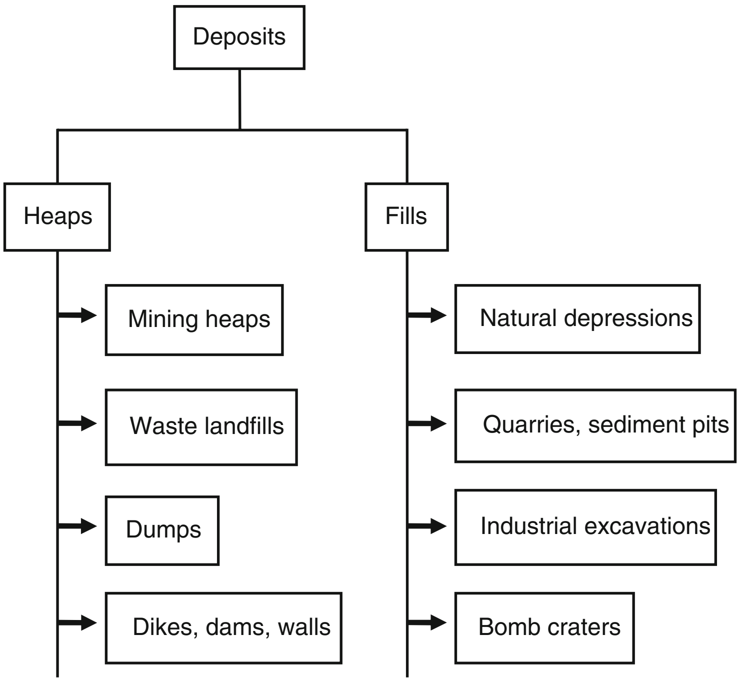



**Fig. 3.19** Classification of deposits

(swamp, peat land), quarries (sandstone, limestone etc.), and pits (sand, clay, loam, marble) leading to a general rising of the ground level (Nathanail and Bardos 2004). In urban areas problematic fills very often remain untouched since clean-up measures and rehabilitation processes are too expensive. The landscape appears as abandoned, derelict land with a brownfield character for a long period of time (Fig. 3.21).

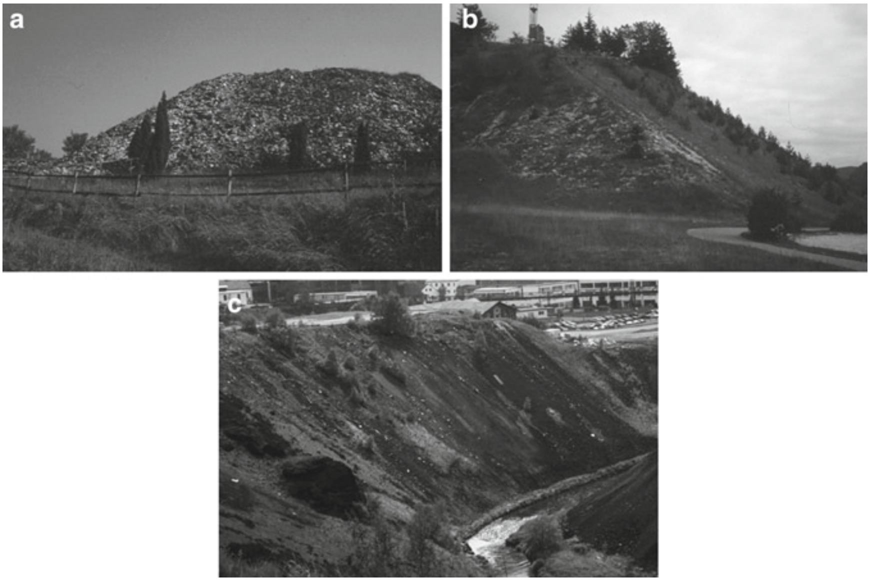



**Fig. 3.20** (**a**) Example for an urban deposit: open dumpsite consisting of household and industrial waste causing wind erosion to adjacent areas nearby Imperia, Italy. (**b**) Example for an urban deposit: unvegetated heavy metal contaminated ore mining heap indicating soil erosion in Bad Bleiberg, Austria. (**c**) Example for an urban deposit: unvegetated and highly contaminated deposits of waste from lead works in proximity of Leipzig, Germany, featuring erosion into a river in the valley below

**Fig. 3.21** Brownfield in Panipat, India (*left*), soil profile of the site (*right*) consisting of waste overlaid by a soil-waste mixture and a sandy layer; the soil properties exacerbate a future use of the built-up area

Occasionally, they are located in industrial areas, where excavation has taken place, or they are the results of warfare such as bomb craters. During the Balkan War in 1999 strategically important targets were destroyed but collateral damage (e.g. treatment facilities for waste and wastewater, fertilizer plants) was also of importance. Regardless of the nature of the targets, a release of hazardous substances into the environment, particularly into the soil, had to be expected. Furthermore, attention must be paid to unexploded bombs and landmines, indicating a serious redevelopment problem. In agricultural land a rapid refill of bomb craters with distinct and partly unknown materials had to take place in order to grow staple food crops. A single 500 pound explosive bomb can cause a bomb crater up to 14 m wide and 9 m deep. No-one knows the number of bomb craters in European countries resulting from the Second World War, but most of the craters were refilled rapidly without any knowledge about the materials that were used. For instance, it is supposed that in the Indochina conflict 25 million craters were caused by air raids (Genske 2003).

Bomb craters are not only results of historical occurrences. Unfortunately, current conflicts will produce new craters that will be refilled with debris from bombed buildings in order to level the surface and to initiate redevelopment. For instance, during the writing of this book the Gaza War took place, destroying large areas of Gaza City (Palestine) with a population of 680,000.

With regard to contamination potential the material to be deposited is mainly of interest. In principle, a distinction is made between construction debris, slag, ash, garbage, mining waste and sludge (see Section 4.1).

With reference to mining areas overground deposits consisting of mining waste are very common (Fig. 3.22). Such heaps were built for nearly all kinds of natural resource extraction by mining shafts, e.g. hard coal, iron, zinc, copper and gold. Lignite coal heaps were rarely deposited, because lignite coal extraction usually takes place as open-cast mining. Salt mining heaps may exist, but due to the better ratio between mining waste and extracted resource (about 0.5:1) the generation of heaps was limited. In the past, until the 1930s, barren cones with

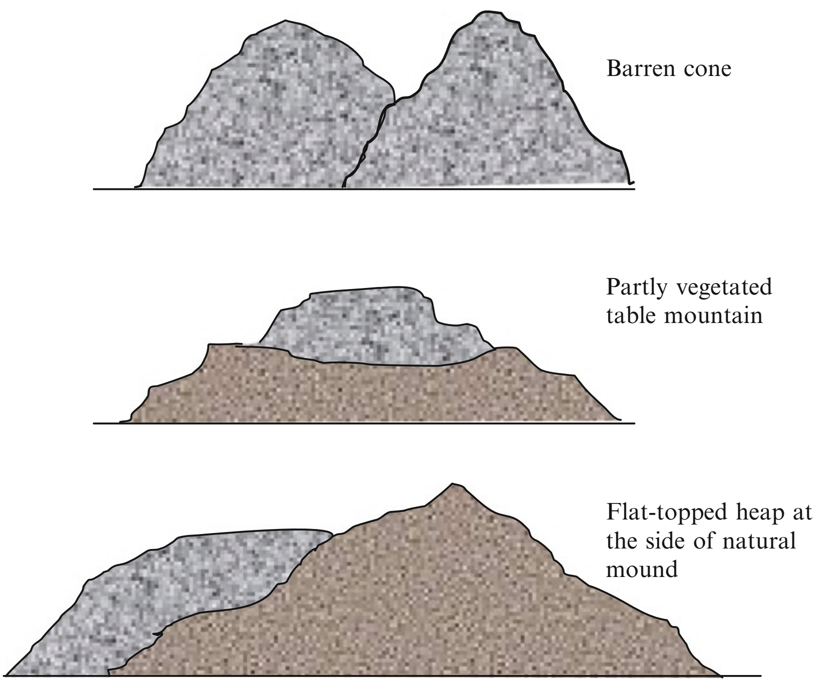
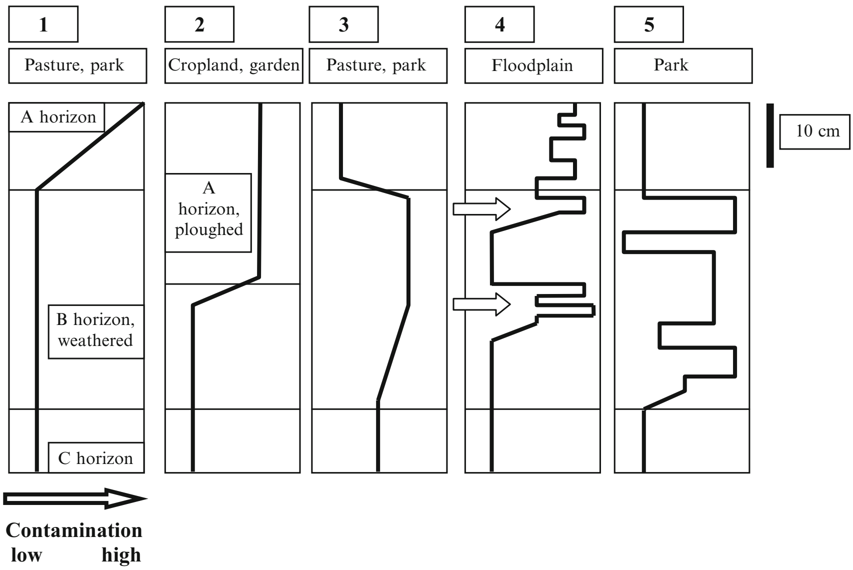



**Fig. 3.22** Types of mining heaps

steep slope gradients and a lack of vegetation provoking accelerated erosion potential were deposited. Afterwards, a new heap generation called a table mountain with a flat top and at least partly vegetated was established. In naturally mountainous areas flat-topped heaps were deposited at the sides of the natural mountains. In developed countries since the 1970s there have been increasing tendencies to imitate natural heap shapes to make them blend in with the landscape design of the region (Genske 2003).

## 3.6 Identification of Soil Contamination

The contaminant sources influence the depth gradient recognizable in soil profiles (Fig. 3.23). If the source is caused by atmospheric input (dust deposition), fertilizing, sewage sludge and pesticide application, within the soil the depth gradient will reveal high values at the top and will decrease with depth. In soils without ploughing and digging activities the contaminant accumulation is limited to the humic topsoil of about 10 or 15 cm. In the case of cropland and garden, where intensive soil cultivation happens,




1: Reason: atmospheric input
Impact: accumulation in humic topsoil

2: Reason: atmospheric input
Impact: accumulation in humic topsoil, dispersion by plowing

3: Reason: parent material (e.g. ore containing geology)
Impact: high values in C horizon (geologically determined), enhanced values in (the upper part of) weathered B horizon

4: Reason: flood
Impact: different relatively thin layers of alluvial deposits depending on industrial periods of time (example: two periods, 1st in 0-20 cm, 2nd in 40-50 cm depth, see arrows)

5: Reason: technogenic materials in subsoil
Impact: completely different contamination depending on the kind of material

**Fig. 3.23** Soil contamination profiles caused by different sources the borderline of potentially enhanced values is deeper, traditionally about 30 cm. Because of mixing processes the contaminant accumulation, however, is lower within the ploughed horizon compared with topsoils of pasture and forest. In the presence of geologically caused contaminated bedrock and parent material, higher values are detectable in the subsoil, particularly in the weathered subsoil horizons below the topsoil. Accordingly, the lithogenically determined concentration is altered by eluviations, illuviation, chemical and physical weathering.

Floods will give different fingerprints on contamination within the soil profile. Alluvial floodplains soils are characterized by long-term sedimentation, hence different thin sedimentation layers will be visible in the long run. Since each layer is based on different ecological and historical circumstances, e.g. duration and extent of the flood, time-related contamination level of the catchments, etc. (see Section 3.3.3), the contamination may vary within the soil profile. While principle tendencies are observable in relation to the contaminant sources mentioned, in the case of derelict land, especially deposited soils, a clear depth gradient can hardly be recognized. The type and extent of technogenic substrate deposited will decide on contamination values within the soil profile. Accordingly, heterogeneity in the horizontal direction should generally be expected (see Section 4.3).

During soil surveys it is possible to get a first impression of contamination by identification of some morphological features listed in Table 3.32. It is difficult to

**Table 3.32** Morphological features in soil profiles, contamination pattern and land-use type related to different causes of soil contamination

| Cause of contamination | Morphological features | Contamination pattern | Land-use type (examples) |
|---|---|---|---|
| Dust deposition | none | Decrease with increasing soil depth | Lawn, pasture, urban forest |
| Flood | Alluvial soil characteristics, sedimented layers | Indifferent | Floodplain |
| Application of mineral fertilizer, stone dust and leather flour | none | Accumulation in topsoil, parameters reduced to Br, Cd, Cr, Cu, Ni, V | Cropland, garden, allotment |
| Application of compost | Artefacts, humus accumulation in topsoil | Accumulation in topsoil, mainly heavy metals | Woodland, flower-bed |
| Application of ashes | Visible ash residues (wide C/N ratio) | Accumulation in topsoil, heavy metals and PAH | Garden, allotment |
| Application of sewage sludge | Perceiving odour | Decrease with increasing soil depth | Cropland, garden |
| Sewage irrigation field | Redoximorphological features, artefacts | Decrease with increasing soil depth (tendency) | Pasture surrounded by dams |
| Deposits | Presence of technogenic substrates | Indifferent | Potentially each land-use type |

recognize deposited dust, since particulate matter normally reveals size diameters below 10 μm. Contaminated soils influenced by floods have to show the typical characteristics of alluvial soils, particularly oxidation – reduction features. Soils contaminated by long-term application of problematical organic manure indicate enhanced organic matter content in the topsoil simultaneously. After sewage sludge application odour of fouling gas and ammonia can develop. Sewage irrigation fields are easily recognizable because of surrounding dam construction (see Section 5.4.5). In addition to the land-use type present and some simple analytical results, e.g. wide C/N ratio in soils fertilized with ashes over a long period, the identification of the contaminant source (and the polluter) appears to be feasible in a large number of contaminated soils investigated.

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# Chapter 4
# Man-Made Substrates

**Abstract** One of the main sources for soil contamination, particularly in the subsoils, is associated with the presence of technogenic substrates such as construction rubble, slag, ash, household and mining waste and sludges. Their technical origin is explained and their distribution in urban soils is calculated. Relevant chemical (e.g. pH value and C content), physical (e.g. porosity) and biological (e.g. biological activity) properties of the substrates mentioned are introduced. In more detail, the contamination potential including heavy metals and organic pollutants is discussed and standardized. In general, soil scientists are not very familiar with the recognition of technogenic substrates. Hence, a key to identify technogenic materials during field work is presented.

**Keywords** Construction debris • Contamination potential • Slag and ash • Substrate identification • Technogenic material • Waste

## 4.1 Origin

Substrates formed or at least altered by man are termed man-made or artificial or technogenic substrates. In Table 4.1 the substrates that are frequently present in urban soils are listed. The main component groups include construction debris, slag, ash, garbage, mining waste and sludge.

### *4.1.1 Construction Debris and Construction and Demolition Waste*

The presence of construction rubble in urban soils results from the demolition of different residential and industrial buildings. The rubble stems from civil demolition or is caused by war. It consists mainly of brick, mortar, concrete and plaster. Wall and roof bricks derived from loamy and clayey parent material are

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**Table 4.1** Classification of technogenic substrates

| Main component group | Origin | Component (examples) |
|---|---|---|
| Construction rubble | Housing and industrial construction | Brick, mortar, concrete, plaster |
| | Asphalt | Bitumen asphalt, tar asphalt |
| Slag | Ironworks | Blast furnace slag, sand and pumice from ironworks |
| | Steelworks | Slag of steel smelting furnace |
| | Heavy metal works | Lead slag, copper slag, zinc slag |
| | Foundries | Slag and sand of foundry |
| Ashes | Hard coal fired-power station | Fly ash, bottom ash |
| | Lignite coal-fired power station | Fly ash, bottom ash |
| | Garbage incinerator | Fly ash, bottom ash |
| Mining waste | Coal mining | Coal gangue, coke, briquette residues |
| | Ore mining | Ore mining waste |
| | Salt mining | Salt mining waste |
| Waste | Household refuse deposits | Glass, metal, paper, plastic, ceramic, organic garbage, wood, bulky refuse |
| | Industrial refuse deposits | Dross, cinder, other specific waste of industrial processes |
| Sludge | Sewage works | Sewage sludge |
| | River, lake and harbour dredging | Dredge spoil |

burnt materials with a visible red to yellow colour and a high percentage of micropores enhancing the water-holding capacity. Plaster due to natural gypsum ($CaSO_4*2H_2O$) indicates a yellowish colour and low stability in soils because of relatively high water solubility. It tends to form a solution, causing corrosion of concrete structures present in foundations. Contrary to mortar and concrete, plaster does not contain aggregates such as natural materials (sand, gravel) and technogenic materials (blast furnace slag, fly ash).

The greyish mortar and concrete fragments in urban soils are originally mixtures of cement, water and aggregates. Concrete may reveal a higher density and weathering resistance and is occasionally combined with construction steel. In the construction industry coal-fired power station ash is often recycled, because it is useful as a concrete ingredient or constituent in building blocks. In this way ash residues transfer into building walls and this may release their constituents after demolition later on. Apart from some problematic aggregates, the cement products may contain asbestos and other mineral fibers (Hiller and Meuser 1998). When soils are moist the danger of asbestos fibers being inhaled is relatively low but in dry conditions the hazardous impact increases considerably (Bridges 1991).

Black-coloured asphalt components result either from oil distillation or from tar production in coking plants. In order to influence the viscosity both raw materials are sometimes mixed (tar-bitumen mixture). In general, the main difference between oil-based and tar-based asphalt is the strong naphthalene odor of the tar products. The pieces of asphalt consist mostly of mixtures between the bitumen/tar and natural aggregates like sand, limestone, dolomite stone, basalt, etc. as well as technogenic aggregates such as bottom ash, fly ash, recycled construction rubble and blast furnace slag (Hiller and Meuser 1998).

Construction debris has to be separated from construction and demolition waste including road construction waste. Apart from the inert materials brick, ceramic, concrete, gypsum, tiles as well as tarred products including asphalt, additional materials are of concern associated with construction operations such as wood, glass, plastic, metals and their alloys, insulation material, soil and dredging spoil. The latter materials are more waste materials than construction debris, and, in Europe for example, consequently listed in the waste catalogues used (Commission Decision 2000). Most of the materials are inert and bulky. Furthermore, construction waste contains soil derived from excavation and leveling operations conducted by bulldozers. The waste is frequently used within the construction industry. In this way waste material from building demolition is re-used for road and car park construction or, even worse, in the context of landscaping. If the waste contains hazardous materials such as asbestos, lead-based paints, coatings, resins, fiber insulation, tar, etc., a wide dispersion cannot be excluded, as observed several times in less developed countries (Williams 2005).

The generation of construction and demolition waste depends apparently on the population size of the city. In Wisconsin, USA in towns smaller than 10,000 inhabitants this produced 0.14 kg day$^{-1}$ capita$^{-1}$, whilst in cities with a population of more than 10,000 people the value was 0.63 kg day$^{-1}$ capita$^{-1}$ (Williams 2005).

### 4.1.2 Slag and Ashes

The terms slag and ash should be separated from each other. Slag is always the result of an ore smelting process, ash results from a combustion process in power stations and garbage incinerators.

Blast furnace slag is generated during iron smelting processes with the aid of calcareous aggregates. Depending on the cooling down conditions, crystalline, grey blast furnace slag is created during slow production processes. Faster cooling duration in the presence of water produces blast furnace sand and in the presence of added fresh air porous blast furnace pumice. Waste products of steelworks are called steelworks slag and are based on reduced iron ore and scrap metal and produced with the aid of calcareous aggregates as well. This slag indicates a crystalline, dense structure with less pores and a very high specific gravity (>3.5 g cm$^{-3}$) (Hiller and Meuser 1998).

Metal works slag results from smelting treatment of distinct heavy metals such as copper, lead and zinc. The kinds of process engineering are comparable with the processes mentioned before leading to a dense slag structure with only some pores and different colours, for example reddish copper slag, dark blue lead slag or dark grey zinc slag (Hiller and Meuser 1998).

Waste products of coal-fired power stations are bottom ash, that can be differentiated at low temperature (1,200–1,500°C), ash consisting of gravelly texture and porous components, and at high temperature (1,400–1,700°C), ash with a sandy to gravelly texture and glassy surfaces derived from quick cooling down processes in the presence of water. Furthermore, there is silty fly ash due to the filter technique and pieces of cinder and dross created during the technical inspection of the equipment in power station facilities. In the latter the boilers are raked out continuously, leading to the production of cinders and dross. The types of ash may be produced independently of the coal origin, irrespective of whether it is anthracite coal, hard coal or lignite coal. Moreover, there are additionally ashes from garbage incinerators, namely bottom and fly ash. Most of the ashes reveal grey colours, ranging from light grey (hard coal) to brownish grey (lignite coal). Garbage incinerator ash, however, is well recognizable, since glass and metal residues are always combined with the ash substrate (Hiller and Meuser 1998).

### *4.1.3 Mining Waste*

On the one hand mining waste is of natural origin (e.g. sandstone, siltstone, shale, coal fragments) but on the other hand the material is considerably altered by man and not only existent in open-cast mines. The material is lifted, washed and broken into small pieces in the line with the coal preparation technique and piled featuring pyrite oxidation due to oxygen contact, which would not occur without seam extraction. The latter leads to changes in structure and colour (red, burnt coal mining waste) (see Section 6.3). Thus, this main component group may belong to the technogenic materials. Moreover, colliery wastes are used in urban environments as fill for road construction and other engineering projects. So, there is a close connection between urban soils and mining waste application. Consequently, it is acceptable to discuss the mining waste in the context of urban soils.

Coal mining waste includes the dark grey to black coal gangue usually deposited in mining heaps but also the light grey, washed coal mining waste produced during the coal preparation, as well as the red-coloured burnt coal gangue due to pyrite oxidation and subsequently coal fire occurrences inside the mining heaps. Furthermore, the mining waste contains a high percentage of coal itself. Other waste products related to the coal extraction are termed silvery to black coloured porous coke created in coking plants and black coal dust or briquette residues (Hiller and Meuser 1998).

Mining waste is continuously produced in ore mining areas and salt mining areas, too. In the case of ore mining waste the substrates indicate different forms and colours with reference to the parent material used for ore extraction. In contrast, the salt mining waste deposited in open-cast heaps has a whitish-pink colour and consists of loose, sandy material.

### 4.1.4 Municipal Solid Waste

The main component group waste includes municipal solid waste (MSW) containing single components such as plastic, glass, ceramics, textiles, metal and wood, as well as materials that are composed of different single components (e.g. bulky refuse). Moreover, organic and potentially biodegradable substrates including paper, garden and kitchen waste belong to the waste group. Additionally, problematical toxic waste like domestic fuel ash, batteries, colourant residues and mineral fibers are of importance (Hiller and Meuser 1998).

There are numerous plastics present in the household waste of today. Mainly low-density polyethylene (e.g. bags), high-density polyethylene (e.g. bottles), polypropylene (e.g. household packets) and polyethylene terephtalate (e.g. drink bottles) are used. Moreover, some more problematical plastics containing chlorinated compounds like polyvinyl chloride (e.g. flooring, cable insulation) are components of household waste (Williams 2005).

The quality of waste generated and landfilled has changed in the course of time. In New York (USA) with a population of 23,200,000 waste of different periods was collected and analyzed (Table 4.2). In the early part of the twentieth century ash from coal combustion for heating and cooking was dominant. This decreased after the establishment of gas fuels in the city. Paper increased enormously due to the use of packaging and the growth of advertising. In the second half of the century plastic was widely used, revealing an increasing tendency up to now. Non-returnable glass bottles and cans were substituted by plastic ones. Otherwise, recycling technology led to a decrease of some components like glass. Accordingly, the age of waste deposits should be taken into account during sampling and assessment of landfills or open dumpsites (Williams 2005).

**Table 4.2** Household waste composition from 1905 to 1989 for New York, USA (Data from Williams 2005)

| | 1905 (%) | 1939 (%) | 1971 (%) | 1978 (%) | 1989 (%) |
|---|---|---|---|---|---|
| Food refuse | 13.4 | 17.0 | 15.6 | 17.8 | 14.1 |
| Ash | 79.9 | 43.0 | 2.8 | 1.5 | 2.3 |
| Paper | 5.0 | 21.9 | 35.5 | 32.9 | 34.7 |
| Plastic | – | – | 2.7 | 8.8 | 9.9 |
| Metal | 0.2 | 6.8 | 11.1 | 13.3 | 5.3 |
| Glass | 0.2 | 5.5 | 23.1[a] | 9.4[b] | 5.5 |
| Textiles | 1.0 | – | 3.9 | 5.7 | 5.2 |
| Wood | 0.1 | 2.6 | 1.2 | 4.5 | 2.4 |
| Yard refuse | – | – | 0.7 | 4.7 | 4.7 |
| Miscellaneous | 0.1 | 3.2 | 3.5 | 1.5 | 15.8 |

[a]Includes ceramics and stones
[b]Reported as combined glass and ceramics

### 4.1.5 Sludges

While each main component group listed above indicates relatively dry conditions, the last one is called sludge involving sewage sludge of wastewater treatment facilities, dredge spoil resulting from the sludge dredging operations in lakes, rivers and harbours and industrial sludges. Typically, the wet sludge is well-known for reductive properties, a black, sapropel-like colour and a conspicuous ammonia or hydrogen sulphide odor.

The texture is usually fine-grained, particularly silt. In the case of sludge derived from ore mining operations the material has been crushed into smaller pieces before extraction operations (solution mining). Afterwards the silty material is pumped into lagoons containing residual amounts of toxic substances. In a similar way to the transportation of the silty fly ash by pipelines into lagoons, the material is quickly able to dry at the surface by means of solar evaporation, causing dust development that may seriously impact neighbouring sites (Bridges 1991).

### 4.1.6 Cleaned-up Substrates

Furthermore, new types of technogenic substrates were re-filled with increasing tendency in urban environments, in particular in the northern hemisphere, where more and more contaminated sites were cleaned-up or at least rehabilitated. New, man-made substrates were used for these purposes, so-called clean materials derived from soil washing facilities, thermal treatment, and biological regeneration pits (Burghardt 1994).

The scrub process in soil washing facilities leads to a separation of the texture classes. Gravel, sand and in some modernized plants the coarse silt are separated by sieve cascades, flotation and hydrocyclones. They are clean, since the adsorption potential for contaminants dependent on the limited specific surface of the single grains is rather low. In contrast, the fine fractions accumulate the contaminants so that the highly polluted residual sludge has to be disposed of or to remediate by another technique after drying in filter presses. The clean portion cannot be classified as soil material, because fine fractions, the organic matter and the edaphon have been removed but the material re-use is generally feasible for construction purposes. The possibility to re-use the material seems to become more problematical, if acids are applied in order to clean-up heavy metal polluted material or if detergents are applied to reduce the concentration of organic pollutants such as benzene, toluene, xylene and the Polycyclic Aromatic Hydrocarbons (PAH).

Material resulting from thermal treatment reveals clear physical, biological and chemical alteration. The low temperature rotary kilns cause complete removal of the living organisms, moderately enhanced pH values, losses of nitrogen, sulfur and phosphorus, a black colour and the development of technogenic soot-like carbon. In the case of high temperature treatment exceeding 650°C the material indicates additionally destroyed clay minerals and a completely changed soil structure. Moreover, the colour appears yellow to red and the pH value can reach values up to 12, in particular in the presence of alkaline technogenic materials such as construction debris residues. The re-use of material treated in rotary kilns is problematical because of the difficulties of compaction. If the material is deposited for rehabilitation purposes, the biological activity has to be renewed by compost and fertilizer application and the use of legumes. Because of the high boiling points of most metals it is not possible for thermal treatment to lower the heavy metal concentrations. Accordingly, the backfill opportunities of the material remain continuously restricted.

Alternatively, the soil clean-up technology provides the biological treatment carried out in regeneration pits. In particular, Total Petroleum Hydrocarbons and BTEX aromates (benzene, toluene, ethyl benzene, xylene) are suitable for successful remediation. After the treatment period the material leads continuously to a high biological activity. Furthermore, some biochemical properties have changed with reference to the improved life conditions of the bacteria. In particular, organic amendments are added such as wood chips, sawdust, compost, bark mulch, straw and peat. In addition, nutrient and air capacity are enhanced and the pH value might indicate a neutral reaction in most cases. It should be noted that contaminants which are not capable of biological degradation such as heavy metals remain untouched.

In conclusion, in the cities of developed countries where the percentage of areas backfilled with cleaned-up soil material is increasing, this type of man-made soil will surely become more common in future.

## 4.2 Characteristics

### *4.2.1 Recognition During Field Work*

It is not very difficult to recognize the different components introduced in Section 4.1. With a key most of the components in dry conditions can be identified (see Appendix).

### *4.2.2 Chemical Properties*

Chemically, clear differences between the technogenic substrates can be measured. Firstly, the pH value ranges from acid to very alkaline (Fig. 4.1). Most of the substrates may reveal neutral to alkaline pH values, resulting predominantly from the presence of calcium carbonate, which, for instance, generally exists in construction rubble. The process of weathering of construction rubble comprising calcium containing substrates such as cement, concrete and plaster produces a permanent release of calcium and subsequently raises the pH value (Craul 1992).

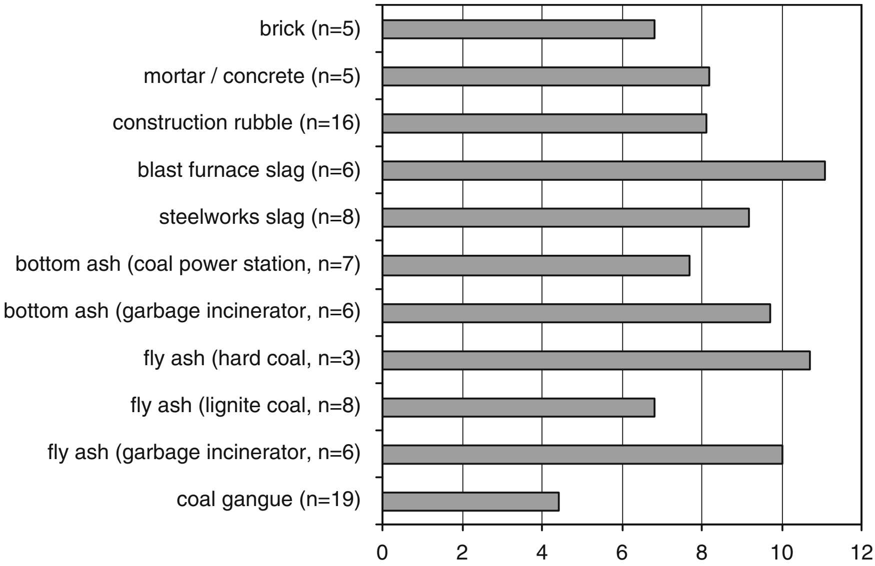


**Fig. 4.1** pH($CaCl_2$) values of technogenic mono substrates, sampled in urban soils of the Ruhr area, Germany (mean values) (Data from Hiller and Meuser 1998)

Moreover, the often calcareous refuse input in garbage incinerators in combination with high processing temperatures leads to an increase in the bottom and fly ash pH value. In the case of slag calcium is added during the metallurgy procedures in ironworks, steelworks and heavy metal works for temperature regulation purposes. In contrast to the above-mentioned technogenic substrates, the pH value of the pyrite containing coal mining waste is usually below 4.0, tending to strong acidification (Hiller and Meuser 1998).

The pH values of technogenic substrates are similar to other chemical properties independent of the area or country where they have been produced and deposited, while the decisive influence is derived from the production process only. Since the procedure leading to the creation of technogenic substrates normally follows the same schedule, the chemical properties of technogenic substrates including the contamination potential are comparable worldwide. For instance, pH mean values of the inert residues foundry sand (9.6), combustion ash (8.6) and steelworks slag (12.1) analyzed in Santiago de Compostela (Spain) with 100,000 inhabitants reached exactly the same level as detected in the Ruhr area (Germany) with a population of 5,300,000 (Camps Arbestain et al. 2005; Hiller and Meuser 1998). According to the results, the pH values of dumped fly ash from coal-fired power stations varied from 8.4 to 8.8 in Panipat (India) with a population of 330,000 (Singh et al. 2002).

The technogenic substrates have relatively high values with regard to the parameter electrical conductivity. If values of 1.0 dS $m^{-1}$ are exceeded, plant growth of ornamental shrubs in parks will be restricted and leaf chlorosis can become visible. Table 4.3 supplies information about the data analysed in the German Ruhr area. Already some

**Table 4.3** Electrical Conductivity EC (dS $m^{-1}$) of technogenic mono substrates, sampled in urban soils of the Ruhr area, Germany (Data from Hiller and Meuser 1998)

| Technogenic substrate | Mean value | Range |
|---|---|---|
| Construction rubble (n = 5) | **0.78** | 0.1–2.3 |
| Slag (n = 11) | **0.43** | 0.1–0.5 |
| Ashes, coal-fired (n = 5) | **0.32** | <0.1–0.7 |
| Coal mining waste (n = 5) | **0.76** | 0.2–0.9 |
| Dredge spoil (n = 6) | **0.40** | 0.3–0.5 |

values tend to exceed this value and further substrate components may indicate still higher values, e.g. fly ash from coal-fired power stations (1.1–4.0 dS $m^{-1}$), bottom ash from garbage incinerators (0.6–2.2 dS $m^{-1}$) and fly ash from garbage incinerators (10.5–20.2 dS $m^{-1}$) respectively. The highest value measured refers to salt slag derived from aluminium works (approx. 120 dS $m^{-1}$) (Hiller and Meuser 1998).

The carbon analysis in soils consisting of technogenic substrates proves to be problematic. Apart from the humus ($TOC_{humus}$) and the inorganic carbon TIC due to the presence of calcium carbonate, an additional carbon source can be detected called technogenic carbon ($TOC_{tech}$). The last one appears in soils with a high percentage of coal, soot and ash, enhancing the Total Carbon (TC) content considerably (see Section 6.2.3). This fraction can be separated by using $H_2O_2$, since this agent is able to remove the humus completely, while carbon based upon hard coal and ash will not be destroyed.

The analysis method was applicable in relation to hard coal and its combustion residues. However, it was not possible to treat lignite coal and peat containing soils in this way, since the degree of coalification is much lower. The average degree of coalification is 60% for peat, 70% for lignite coal, 80% for hard coal and 95% for anthracite coal. The estimation of $TOC_{tech}$ follows the equation

$$TOC_{tech} = TC - TIC - TOC_{humus} \tag{4.1}$$

It should be noted that $TOC_{humus}$ can be detected by carbon analysis with and without hydrogen peroxide addition. Table 4.4 shows the high percentage of $TOC_{tech}$ for some technogenic substrates and typical horizons of urban soils. In the presence of coal or coal-based ingredients almost the Total Carbon (TC) value was covered by $TOC_{tech}$. In contrast, humic materials such as the humic topsoil and the dredge spoil consisting of organic lake or stream sediments revealed both carbon components, $TOC_{humus}$ and technogenic carbon, resulting, for instance, from dust deposition in urban environments (Makowsky and Meuser 2007).

In principle, it can be expected that the cations determining the cation exchange capacity CEC of technogenic substrates relate mainly to calcium, magnesium and potassium, since the alkaline technogenic substrates are predominant sources of bases. Coal gangue may indicate the only substrate initiating acid soil conditions, and consequently adsorption of protons and aluminium. The CEC was analyzed in

**Table 4.4** Total Carbon (TC), Total Inorganic Carbon (TIC), Total Organic Carbon – humus ($TOC_{humus}$) and Total Organic Carbon – technogenic ($TOC_{tech}$) of technogenic monosubstrates and deposited soils, sampled in urban soils of the Ruhr area, Germany (%) (Data from Makowsky and Meuser 2007)

| | TC | TIC | $TOC_{humus}$ | $TOC_{tech}$ |
|---|---|---|---|---|
| Monosubstrate | | | | |
| Hard coal | 95.7 | <0.1 | 1.0 | 94.7 |
| Bottom ash | 4.5 | <0.1 | nd | 4.5 |
| Coke | 73.7 | <0.1 | nd | 75.7 |
| Briquette | 52.9 | <0.1 | 2.6 | 50.3 |
| Soil horizon | | | | |
| Humic topsoil | 1.9 | <0.1 | 1.7 | 0.2 |
| Ash-soil mixture (subsoil) | 15.4 | 0.6 | nd | 14.8 |
| Coal gangue-soil mixture (subsoil) | 23.7 | <0.1 | 2.0 | 21.7 |
| Construction rubble-soil mixture (subsoil) | 1.6 | 0.7 | nd | 0.9 |
| Dredge spoil (sediment) | 28.1 | <0.1 | 10.9 | 17.2 |

nd = not detectable

**Table 4.5** Cation Exchange Capacity CEC ($cmol_c\ kg^{-1}$) of deposited soil horizons consisting of technogenic monosubstrates sampled in different cities in Germany (Data from Blume and Schleuss 1997)

| Substrate | Horizons investigated | Arithmetic mean | Range |
|---|---|---|---|
| Construction debris | 26 | **9.1** | 2–25 |
| Ashes | 19 | **36.9** | 5–142 |
| Slag | 5 | **47.4** | 14–111 |
| Coal gangue | 8 | **16.4** | 7–32 |
| Refuse (waste) | 22 | **20.6** | 3–67 |
| Sludges | 11 | **47.6** | 30–76 |

soil horizons of deposited monosubstrates in different locations throughout Germany (Table 4.5). In accordance with the representative results it can be concluded that ashes, slag and sludges enable cation adsorption that is even able to exceed natural soils, while construction rubble, coal gangue as well as household waste appeared to indicate reduced CEC values. Apart from the slag, there seemed to be a context to the particle size distribution, because technogenic substrates with high CEC tended to have a silty to clayey texture, in contrast to construction rubble, coal gangue and waste exhibiting coarser texture classes (sand, gravel, stones). Another reason may be linked to the carbon and organic matter content, increasing CEC for ashes and (organic) sludges (Blume and Schleuss 1997).

**Table 4.6** Ranges of hydraulic conductivity (m $h^{-1}$) and bulk density (g $cm^{-3}$) of deposited technogenic substrates (Data from Meuser and Blume 2004)

| Substrate | Horizons investigated | Hydraulic conductivity | Bulk density |
|---|---|---|---|
| Construction debris | 20 | 9.6–57.6 | 1.1–1.7 |
| Slag | 8 | 180–360 | 0.5–1.3 |
| Bottom ash | 18 | 88.8–456 | 0.7–0.8 |
| Coal mining waste | 10 | 12.0–528 | 1.2–1.4 |
| Household waste | 9 | 264–1,728 | 0.5–1.2 |

### *4.2.3 Physical Properties*

Physically, most of the technogenic substrates belong to the coarser texture classes sand, gravel and stone. Silty to clayey texture is only related to fly ash and sludge and to some technogenic substrates such as bricks after long-term weathering processes. Deposited technogenic substrates contain a lot of pores, enhancing the hydraulic conductivity and decreasing the bulk density (Table 4.6). Pores covering the substrate surface visibly are responsible for high hydraulic conductivity with reference to construction debris (brick, mortar and, to a lesser extent, concrete), nearly all kinds of slag and bottom ash of garbage incinerators and coal-fired power stations. The increased conductivity of mining waste is based on voids and cracks created by rapid weathering processes after depositing. In refuse containing deposits the bulky components are ultimately responsible, because between them air-filled spaces are present that are able to reach diameters of a centimeter or even a metre (see Sections 6.1.3 and 6.1.4). Since the bulk density depends on the relation between weight and volume, in accordance with the pores, voids and cracks mentioned above, bulk density is sometimes in a range typical for peaty soils. In turn, the specific gravity of technogenic components did not vary from natural substrates, e.g. brick 2.72 g $cm^{-3}$ and blast furnace slag 2.17 g $cm^{-3}$, respectively. Only some slag like the lead slag (up to 4.0 g $cm^{-3}$) may reach very high values (Meuser and Blume 2004).

### *4.2.4 Biological Properties*

Biologically, pure deposits of technogenic substrates induce relatively low microbiological activity. The microbial biomass in the upper 30 cm of deposited soils revealed very low values at the beginning. With increasing age and simultaneous development of humic horizons (see Sections 6.2.3 and 6.3) the biomass results in higher values comparable with natural soils (Table 4.7) (Meuser and Blume 2004).

Some topsoils consisting of technogenic monosubstrates deposited in the district of Osnabrück, Germany, with a population of 330,000 inhabitants, were analyzed in relation to the enzyme activities (Table 4.8). Compared to the cultivated garden soil (Anthrosol), the potential nitrification of coal mining waste, sand from a sandstone washing facility, ash and household waste were clearly lower, but metal works slag and

**Table 4.7** Standardized microbial biomass (kg ha$^{-1}$) in 0–30 cm of deposited soils consisting of technogenic monosubstrates (Data from Machulla et al. 2001, cited in Meuser and Blume 2004)

| Substrate | Horizons investigated | Fresh substrate deposit | Deposit with initial humic horizon | Deposit with humic topsoil |
|---|---|---|---|---|
| Construction debris | 27 | | | |
| Debris <30% | | 200–400 | 400–800 | 800–1,600 |
| Debris >30% | | <200 | 200–400 | 400–800 |
| Slag, ashes | 12 | <200 | 200–400 | 400–800 |
| Coal mining waste | 30 | <200 | 200–400 | 400–800 |
| Refuse (waste), sewage sludge | 5 | 400–800 | 800–1,600 | 1,600–3,200 |

**Table 4.8** Enzyme activity in different anthropogenic soils at a depth of 0–30 cm in the Osnabrück district, Germany (Data from Gromes 2006)

| | Potential nitrification | Urease | Saccharase | Acid phosphatase |
|---|---|---|---|---|
| Soil | ng N g$^{-1}$ 5 h$^{-1}$ | μg N g$^{-1}$ 2 h$^{-1}$ | μg glucose g$^{-1}$ 3 h$^{-1}$ | μg nitrophenol g$^{-1}$ h$^{-1}$ |
| Garden soil | 450 | 28 | 365 | 332 |
| Coal mining waste heap | 70 | 45 | 330 | 600 |
| Sand after washing | 8 | 3 | 55 | 26 |
| Metal works slag heap | 4,100 | 55 | 290 | 601 |
| Ore mining waste heap | 2,700 | 96 | 433 | 768 |
| Ash deposit | 370 | 13 | 165 | 202 |
| Landfill soil | 75 | 77 | 1,040 | 611 |

ore mining waste showed rather high results. It was possible to detect relatively high values for the wooded slag and ore mining heap with reference to urease and saccharase activity as well. Apparently, the household waste site indicated considerable values of saccharase exceeding the cultivated soil. The acid phosphatase reacted indifferently. High values were found for the coal mining waste site in acid conditions (pH 3–4) but also for the slag heap, where the pH value was higher than 12 (Gromes 2006).

## 4.3 Contamination Potential

### *4.3.1 Texture Influence*

There is no doubt about the influence of technogenic substrates on soil contamination. How much the contamination is going to increase, if technogenic additions take part in soil stratigraphy, is hard to assess. It depends on quality and percentage of the single substrate components and particularly the soil texture. Because soil sampling procedures are usually reduced to the fine fractions (clay, silt, sand) and the gravelly and stony parts of the soil sample are removed before chemical analysis, the

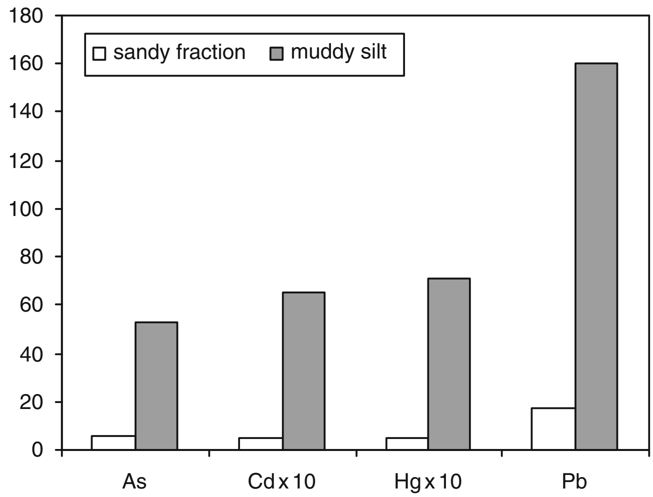


**Fig. 4.2** Heavy metal concentrations (mg $kg^{-1}$) of harbour-dredged sludges divided into different texture classes in Hamburg, Germany (Data from Meuser and van de Graaff 2010). Metal extraction method: aqua regia

contamination seems to be lower than the percentage of technogenic substrates would suppose. Moreover, in sandy soils reduced values are to be expected, because there is a thinning impact on the analysis results. In general, the following explanations related to heavy metals supply information about total concentration (aqua regia extraction) according to the contamination potential only.

In general, the texture class influences the contamination, since fine earth is able to adsorb more cations (e.g. Cd, Pb, Zn) than the coarse fractions. Therefore, some technogenic substrates consisting of silt or clay may be more polluted. For instance, harbour sludge fields in Hamburg, Germany, with 1,800,000 inhabitants show different heavy metal values depending on the texture class (Fig. 4.2) (Meuser and van de Graaff 2010).

The texture influence, however, can be modified with reference to solid technogenic substrates. Consequently, the resistance to weathering may influence the contamination level of the fine earth. In Münster (Germany) with 270,000 inhabitants city center subsoils which were partially affected by industry were investigated, distinguishing between the course and the fine fraction. In the case of lead usually combined with the presence of construction rubble the fine earth indicates higher results than the skeleton, since the material is already weathered. In contrast to this, the nickel values are much higher in the skeleton fraction compared with the fine earth due to the kind of technogenic substrate, mainly weathering-resistant steelworks slag (Fig. 4.3).

To neglect the coarse fraction in analyses is generally problematic in urban soils. The fraction can consist of both natural and technogenic substrates with different capacities to release contaminants like heavy metals. Results from a research project in Marrakech (Morocco) with a population of 1,200,000 showed that the fine fraction <2 mm mainly contributes to the contamination, particularly to the water-extractable percentage of metals. On the other hand raised concentrations have been found

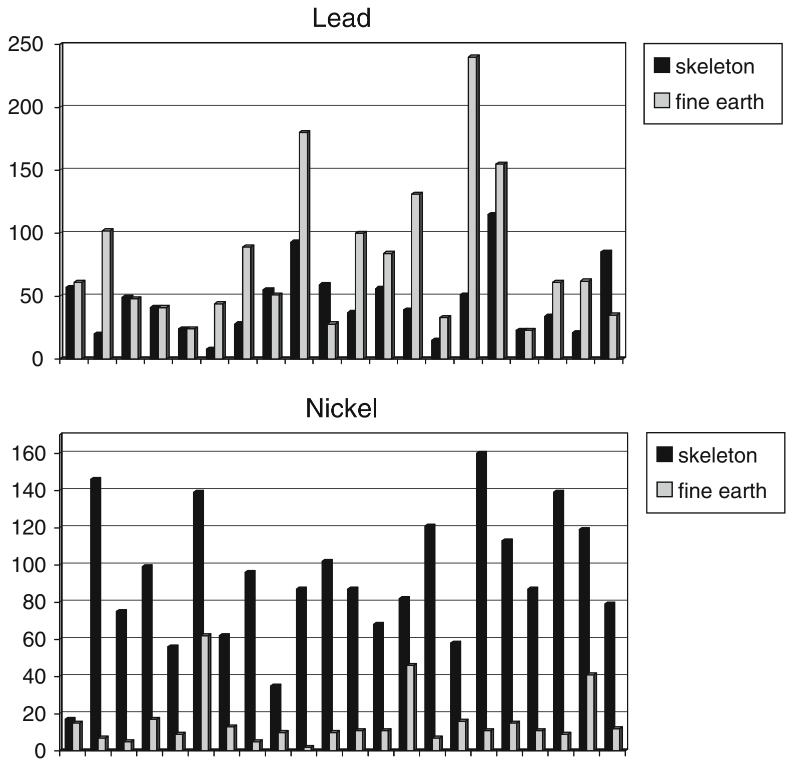


**Fig. 4.3** Pb (20 samples) and Ni (22 samples) content (mg $kg^{-1}$) of skeleton and fine earth in slag and ash containing urban soils (depth 0–100 cm) in Münster, Germany

related to the coarse fraction of waste components such as bones, wood, plastic, fabric and paper, respectively. For instance, the water-extractable percentage of the coarse fraction (Cd + Cu + Ni + Zn) reached 33.4% for bones, 14.1% for wood, 11.8% for fabric and paper, and finally 8.2% for plastics. The authors concluded that composition and heterogeneity of urban soils may lead to modifications of contaminant mobility depending upon the technogenic fractions present. Just as the natural material technogenic substrates are affected by weathering processes, influencing the availability of metallic pollutants in the long run (El Khalil et al. 2008).

### *4.3.2 Substrate Differences*

In Table 4.9 technogenic substrates and their general contamination potential are summarized (Meuser and van de Graaff 2010). Table 4.10 presents the values which can normally be expected when urban soils consist of more than 15% technogenic substrates.

**Table 4.9** Expected contamination of urban soils containing a high percentage of technogenic substrates

| Main component group | Contamination potential |
|---|---|
| Construction rubble | • Mixtures generally more highly contaminated than individual components<br>• Problematical components: concrete based on fly ashes (Pb), asbestos containing debris (fibres), tubes containing debris (Cu, Pb), debris influenced by fire damage (PAH)<br>• Tar asphalt generally more highly contaminated than bitumen asphalt (PAH, phenol) |
| Slag | • High heavy metal concentrations in metal works slag<br>• High Cr and Ni concentrations in steelworks slag<br>• Low heavy metal values of foundry sand, blast furnace slag, sand and pumice of ironworks |
| Ashes | • Fly ash generally more highly contaminated than bottom ash<br>• Garbage incinerator ash generally more highly contaminated than ash from coal power stations<br>• Problematical components: dross, bottom ash (PAH) |
| Mining waste | • Heavy metal contamination (ore mining waste)<br>• Rdioactive emission (uranium mining waste)<br>• Strong acidity (hard coal mining waste)<br>• Salt leaching (salt mining waste) |
| Refuse (waste) | • Problematical components: some plastics (heavy metals), wood (pesticides), organic garbage (methane formation)<br>• Generally indifferent contamination<br>• Occasionally high levels of industrial components |

PAH = polycyclic aromatic hydrocarbon

Components of construction rubble demonstrate relatively low concentrations in relation to the analyzed parameters. The only exception refers to tar asphalt which indicates a high Polycyclic Aromatic Hydrocarbon (PAH) level. While slag from ironworks can be described as uncontaminated, other slag types such as steelworks slag reveal very high heavy metal concentrations. In particular, chromium and nickel should be mentioned with reference to the scrap metal input, which contains alloy metals and heavy metal slag that are enormously contaminated with different metals, namely arsenic, cadmium, chromium, copper, nickel, lead, zinc.

It should be concluded that materials in contact with high temperatures may present low mercury, PAH and PCB (Polychlorinated Biphenyls) concentrations because of volatilization. On the other hand, incomplete burning processes initiate PAH development. In the capital of Germany, Berlin, with 3,400,000 inhabitants, soils containing debris from building demolition and debris from bombing of the Second World War were compared. The $PAH_{EPA}$ level of the debris from civil demolition ranged from 0.2 to 13.7 mg $kg^{-1}$ and of the war influenced debris from 0.2 to 643.6 mg $kg^{-1}$. It was possible to detect similar results with reference to benzo(a) pyrene, a five-ring PAH (civil demolition rubble: <0.1–0.5 mg $kg^{-1}$, debris from bombing: <0.1–33.3 mg $kg^{-1}$) (Mekiffer et al. 2000).

**Table 4.10** Contaminant concentrations (mg $kg^{-1}$, %) of different technogenic mono-substrates present in urban soils of different locations in Germany (Data from different sources, e.g. Hiller and Meuser 1998)

| Substrate | n | As | Cd | Cr | Cu | Hg | Ni | Pb | Zn | Bap | ΣPCB |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Brick | 12 | 10–25 | 0.5–2 | 20–50 | 10–70 | <0.3 | 10–30 | 40–110 | 20–60 | < 0.8 | 0.1–0.4 |
| Mortar/concrete | 8 | <20 | <0.2 | 10–50 | a | <0.1 | 10–20 | 10–40 | a | < 0.1 | < 0.1 |
| Asphalt (tar) | 29 | 10–20 | 1–1.5 | 10–80 | a | a | a | 80–100 | a | 30–150 | < 0.1 |
| Blast furnace slag | 5 | <10 | <0.6 | 10–50 | <30 | <0.1 | <10 | 10–40 | 10–140 | < 0.1 | < 0.1 |
| Steelworks slag | 11 | <30 | <0.7 | 90–4.5% | 20–180 | <0.2 | 20–250 | 10–170 | 130–410 | < 0.3 | < 0.1 |
| Lead slag | a | 1,000–7,000 | 5–20 | 1,000–4,000 | 1,000–4,000 | <0.1 | 100–1,500 | 2–5% | 7–13 % | a | a |
| Copper slag | a | 300–1,000 | 5–20 | 200–2,000 | 7,000–12,000 | <0.1 | 1,000–2,000 | 2,000–4,000 | 5–10 % | a | a |
| Bottom ash (coal-fired) | 12 | 2–25 | 0.1–1.7 | 10–60 | 90–110 | <0.1 | 30–80 | 10–180 | 10–230 | < 0.8 | < 0.1 |
| Fly ash (garbage incinerator) | 9 | 10–230 | 270–380 | 160–3,600 | 630–1,300 | 7.5–9 | 100–1,300 | 5,500–8,300 | 1.5–2.7 % | a | a |
| Bottom ash (garbage incinerator) | 11 | 2–90 | 2–77 | 40–1,300 | 510–3,400 | <2.0 | 40–280 | 410–3,200 | 840–4,800 | a | a |
| Coal gangue | 71 | 3–70 | 0.2–1.2 | 10–50 | 50–80 | <0.3 | 20–90 | 20–180 | 130–330 | < 0.5 | a |

Bap = benzo(a)pyrene

[a]Data not available

Metal extraction method: aqua regia

The products of garbage incinerators have comparably high values as well, especially the fly ash substrate. It is caused by the potentially polluted waste materials that have been burnt and the residual accumulation of metals with a high boiling point. It should be mentioned that 200–350 kg bottom ash is produced per ton of incinerated waste. Additionally, between 20 and 40 kg fly ash $kg^{-1}$ is created. Consequently, it can be assumed that a lot of the ashes are disposed of and are used for construction purposes (see Section 6.1.1).

In fresh municipal solid waste the cadmium concentration varied in Leeds (United Kingdom) with a population of 440,000 between 0.4 and 1.8 mg $kg^{-1}$. In German garbage incinerator bottom ash values ranging between 2 and 77 mg $kg^{-1}$ and in garbage incinerator fly ash ranging between 270 and 380 mg $kg^{-1}$ were measured. The accumulation process was confirmed taking other elements with very high boiling points like Pb into consideration. In waste the concentration was 33–247 mg Pb $kg^{-1}$, in bottom ash 410–3,200 mg Pb $kg^{-1}$ and in fly ash 5,500–8,300 mg Pb $kg^{-1}$ (Hiller and Meuser 1998; Williams 2005). The partitioning between bottom ash, fly ash and gas depends on the physico-chemical properties of the elements. Volatile mercury and cadmium with high vapour pressure and a low boiling point showed a high percentage of gaseous losses. Elements with a medium boiling point such as lead and zinc tended to be retained in both bottom ash and electrostatic precipitator dust. Ultimately, elements like copper and iron with high boiling points are completely concentrated in bottom ash. Because of volatilization the concentration of most organic pollutants is relatively low in garbage incinerator ashes. Based on international studies the dioxin concentration was intensively analyzed. In bottom ash the Polychlorinated Dibenzodioxin (PCDD) concentration varied between 0.25 and 0.48 μg $kg^{-1}$ and the Polychlorinated Dibenzofuran (PCDF) concentration between 0.54 and 0.10 μg $kg^{-1}$. Considerably higher results were yielded in relation to the fly ash, ranging from 115 to 1,040 μg $kg^{-1}$ (PCDD) and from 48 to 280 μg $kg^{-1}$ (PCDF) (Williams 2005).

The coal-fired ashes do not seem to be highly contaminated, since the original material reveals relatively low contaminant concentrations. Consequently, the ash from coal-fired power stations is visibly less contaminated than the garbage incinerator ash. However, it should be taken into account that the heavy metal concentrations of coal are different depending on the geological origin. Accordingly, the heavy metal concentrations of ash products may differ enormously. Table 4.11 summarizes results from fly ash in completely different locations. The result from United Kindom reveals the metal variability derived from different geological strata. In this study additional values for cobalt (2–115 mg $kg^{-1}$), mercury (<0.1–0.6 mg $kg^{-1}$), selenium (4–162 mg $kg^{-1}$), tin (933–1,847 mg $kg^{-1}$), vanadium (292–1,329 mg $kg^{-1}$) and cyanides (below detection limit) were presented (Sear 2001).

Usually, investigations dealing with contaminated land consider a certain group of metals only. Therefore, the quality standards in different countries cover chosen parameters for contaminated land assessment purposes and neglect further trace elements. Typical examples of elements frequently neglected in standardized investigation processes are the metalloids beryllium and selenium as well as the heavy metals antimony, cobalt, tin and vanadium, respectively. In heavily industrialized

**Table 4.11** Heavy metal concentration (mg $kg^{-1}$) of fly ash from coal-fired power stations in different locations (Data based on own analyses (Bulgaria, China, and Germany) and Sear 2001)

| Location | Origin | Cd | Cr | Cu | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|
| Huainan, China (mean value) | One sludge field | 0.2 | 20 | 26 | 16 | 10 | 40 |
| Pernik, Bulgaria (mean value) | One sludge field | 0.2 | 22 | 92 | 44 | 14 | 47 |
| Tamil Nadu, India (range) | Untreated power plant samples | 3.8–4.8 | 35–44 | 20–30 | – | 3–11 | 40–230 |
| Ruhr area, Germany (range) | One sludge field | 0.7–14.0 | 70–200 | 90–360 | 40–430 | 140–1,100 | 230–1,800 |
| United Kingdom (range) | Untreated power plant samples | <0.1–4 | 97–192 | 119–474 | 108–583 | <1–976 | 148–918 |

Deposited sludge in the German example originates from power stations using hard coal, lignite coal and dredged sludge from canal maintenance; data based on own analyses
Metal extraction method: aqua regia (total concentration)

areas, however, these elements should be taken into account, since they may cause danger to ecological compartments and human health.

Research studies in the cities of Essen with 580,000 inhabitants and Oberhausen with 220,000 inhabitants (Germany), well-known for their coal mining and metal processing industry, revealed raised concentrations of the mentioned parameters caused by atmospheric deposition and presence of technogenic substrates within the soil profiles. Compared to the geogenic concentrations of sedimented materials such as loess and shale being present as parent material in the area in question, the elements tended towards increased values in association with humic topsoils influenced by long-term atmospheric deposition as well as with some techogenic materials, particularly metal works slag and coal-fired bottom ash. Results from metals concentrations of the technogenic substrates isolated and analyzed are shown in Table 4.12 (Meuser 1996).

Construction debris (gypsum), ashes, slag and coal mining waste are sulphur-enriched materials that undergo physico-chemical weathering processes in soils after depositing. Accordingly, below large-scale deposits containing the materials in question sulphate leaching is expected. In heavily industrialized territories like Essen located in the Ruhr area, Germany observation wells for drinking-water quality have revealed accumulated sulphate concentration in contrast to nitrate predominantly associated with agricultural use. In this way, a city–rural area gradient was obviously found. Whilst nitrate is mainly linked to agriculturally used areas, sulphate and chloride, the latter derived from the application of deicing salt, are typical anions percolating through anthropogenic and technogenic soils (Fig. 4.4).

**Table 4.12** Arithmetic means of some trace elements (mg kg$^{-1}$) in parent materials, topsoils and technogenic substrates in the cities of Essen and Oberhausen, Ruhr area, Germany (Data from Meuser 1996)

| | | Be | Co | Sb | Se | Sn | V |
|---|---|---|---|---|---|---|---|
| Parent materials | Shale | 3.0 | 19 | 0.3 | 0.5 | 2.5 | 130 |
| | Loess | uk | 9 | 0.6 | uk | 1.8 | 64 |
| Topsoils, 0–30 cm (n = 306) | | uk | 15 | 3.7 | uk | 18.7 | 51 |
| Construction debris (n = 6) | | 1.0 | 9 | 3.2 | nd | 4.8 | 75 |
| Blast furnace slag (n = 5) | | 3.5 | 6 | nd | 0.9 | 1.7 | 52 |
| Metal works slag (n = 4) | | 1.9 | 93 | 30.6 | 0.3 | 9.0 | 164 |
| Bottom ash (n = 5) | | 2.4 | 121 | 0.8 | 1.8 | 3.3 | 59 |
| Coal gangue (n = 5) | | 1.4 | 18 | 2.8 | nd | 5.8 | 65 |

nd = not detectable; uk = unknown
Metal extraction method: aqua regia

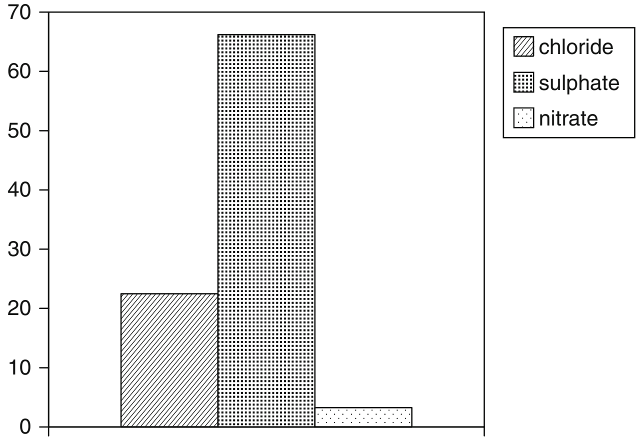


**Fig. 4.4** Percentage of groundwater samples (%) in observation wells exceeding in 1994 and 1995 German thresholds for drinking water quality (chloride: 120 mg L$^{-1}$, nitrate: 50 mg L$^{-1}$, sulphate: 240 mg L$^{-1}$) in Essen, Germany (n = 151) (Data unpublished)

Waste is the most heterogeneous group and consequently it is problematical to assess the contamination potential. The compositions of the single substrates differ considerably. Based on investigations of waste generated in the European Union, the average percentage in the first years of the current century amounted to 33% paper, 20% food and garden waste, 10% glass, 8% metals, 7% plastics, 4% textiles and 18% others. As observed in New York, USA, the household waste composition changed in the course of time (see Section 4.1.4).

In Table 4.13 ranges of heavy metals are listed, indicating a wide spectrum. Waste generated in Leeds, United Kingdom was analyzed in more detail, showing generally lower values in comparison with the results of other regions presented. The mean values appeared to be relatively low but some individual values were obviously raised. One important source of heavy metal contamination is surely the

**Table 4.13** Analysis from household waste samples in Leeds, United Kingdom and from approx. 35 waste samples in different American, Asian and European countries (Data from different sources, cited in Williams 2005)

| Element | Results from Leeds: Arithmetic mean | Results from Leeds: Range | General results: Range |
|---|---|---|---|
| Bromide (%) | **0.01** | 0.01–0.01 | – |
| Chloride (%) | **0.40** | 0.21–0.97 | 0.4 |
| Fluoride (%) | **0.01** | 0.01–0.01 | – |
| Antimony (mg $kg^{-1}$) | **2.4** | 0.68–3.7 | – |
| Arsenic (mg $kg^{-1}$) | **5.2** | 1.8–10.0 | 2–5 |
| Cadmium (mg $kg^{-1}$) | **0.8** | 0.4–1.8 | 1–150 |
| Chromium (mg $kg^{-1}$) | **42.5** | 24–61 | 40–200 |
| Cobalt (mg $kg^{-1}$) | **–** | – | 3–10 |
| Copper (mg $kg^{-1}$) | **64.0** | 41–100 | 200–700 |
| Lead (mg $kg^{-1}$) | **101.3** | 33–247 | 100–2,000 |
| Mercury (mg $kg^{-1}$) | **–** | – | 1–50 |
| Nickel (mg $kg^{-1}$) | **43.5** | 17–105 | 30–50 |
| Vanadium (mg $kg^{-1}$) | **–** | – | 4–11 |
| Zinc (mg $kg^{-1}$) | **225.8** | 149–313 | 400–1,400 |
| PCB (μg $kg^{-1}$) | **–** | – | 200–400 |
| PCDD/PCDF (μg $kg^{-1}$) | **–** | – | 0.050–0.150 |

**Table 4.14** Non-ferrous metals in the household waste (examples) (Data from Williams 2005)

| Metal | Examples |
|---|---|
| Copper | Electrical fittings and wire, plumbing fittings, kitchenware consisting of brass, screws, plated products, decorative waste |
| Nickel | Plating, cutlery, components of kitchen goods |
| Lead | Pipes, electric bulb contacts, wine bottle closures, lead-acid batteries, plumbing products, lead-based paints |
| Tin | Cans, containers, toys, kitchen ware, electrical contacts, solder, constituent of bronze |
| Zinc | Carbon-zinc batteries, components in die-castings, door fittings, domestic appliances, toys, domestic kitchen and garden equipment |

presence of non-ferrous metals (Table 4.14). In general, landfilled waste may cause problematic leachate leading to risk to the groundwater (see Section 5.4.2)

Waste contamination arises significantly in the presence of household hazardous waste including garden pesticides, paints, medicines, oils, batteries, solvents and other materials. The percentage is calculated to be between 1 and 5 kg household$^{-1}$ year$^{-1}$ and accordingly relatively low. The risk, however, is hard to assess, because no-one knows exactly human throw-away behavior (Williams 2005).

## 4.4 Distribution

Systematic soil survey in the urban environment covering different land-use types and representative large areas has been relatively seldom carried out. Thus, most of the scientific findings result from German soil inventories conducted in the last 2 decades with reference to the drawing up of so-called Soil Contamination Maps. In the meantime, the methodology of the maps has been applied to other countries like China (Beneke et al. 2007).

Considering the soil inventory of the German city Münster (Fig. 4.5), soils of the city centre as well as industrially and administratively used areas reveal a lot of deposited soils, while garden soils (hortisols) dominate the residential areas and, particularly, vacant allotments (see Section 5.3.2). Furthermore, deposited soils can be found in parks (refilling processes after the Second World War) and sports fields consisting of man-made, compacted soils to enable sports activities.

In Stuttgart (Germany) with a population of 600,000, 53% of the whole territory in the upper 100 cm of the systematically investigated soils revealed more or less technogenic ingredients (Table 4.15). Building debris, cinders and ashes in soils of the city centre, bricks, mortar and composting residues in old villages of the suburbs and ashes, cinders and other industry-specific waste in the older industrial areas were mentioned as technogenic substrates. The highest technogenic skeleton content was associated with block buildings and involved subsoils as well. In covered topsoils, partly derived from cropland sites and deposited after construction activities, the percentage of technogenic substrates was much lower, but the topsoils were enriched with organic substrates like compost. In residential areas consisting of

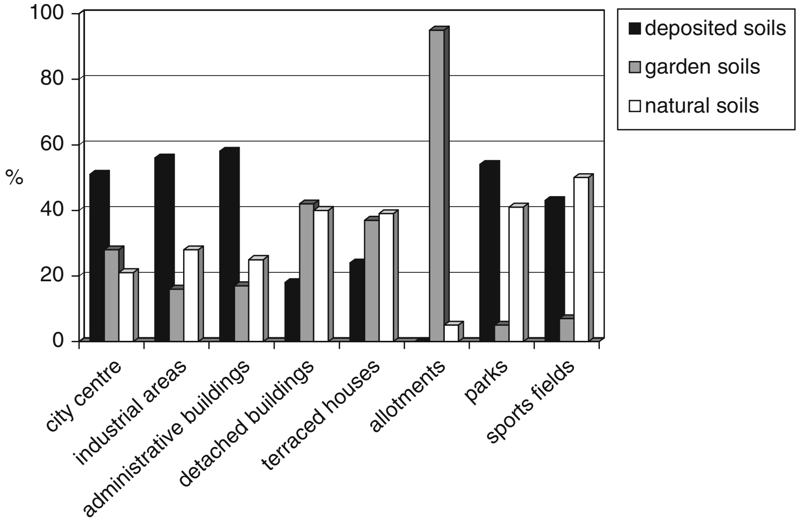


**Fig. 4.5** Percentage of deposited soils (Technosols), garden soils (Anthrosols) and natural soils according to the land-use type in Münster, Germany

**Table 4.15** Distribution of soils with and without anthropogenic influence in Stuttgart, Germany based on systematic soil investigation (Data from Stasch et al. 2000)

| Soil group | Percentage |
|---|---|
| Natural soils, unsealed | 42.0 |
| Anthropogenic soils consisting of natural material only, sealing <15% | 5.0 |
| Anthropogenic soils consisting of natural material and technogenic additives, sealing <15% | 4.5 |
| Anthropogenic soils consisting of natural material and technogenic additives, sealing 16–75% | 28.5 |
| Anthropogenic soils consisting of natural material and technogenic additives, sealing 76–100% | 12.5 |
| Anthropogenic soils predominantly consisting of technogenic substrates | 7.0 |

terraced houses and one-family-houses the percentage of technogenic mixtures increased with proximity to the buildings and decreased with soil depth. Nevertheless, the highest technogenic amounts related to the industrial soils (Stasch et al. 2000).

In Essen, Germany, soil samplings conducted between 1987 and 1996 and involving more than 900 sampling areas showed that 34% of the city soils are natural soils, while 66% must be classified as anthropogenic soils, divided into soils consisting of filled natural materials (11%), consisting of both natural and technogenic substrates (39%) and consisting of only technogenic substrates (16%) respectively (Hiller and Meuser 1998). Soil surveys of 36 parks in Hamburg, Germany, showed deposited soils, 26.5% with natural material and an additional 20.5% with a mixture of both natural and technogenic material (Däumling 2000).

The high percentage of anthropogenic soils appeared to be present not only in big cities and agglomeration areas. In Eckernförde, Germany, a small town numbering 20,000 inhabitants, covering 1,800 ha only and located on the Baltic Sea, an intensified soil survey with approximately 1,700 soil profiles was carried out. The city is influenced by harbour traffic and tourism and serves as an important home port for the navy. Anthropogenic soils consisting predominantly of deposits with natural material but also containing technogenic components like construction debris, waste and sludges covered about 46% of the whole territory (Blume and Schleuss 1997).

In Halle (Germany) with 230,000 inhabitants, where a soil survey was conducted in a comparable manner, the percentage of anthropogenic deposits that consist mainly of technogenic substrates amounted to 28.8%. A further 11.0% belonged to coal mining waste deposits, while only 36.9% of city areas were natural soils. Halle looks back on a long industrial history, particularly lignite coal extraction history. After the Second World War which caused considerable damage in the city area because of numerous bombings, the city expanded rapidly. In the outskirts new residential areas were built, especially between 1950 and 1970, when lignite coal was extracted in open-cast mines as well as by pit working. Consequently, the long-term anthropogenic impact on soils caused intensive disturbance by means of excavation, depositing and levelling and thus led to the high percentage of man-made soil profiles of today (Blume and Schleuss 1997).

This effect of the Second World War occurred also in Asian countries involved. Most of the green belts remaining in the capital of Japan, Tokyo, with 12,900,000 inhabitants developed in spite of the damages caused by the Second World War. Their role was later modified to prevent expansions of urban areas resulting from the economic growth. In recent years many parks have become catchments of urban garbage, waste soils and materials of construction residues. Urban park soils in western Tokyo originated from reclamation of volcanic ash soils and, particularly, from construction waste deposits (Takeda et al. 2005).

In the capital of the USA, Washington DC, with 580,000 inhabitants, 81% of the city area was termed disturbed land and 14% was classified as fills. Exact soil survey in the famous Central Park of New York, USA resulted in a percentage of man-made soils. 66.4% of the park soils were changed, the percentage of completely sealed pavement was 6.9%. Only about 10% of the territory consisted of natural materials, particularly rock outcrops (Craul 1992) (Fig. 4.6).

It has been reported that in the capital of Russia, Moscow, with 10,500,000 inhabitants, the soil composition contained about 14.4% of construction wastes. The results of a soil survey were that a low percentage of natural soils were present. Only parks and urban forest created islands consisting continuously of natural, undisturbed soils (Stroganova et al. 1998).

Industrial sites where the topsoil is usually stripped off mostly show more land-shaping procedures during the long industrial periods of time, since buildings were altered, re-decorated, even removed and re-built. Cut and fill operations cause a lot of profile modifications, leading to greater variability and site complexity (Craul 1992).

**Fig. 4.6** Man-made soils in the Central Park of New York, USA, interrupted by natural rock outcrops

Cut and fill operations can even occur with respect to measures which were originally environmentally protective. It might be an anachronism that the new construction of vernal pools compensating for destroyed natural biotopes during road and home construction may alter natural soils at other sites. The construction alters natural landscape and soil horizons radically. In the vicinity of Sacramento (USA) with a population of 1,400,000 changes in natural landscapes due to the establishment of new vernal pools in a line with compensation were investigated. On the top of undisturbed soil mixtures of different soil texture classes and broken technogenic fragments were deposited reaching thicknesses of up to 100 cm and being extremely variable in the horizontal and vertical direction. Previously, the soil was excavated to a depth of nearly 2 m. In conclusion, although the measure should be understood as environmentally protective, the soil surrounding the vernal pools was completely altered and must ultimately be designated as man-made soil (Cook and Whitney 2000).

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# Chapter 5
# Anthropogenic Soils

**Abstract** Anthropogenic soils are divided into Anthrosols and Technosols. The first group covers considerable areas in urban environments, in particular garden soils in residential areas. The most important properties of Anthrosols, including the nutrient capacity and the contamination potential, are presented. Furthermore, characteristics of plaggen soils located in the periphery of cities are mentioned. The main focus, however, is on the deposited, man-made soils usually consisting of technogenic substrates. Attention is given to dry deposits such as landfills and mining soils and to sludge fields associated with numerous human activities. Potential and actual mobility of contaminants in the soils of concern are introduced and discussed. Important features like reductomorphic conditions in landfill soils, acidification of coal mining heaps containing pyrite, failure of vegetation to establish itself in ore mining soils and dehydration of dredged harbour sludge fields are taken into consideration in the context of the contamination problem. Additionally, completely artificial urban soils such as planters and rooftop gardens are integrated. In order to give a detailed description of the properties of anthropogenic soils for most of the presented soils an example located in the German Osnabrück district is described in more detail.

**Keywords** Dredged harbour sludge • Garden soil • Municipal solid waste • Plaggen soil • Pyrite oxidation • Tailing pond • Technosol

## 5.1 Definitions

In general, the anthropogenic soils are termed Anthrosols that have been modified through human activity, especially the amendment of organic fertilizers and waste, intensive cultivation, etc. In the case of cultivation influence only the term Anthrosol is applicable, soils dominated by technical origin, however, will be called Technosols. These soils are usually deposited containing different artificial materials such as technogenic substrates (see Section 4) and they are soils which

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have been excavated, transported and backfilled somewhere. Moreover, Technosols can be sealed by technic hard rock like buildings and pavement. Subsequently, Technosols are mainly found in urban, industrial, traffic, mine and military areas (WRB 2006).

Anthrosols are the most likely soils to have been treated with high amounts of nutrients, especially P and N, from farmyard manures and fertilizers, as well as metalloids and metals such as As and Cu (from fungicides) and organic contaminants such as DDT and dieldrin (from pesticides). They are associated with long-term horticultural or agricultural use, indicating man-made profile changes. The soils, however, are based on natural parent material. Typically, the humic topsoil is deeper than the usually ploughed soil depth of 30 cm. We can distinguish between several kinds of soil within this group (Meuser and Blume 2004).

Anthropogenic – usually – deposited soils, called technosols (WRB 2006) consist of natural and/or technogenic, man-made material (deposols in some European nomenclatures). These soils are very likely to contain contaminants as they are often waste soils from excavations at previous urban sites and can contain almost anything, from rubble to mine waste and fly ash. Detailed information about classification is given in Section 6.3.

Because soils in the urban environment are usually stripped, filled, mixed, compacted and supplemented with artificial materials, soil profiles are enormously modified, leading to high spatial and vertical heterogeneity. The boundary between distinct horizons is characterized by abrupt alteration with reference to physico-chemical properties such as texture, structure, bulk density, aeration, hydraulic conductivity, colour, pH value and chemical compositions. Consequently, an urban soil exhibits a strong lithologic discontinuity within the profiles. A general feature is the deposit of humic topsoil in a line of levelling and rehabilitation operations, making the technogenic subsoil disappear. In the first instance the topsoil deposit seems to be necessary, if the land-use types park, sports fields, gardens, etc. are planned.

The deposit of humic topsoil is required, if the area is used as medium for plant growth, as we can observe in lawns (parks), gardens as well as rehabilitated land. Protection from erosion and runoff realized by drainage and improvement of structural stability must be included in the case of restored land and disposal areas.

A systematic survey in New York (USA) with its population of 23,000,000 resulted in a topsoil thickness of 15–25 cm and in street side soils of 6–35 cm. The topsoil was from the original soil or it was excavated from another site to avoid stockpiling. Below, the soils contained a high proportion of building rubble and waste produced during construction processes of buildings (foundations) and utility networks. Depending upon the future use the topsoil was amended with organic fertilizers like compost, especially in garden areas. In contrast to the topsoil getting periodic tillage, the backfilled subsoil displaced from its natural setting and transported exhibited visible structural damages and compaction (see Section 6.1.3) (Craul 1992).

## 5.2 Artificial Soils

Because of the planned use a lot of urban soils are artificially constructed. Drainage, for instance, is necessary in sports fields and disposal areas, leading to the construction of drainage layers below. A layer with a high load-bearing capacity must be integrated with regard to sports fields and footpaths. Such compacted layers in sports fields, however, may cause conflicts between the playability of the surface and the plant growth of sports fields consisting of a grass cover. Moreover, the drainage layer must be constructed as well in order to ensure the fast removal of water during precipitation occurrences. If the drainage layer fails and an intensive use (perhaps overuse) of the field takes place, the turf cover will be damaged and consequently the surface will become artificial plastic and slippery. In the end, sports activity is not possible in such detrimental conditions (Mullins 1991).

Sports fields make specialized construction necessary. Golf courses frequently lay out alongside rivers or low-lying areas in town districts due to beautiful scenery and availability of shallow undergroundwater for irrigation need to be drained. Thus, a sandy growth medium for turf grasses is favoured, and it is used especially for the so-called greens. Consequently, intensive construction activities must occur to make the drainage conditions perfect.

Another example deals with the bowling greens. As seen in South African bowling greens, good drainage had to be realized to ensure continuously dry playing surfaces, short turf grass cutting procedures, that are not feasible in wet conditions, and in order to avoid detrimental soil compaction. Soccer, hockey and rugby sports fields also needed optimal subsoil construction, particularly related to the drainage conditions. Therefore, deep excavation combined with refill and drainage operations led to intensive changes of soil profiles and the establishment of completely new, man-made soils. To avoid interruption of play during rainy periods and to avoid waterlogged areas additional drainage systems had to be installed and the compaction must be reduced as far as possible. On the other hand, the filling material was compacted well in order to exclude subsidence. In conclusion, sports fields are one of the most manipulated soils in urban environments (Van Deventer 2000a).

City centres are dominated by buildings and roads and show a lack of open green space. In streets utility lines are densely packed at a low depth below the paved area. Consequently, there are often restricted opportunities for roots to grow. To improve living conditions for the inhabitants artificial greenery is constructed and used as planter, container, podium and rooftop. In these niches good-quality plants are generally found in city areas. The plants, however, have to live in relatively bad conditions meaning restricted soil volume associated with moisture stress, detrimental nutrient supply and unusual soil temperature variations. Attention must be paid to size and dimension of containers and planters, the right choice of species grown and specifications of the soil material. An adequate soil volume should be chosen with respect to artificial, more or less constructed open green space that is planned to cover underground road tunnels, subway lines as well as underground car parks which are present in city centres to a large extent.

Moreover, planters are commonly used in pedestrian zones and sidewalks. Among other disadvantages, in containerised soils water and air cannot freely move horizontally (Craul 1992). In Fig. 5.1 the ecological problems associated with planter equipment are illustrated.

There is no doubt that an extreme kind of artificial soil can be found in rooftop garden soils. Grass-covered permeable rooftops decrease the amount of impermeable surfaces, decrease the pollutant load of runoff water and reduce the urban heat build-up. Some roof materials are responsible for contaminated roof runoff, e.g. bituminous materials (Polycyclic Aromatic Hydrocarbons PAH), roofs made of

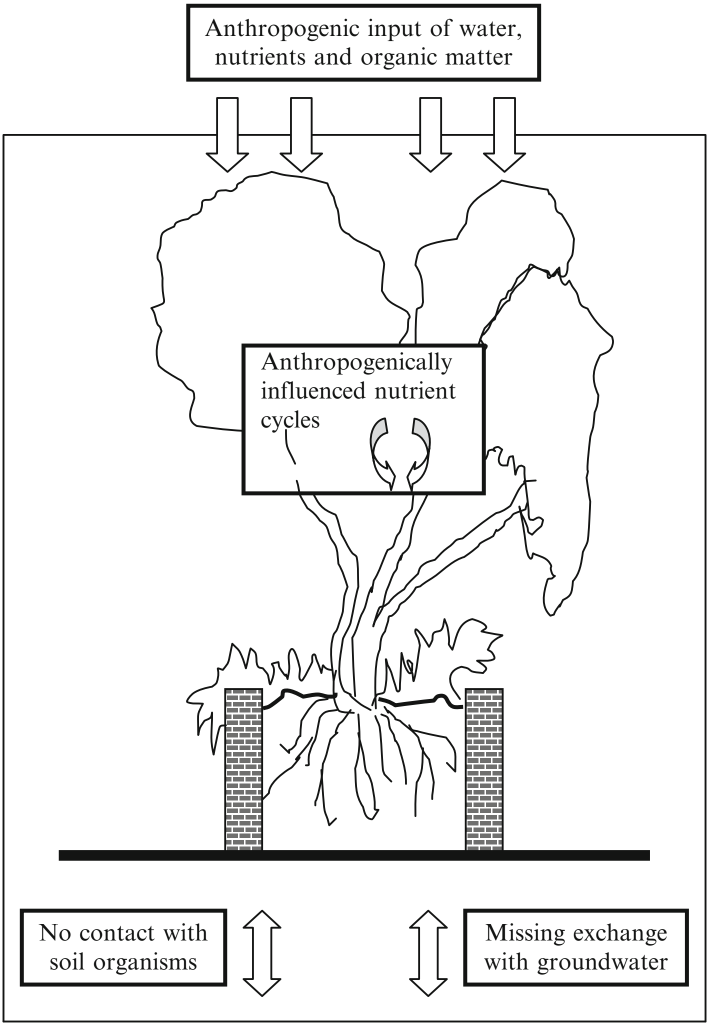


**Fig. 5.1** Planter construction and its ecological impacts (Modified from Pietsch and Kamieth 1991)

**Table 5.1** Principle construction of an extensive rooftop garden

| Level | Characteristics | Species/material |
|---|---|---|
| First level plants | Resistant to dryness, radiation, wind as well as frost minimal plant cover 60% | *Sedum*, *Dianthus*, sweet-swelling herbs (thyme, lavender), grasses (*Festuca, Bromus*), pine (*Pinus mugo*) |
| Second level substrate | High air capacity, high available field capacity, high CEC resistant to deflation, erosion | Mixture of rubble/pumice, bark mulch, compost |
| Third level drainage layer | Storage hollows (above), drainage system (below), steam diffusion openings | Polyethylene/polystyrene |
| Fourth level protection layer (above roof construction) | Impenetrable to roots, very thin | Polyethylene or bitumen |

CEC = cation exchange capacity

zinc and asbestos cement (Zn) and roofs made of pantiles (Cu). Rooftop gardens are energy-saving because they reduce radiation in wintertime and facilitate a pleasant indoor climate in hot summers. Furthermore, they adsorb dust, increase evaporation and transpiration and, finally, they are habitats for drought-resistant vegetation. The principle construction of extensive rooftop greenery can be seen in Table 5.1. Extensive types have a height of 8–22 cm and a weight of 70–150 kg $m^{-2}$, which is comparable with gravel-filled roofs ranging between 90 and 150 kg $m^{-2}$. They enable an optimized water capacity and can generally be constructed with a maximum allowable slope gradient of about 25°.

These soils were investigated in detail in Dallas (USA) with its 1,300,000 inhabitants. The rooftop materials consisted of mixtures with ingredients that are traditionally used in organic potting material. Expanded shale, for example, was a beneficial means of absorbing water and retaining nutrients, which are supplied on the basis of diffusion processes. Organic amendments had an additional positive influence on plant growth. The combination of both mineral materials and organic substrates enabled optimal plant growth of annual, perennial and even woody plants. The applied expanded shale reached a water holding capacity of 37.8%, a cation exchange capacity of 2.75 $cmol_c$ $kg^{-1}$, sufficient presence of cationic elements such as potassium, magnesium and calcium and ultimately a pH($H_2O$) value of 8.3. Unlike in gravel and sand, the living conditions for plants could be estimated as favourable (Sloan et al. 2000).

For rooftop green construction extensive plantation is usually favoured, since it does not require much after-care. There is a wide range of succulent species useful for roof plantation. In contrast, intensive roof plantation requires higher investments and is only feasible, if the loading capacity and stability of the building roof is guaranteed. It is sometimes used on the top of representative buildings of banks and global player companies as well as hospitals and old people's homes. On the other hand, the extensive kind must ensure good living conditions for the plants, irrespective of frost and wind resistance. In the capital of Hungary, Budapest, with a population of 1,700,000, different substrates were applied including fibrous and ground peat, compost, rice hull, wood bark, perlite and slag. The inorganic substrates exhibited sufficient water permeability and retention, good air conditions and stabilized the structure. The organic materials improved the structure as well. Their main advantage, however, lay in better nutrient supply and enhanced biological activity. In conclusion, a mixture of both seemed to be the best solution for plant establishment (Forro and Draskovits 2000).

The following Sections 5.3 and 5.4 introduce typical anthropogenic soils. Physical and chemical properties are considered. In addition to anthropogenic soils of other locations some soils from the Osnabrück district (Germany) with a population of 330,000 will be described in detail.

## 5.3 Cultivated Soils (Anthrosols)

### *5.3.1 Plaggen Soils*

Plaggen management existed for approximately 3,000 years and was found in Northern Germany, The Netherlands, Belgium, Ireland and Scotland. Sods (brown plaggen) from the neighbouring low-lying areas and heathland were principally used but forest plaggen was probably also employed. The plaggen was used for animal bedding in stables, and subsequently spread onto the fields together with composted farmyard manure. The plaggen management was replaced by mineral fertilization at the end of the nineteenth century. Fertility on the intensively cultivated soils improved significantly, while the areas where the removal of the plaggen took place deteriorated to a great extent. Even the nutrient depletion of the adjacent soils led to restrictions through the Public Authority in some places. Mineral matter was also spread with the organic material, resulting in the terrain surface becoming distinctly raised (Fig. 5.2). The thickness of the plaggen fill reached between 30 and 90 cm (Blume and Leinweber 2004).

Comparable human activities showing a lot of similarities to the Western European plaggen management have been described for other locations in the world. Composted peat with cattle excrements and organic manure were applied to improve soil fertility over a long period of time near St. Petersburg (Russia) with 4,500,000 inhabitants, leading to a humic topsoil of a thickness between about 30 and 50 cm. Even in cold climate conditions present in the Archangelsk Region, Russia the human land management reminds one of the plaggen cultivation, since farmers used peat mixed with stable dung and mineral additions in order to spread it onto fields later on. Subsequently, the soil characteristics like the high phosphorus content and the presence of artificial remnants are similar to the management results in Western Europe (Hubbe et al. 2007). Earth mixed with organic manure from stables was applied to loess plateau soils in Northwest China for thousands of years to form humus enriched topsoils of more than 1 m (Blume and Leinweber 2004).

**Fig. 5.2** Raised terrain of a plaggen soil grown with maize in the periphery of Osnabrück, Germany

In some regions in Europe large-scale plaggen management took place, for instance the sandy areas of The Netherlands were totally influenced by plaggen activity. In general, areas near villages were the predominantly used locations for that purpose. Because the areas nearby villages or small towns had to be used in relation to the problematical transport capacity of the heavy plaggen sods in the past, today we are able to find plaggen soils in the vicinity of urban areas, particularly on the periphery of urban agglomerations.

Most of the plaggen soils revealed a sandy texture characterized by a black to dark grey colour and a humus content of 1–8%, while the more seldom loamy and sandy-loamy plaggen horizons were dark to yellowish brown with a lower humus content of 1–3%. Decisive properties were the high phosphorus content and the presence of anthropogenic artifacts such as pieces of pottery and brick fragments (Blume and Leinweber 2004). The fill helped to preserve not only fossil soils (for instance podzolic soils out of sand), but also archaeological findings. Due to the spreading of organic material and subsequent soil cultivation, the fill and A horizon of the Podzol became mixed together; as a result, most of the archaeological findings were detected at the lower edge of the plaggen horizon. Some plaggen soils provided evidence of former agricultural use but after the Second World War several sites were reforested with coniferous trees (Blume and Leinweber 2004).

A characteristic feature of the presented soil profile (Table 5.2, Fig. 5.3) is the very thick plaggen horizon (60 cm), showing features of a ploughed A horizon in the surface soil, due to earlier long-term agricultural use. The soil is basically an anthropogenic deposited soil which, however, in this case, consists of natural organic-mineral material. Charcoal remains were discovered on the border between the plaggen horizon and mineral subsoil, thus emphasizing the anthropogenic filling character of the soil.

**Table 5.2** Profile description of a typical plaggen soil (Data from Meuser and Blume 2001; Meuser 2006)

| | |
|---|---|
| Site | Kalkriese (Osnabrück district), Germany |
| Relief | Midslope |
| Land useVegetation | Spruce woodland*Picea abies*, lacking in underwood |
| Cut and fill | Plaggen fill |
| Groundwater level | Extremely deep (>20 dm) |
| Depth (cm) | Description |
| 0–30 | Slightly silty sand, skeleton-free, medium humic, 10YR 2/3, low to medium compactness, sub-angular blocky structure, strong root penetration, carbonate-free, distinct transition |
| | Additions: brick and metal (isolated findings) |
| 30–60 | Slightly silty sand, skeleton-free, slightly humic, 10YR 3/4, low to medium compactness, single grain structure, poor root penetration, carbonate-free, distinct transition |
| | Additions: charcoal (3–15%), very low technogenic C-level |
| 60–130 | Sand, very low skeletal content, humus-free, 10YR 5/8 (dominant), low compactness, single grain structure, no rootlets, carbonate-free, root tubes, reduction-oxidation (dark rust colours, brown-black), diffuse transition |
| | Additions: none |
| 130–160+ | Sand, very low skeletal content, humus-free, 10YR 7/4, low to medium compactness, single grain structure, no roots, carbonate-free, reduction-oxidation (brown-black) |
| | Additions: none |
| Type of soil (WRB) | Plaggic Anthrosol |
| Parent material | Sands of slope position |

The most significant results of the analyses were as follows (Table 5.3):

- The slightly higher silt, clay and humus content of the plaggen horizons (0–60 cm) induced a slightly increased effective field moisture capacity compared to the sands of the subsoil.
- Despite earlier plaggen farming, the site must be classified as badly supplied as far as macronutrients are concerned.
- The supply of humus to the plaggen horizon is also classified as poor to medium; the carbon belonged to the humus and only low levels of charcoal were present.
- The soil showed strong acidic properties (low pH values, low base saturation).
- There was no soil contamination from heavy metals or Polycyclic Aromatic Hydrocarbons (PAH) (Meuser and Blume 2001; Meuser 2006).

## 5.3.2 Garden Soils

While plaggen soils are typical for locations on the periphery of urban agglomerations which had previously been used for agricultural purposes for a long time, garden soils are present in city areas as well. Residential areas with terraced buildings, but also

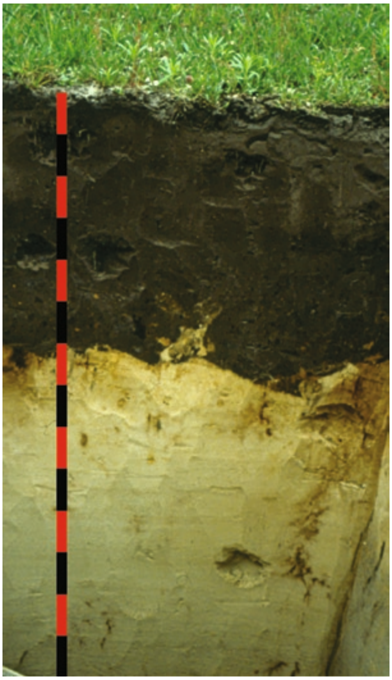

**Fig. 5.3** Typical plaggen soil in the district of Osnabrück, Germany (One measuring staff segment = 10 cm)

allotments and nurseries, contain garden soils used for both ornamental plants and vegetables. In garden soils (hortisols) the humic topsoil is usually thicker than 40 cm, the soil has accumulated humus as a result of continuous application of compost, dung, etc., indicates enhanced biological activity and has experienced repeated digging procedures. Section 6.2.5 reflects the nutrient status of garden soils in more detail.

Properties of garden soils are also visible in soils of parks, indicating a long history. In the capital of Germany, Berlin, with 3,400,000 inhabitants the Tiergarten located in proximity to the famous sights Brandenburger Tor and Reichstag and redeveloped in the year 1697, was intensively studied. Apart from the straightened relief and soil compaction caused by trampling feet the green areas which originated in woodland remained intact. Other parts that were tilled, fertilized with organic and mineral manure and irrigated during the dry season changed significantly in a similar way to horticultural soils. Lawns frequently used for recreational purposes of the inhabitants were more compacted and showed rill erosion in some overused areas. Moreover, along the footpaths eutrophication was visible due to waste thrown away. Due the long history of urban utilization the soil profiles indicated typical horticultural features. The humic topsoil of some investigated soil profiles reached a thickness of up to 40 cm, the organic matter content amounted to 0.97–1.53% and the C/N ratio was about 19 (Blume 1986).

**Table 5.3** Physical and chemical properties of the plaggen soil (Data from Meuser and Blume 2001; Meuser 2006)

| **Depth (cm)** | **Skeleton (mass %)** | **Sand (mass %)** | **Silt (mass %)** | **Clay (mass %)** | **Bulk density (g $cm^{-3}$)** |
|---|---|---|---|---|---|
| 0–30 | 0 | 90 | 5 | 5 | 1.38 |
| 30–60 | 0 | 93 | 2 | 5 | 1.51 |
| 60–105 | 0 | 98 | 2 | 0 | 1.55 |
| 105–130 | 0 | 98 | 2 | 0 | 1.54 |
| 130–160+ | 2 | 98 | 2 | 0 | 1.55 |
| | | **vol% Water at pF** | | | |
| **Depth (cm)** | **TPV (vol%)** | **1.8** | **2.0** | **2.5** | **4.2** |
| 0–30 | 47 | 26 | 21 | 11 | 7 |
| 30–60 | 42 | 20 | 14 | 6 | 3 |
| 60–105 | 41 | 16 | 10 | 4 | 2 |
| 105–130 | 41 | 21 | 16 | 5 | 3 |
| 130–160+ | 41 | 15 | 10 | 3 | 2 |
| **Depth (cm)** | **$C_t$ (mass %)** | **$C_{humus}$ (mass %)** | **$C_{technogenic}$ (mass %)** | **$N_{org.}$ (mass %)** | **C/N** |
| 0–30 | 1.3 | 1.3 | 0.0 | 0.086 | 15 |
| 30–60 | 0.8 | 0.8 | 0.0 | 0.047 | 17 |
| 60–105 | 0.2 | 0.2 | 0.0 | 0.009 | – |
| 105–130 | 0.2 | 0.2 | 0.0 | 0.005 | – |
| 130–160+ | 0.1 | 0.1 | 0.0 | <0.001 | – |
| **Depth (cm)** | **pH $CaCl_2$** | **pH $H_2O$** | **P-$H_2O$ (mg $100g^{-1}$)** | **EC (dS $m^{-1}$)** | **$CaCO_3$ (mass %)** |
| 0–30 | 3.4 | 4.0 | 1.1 | 0.04 | 0.0 |
| 30–60 | 4.2 | 5.0 | 0.8 | 0.03 | 0.0 |
| 60–105 | 4.3 | 4.8 | 0.1 | 0.03 | 0.0 |
| 105–130 | 4.4 | 4.7 | 0.2 | 0.03 | 0.0 |
| 130–160+ | 4.3 | 4.9 | 0.1 | 0.02 | 0.0 |

| | Exchangeable bases ($mmol_c\ kg^{-1}$) | | | | | |
|---|---|---|---|---|---|---|
| **Depth (cm)** | **Ca** | **Mg** | **K** | **Na** | **$\Sigma PAH_{EPA}$ ($mg\ kg^{-1}$)** | |
| 0–30 | 3.2 | 1.2 | 0.3 | 0.2 | nd | |
| 30–60 | 12.6 | 1.1 | 0.3 | 0.3 | nd | |
| 60–105 | 1.6 | 0.2 | 0.2 | 0.1 | nd | |
| 105–130 | 0.8 | 0.1 | 0.2 | 0.1 | nd | |
| 130–160$^+$ | 1.6 | 0.2 | 0.2 | 0.1 | nd | |
| **Depth (cm)** | **Cd ($mg\ kg^{-1}$)** | **Cr** | **Cu** | **Ni** | **Pb** | **Zn** |
| 0–30 | 0.1 | 11 | 4 | 3 | 18 | 12 |
| 30–60 | nd | 8 | 3 | 2 | 8 | 10 |
| 60–105 | nd | 6 | 2 | 2 | 4 | 18 |
| 105–130 | 0.1 | 7 | 2 | 4 | 3 | 23 |
| 130–160$^+$ | 0.0 | 6 | 1 | 4 | 3 | 11 |

nd = not detectable

PAH = polycyclic aromatic hydrocarbons; TPV = total pore volume

Metal extraction method: aqua regia

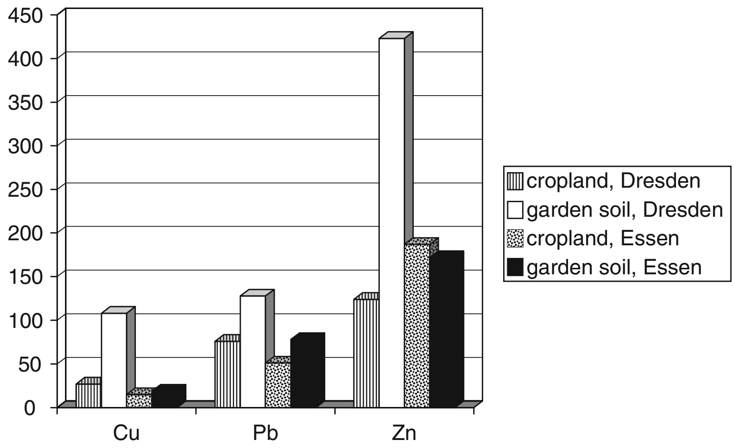


**Fig. 5.4** Influence of ash application on heavy metal values (mg $kg^{-1}$) in garden topsoils of two German cities

It should be taken into consideration that garden soils can be contaminated resulting from ash application to the soil surfaces in order to improve nutrient supply. The ashes, however, may consist of a high concentration of heavy metals, leading to an enhanced contaminant level compared with neighbouring cropland (Meuser and Anlauf 2007) (Fig. 5.4). The ashes applied stem not only from the burning of domestic fuels such as coal – waste has additionally been burnt, causing the high heavy metal concentrations. Moreover, comparable results have been found for other contaminants. For instance, it was not possible to detect benz(a)pyrene in cropland topsoils in the suburbs of Essen (Germany) with 580,000 inhabitants. Garden topsoils, however, revealed concentrations between 0.5 and 0.8 mg $kg^{-1}$.

Ashes derived from coal-fired power plants are often used to improve soil conditions in agricultural and horticultural sites in the vicinity of cities. The fly ash addition attributes increased water holding capacity and nutrient supply. In field experiments in different locations in India the application was related to yield increase of different crops such as groundnut, sunflowers and rice. For instance, results of experiments with *Arachis hypogea* and *Oryza sativa* showed significantly higher nutrient uptake and increased yield due to the addition of 20 and 40 t $ha^{-1}$ fly ash of coal-fired power stations in Mettur, India, a town with 60,000 inhabitants. This type of ash exhibited a low level of heavy metals (see Section 4.3.2) amounting to 35–44 mg Cr $kg^{-1}$, 20–30 mg Cu $kg^{-1}$, about 3 mg Pb $kg^{-1}$ and 40–190 mg Zn $kg^{-1}$. Cadmium was the only element with an enhanced concentration ranging between 3.8 and 4.8 mg $kg^{-1}$. Consequently, toxic elements were present only in traces in aboveground plant tissue like rice straw and rice corn. It was caused by different factors, reduced solubility in alkaline pH (8.2–9.2), leaching down to the subsurface layers and poor translocation from roots to the arial part of the plants (Baskar and Karthikeyan 2000).

Additions of household waste for fertilizing purposes increased the total phosphorus content significantly compared to natural soils. While in natural soils values from 1 to 5 mg $100g^{-1}$ were measured, in the urban area of Moscow concentrations between 200 and 550 mg $100g^{-1}$ were analyzed. An urban impact revealed other parameters providing nutrient status as well. For instance, the potassium content reached results up to 53 mg 100 $g^{-1}$ in anthropogenic soils in contrast to 6–14 mg $100g^{-1}$ in podsolic natural soils (Stroganova et al. 1998). The adverse effects of household waste amendment are mentioned in Sections 4.3.2 and 5.4.2.

In the following a typical garden soil will be introduced in detail (Table 5.4, Fig. 5.5). This site also demonstrates a history influenced by plaggen farming, a thick plaggen horizon developed as a result of the brown plaggen fill (see Section 5.3.1). However, the horticultural use of the site over several decades is even more significant for the profile characterization than plaggen farming. The area was used as a nursery from 1965 to the end of the nineties of the last century. Flowers were cultivated and trees were planted. Organic soil improvement agents, such as compost, peat and mulch, were spread out and worked in. Soil shifts took place as a result of planting and intercultivation. At the lower edge of the deeply cultivated horizon, signs of deep cultivation can still be observed today. In the mid-1990s intensive use of the area was terminated and since then most of the site has begun to ruderalize.

**Table 5.4** Profile description of a horticultural used soil (garden soil) (Data from Meuser and Blume 2001; Meuser 2006)

| | |
|---|---|
| Site | Osnabrück, Germany |
| Relief | Plane/depth position |
| Land use | Garden fallow land |
| Vegetation | Nitrophile ruderal vegetation (*Cirsium, Urtica*) |
| Cut and fill | Deposit of organic substrates |
| Groundwater level | Deep (8–13 dm) |
| Depth (cm) | Description |
| 0–45 | Medium loamy sand, very low skeletal content, medium to high humus, 10YR 2/2, low to medium compactness, friable structure, very strong root penetration, carbonate-free, earthworm and root tubes, sharp transition |
| | Additions: brick, ceramics, coal (each isolated findings) |
| 45–80 | Medium loamy sand, low skeletal content, very slightly humic, 10YR 4/4, mean compactness, subangular blocky to single grain structure, poor root penetration, carbonate-free, root tubes, distinct transition |
| | Additions: brick and coal (isolated findings) |
| 80–110 | Slightly loamy sand, low to medium skeletal content, humus-free, 10YR 5/6, low compactness, sub-angular blocky to single grain structure, very poor root penetration, carbonate-free, root tubes, distinct transition |
| | Additions: none |
| 110–120$^{+}$ | Slightly loamy sand, low to medium skeletal content, humus-free, 10YR 6/6, low compactness, single grain structure, no roots, carbonate-free, macropores, cracks, reduction-oxidation (light rust patches, bleaching) |
| | Additions: none |
| Type of soil (WRB) | Hortic anthrosol |
| Parent material | Tillite (over Lower Triassic weathering residues >120 cm) |

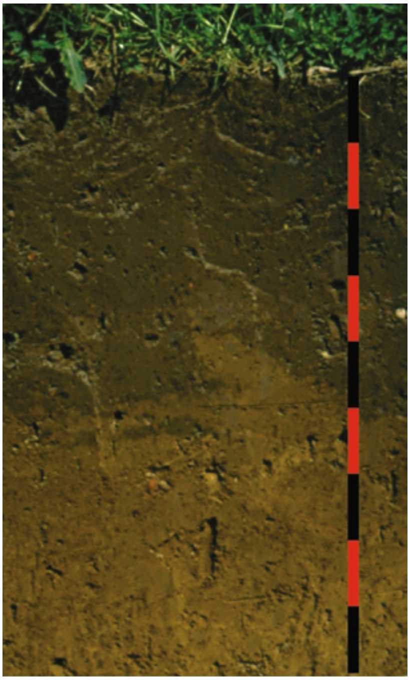

**Fig. 5.5** Long-term horticultural used soil in Osnabrück, Germany (One measuring staff segment = 10 cm)

In its present condition, a few years after horticultural use of the area terminated, the site contained lush, nitrophile ruderal vegetation. Organic material was continuously spread and worked in here over a long period of time, leading to the formation of thick, humic topsoil. Underneath this a brown plaggen horizon can be found, due to former agricultural use (up to the beginning of the twentieth century), so that altogether the thickness of the anthropogenic fill amounted to 80 cm.

The most significant results of the analyses were as follows (Table 5.5):

- Due to long-term horticultural use (addition of compost and peat, planting hole fertilization of planted trees, mulch application, etc.) the topsoil has an increased humus content, a close C/N ratio and a good nutrient supply.
- Air capacity and field moisture capacity up to a depth of 110 cm were high.
- There was no soil contamination from heavy metals or PAH at this site either (Meuser and Blume 2001; Meuser 2006).

### 5.3.3 Cemetery Soils

Specified conditions can be expected with reference to deep cultivated cemetery and churchyard soils called necrosols. Most religions (e.g. the Christian, Jewish and Islamic religions) bury the corpses of dead persons in the ground after excavation

**Table 5.5** Physical and chemical properties of the soil with long-term horticultural use (Data from Meuser and Blume 2001; Meuser 2006)

| Depth (cm) | Skeleton (mass %) | Sand (mass %) | Silt (mass %) | Clay (mass %) | Bulk density (g $cm^{-3}$) | | |
|---|---|---|---|---|---|---|---|
| 0–45 | 8 | 67 | 25 | 8 | 1.44 | | |
| 45–80 | 9 | 68 | 26 | 7 | 1.41 | | |
| 80–110 | 19 | 70 | 24 | 6 | 1.46 | | |
| 110–120$^+$ | 12 | 75 | 20 | 4 | – | | |
| | | **vol% Water at pF** | | | | | |
| **Depth (cm)** | **TPV (vol%)** | **1.0** | **1.8** | **2.5** | **3.0** | **3.7** | **4.2** |
| 0–45 | 46 | 37 | 31 | 21 | 19 | 9 | 7 |
| 45–80 | 46 | 34 | 26 | 14 | 11 | 6 | 3 |
| 80–110 | 45 | 32 | 20 | 12 | 11 | 6 | 3 |
| 110–120$^+$ | – | – | – | – | – | – | – |
| **Depth (cm)** | **Ct (mass %)** | **$C_{humus}$ (mass %)** | **$C_{technogenic}$ (mass %)** | **$N_{org.}$ (mass %)** | **C/N** | **$CaCO_3$ (mass %)** | |
| 0–45 | 1.8 | 1.8 | 0.0 | 0.147 | 12 | 0.0 | |
| 45–80 | 0.6 | 0.6 | 0.0 | 0.053 | 11 | 0.0 | |
| 80–110 | 0.2 | 0.2 | 0.0 | 0.035 | – | 0.0 | |
| 110–120$^+$ | 0.1 | 0.1 | 0.0 | 0.015 | – | 0.0 | |
| **Depth (cm)** | **pH $CaCl_2$** | **pH $H_2O$** | **P-$H_2O$ (mg $100g^{-1}$)** | **EC (dS $m^{-1}$)** | | | |
| 0–45 | 5.4 | 6.3 | 2.2 | 0.05 | | | |
| 45–80 | 5.3 | 6.3 | 0.9 | 0.03 | | | |
| 80–110 | 5.3 | 6.2 | 0.4 | 0.03 | | | |
| 110–120$^+$ | 5.3 | 6.4 | 0.2 | 0.02 | | | |

(continued)

**Table 5.5** (continued)

| Depth (cm) | Exchangeable bases ($mmol_c$ $kg^{-1}$) Ca | Mg | K | Na | $\Sigma PAH_{EPA}$ ($mg\ kg^{-1}$) | |
|---|---|---|---|---|---|---|
| 0–45 | 52.1 | 6.0 | 3.3 | 0.2 | 0.7 | |
| 45–80 | 18.9 | 4.0 | 4.2 | <0.1 | nd | |
| 80–110 | 11.0 | 3.4 | 4.4 | <0.1 | nd | |
| $110–120^{+}$ | 11.0 | 3.8 | 4.3 | <0.1 | nd | |
| **Depth (cm)** | **Cd ($mg\ kg^{-1}$)** | **Cr** | **Cu** | **Ni** | **Pb** | **Zn** |
| 0–45 | 0.3 | 24 | 14 | 8 | 33 | 57 |
| 45–80 | 0.1 | 15 | 7 | 7 | 11 | 32 |
| 80–110 | nd | 21 | 7 | 9 | 7 | 36 |
| $110–120^{+}$ | nd | 21 | 5 | 10 | 7 | 28 |

nd = not detectable
PAH = polycyclic aromatic hydrocarbons; TPV = total pore volume
Metal extraction method: aqua regia

**Fig. 5.6** Cemetery landscape (left) and soil profile (opened grave, right) in Stuttgart, Germany (One measuring staff segment = 10 cm)

to a depth between 120 cm and approximately 180 cm. For re-fill, apart from the corpses, ingredients and soil material are used. The uppermost layer is often overlaid by humic topsoil in order to establish ornamental design (Fig. 5.6). Cemetery soils including abandoned cemetery soils in urban areas may amount to considerable percentages.

Due to the religious customs intensified soil investigation understandably did not occur in these areas but some characteristics have already been investigated. A possible cemetery contamination is reported due to incomplete decomposition of buried corpses. In moist soil tending to reductive conditions the cemeteries are confronted with the problem of insufficient decomposition, which prevents re-use of graves after a normal resting time of approximately 25 years. The creation of lipids (adipoceres) is mainly responsible for the limited corpse decomposition.

The slow and anaerobic decomposition is related to a number of disadvantageous effects. Fouling gases such as ammonia and $H_2S$ develop, causing unpleasant odor emission if the thickness of the cover layer is inappropriate. The survival time of the immense number of pathogenic microorganisms can increase, leading to a long-term transport into groundwater. A potential hazard to groundwater quality cannot be excluded. This is caused by pathogens created during corpse decomposition and by pathogens already incorporated in the body during lifetime. In cemetery soils the survival time of *Escherichia coli* amounts to more than 2 years, while some *Aspergillus* species might survive for up to 10 years. Very problematical pathogens possibly responsible for the death of the buried corpse such as plague, typhus and cholera bacteria are assumed to survive a relatively long period of time. In particular, in mass burial sites attention should be paid to the problem of substances administered prior to the death of the persons.

Radioactive substances used in diagnostic and therapeutic measures are also of concern, since incorporated radio isotopes remain unchanged in cemetery soils for a long time. Moreover, accessories and ingredients buried together with the corpse, amalgam fillings in teeth (Hg) and coffin fittings (Zn) can be sources of soil contamination in cemeteries (Evans 1963; Hawrylewicz et al. 1966; Kalberlah et al. 1997; Urban 2002).

## 5.4 Deposited Soils (Technosols)

### *5.4.1 Soils of Urban Built-up Areas*

As shown in Section 4.4, in the urban environment including residential, recreational, commercial and industrial areas a high percentage of man-made soils derived from deposits of technogenic materials has to be expected. The purpose of the examples below is to underline the deposition character of urban built-up areas. The sampling depth of most urban soils investigated is limited to the topsoils, since the causes of contamination are expected to be associated with atmospheric deposition. If the soil contamination is supposed to relate to technogenic material deposited, deep-reaching soil sampling will be necessary. Hence, the following built-up areas were discussed in relation to the whole soil profile.

Qingdao is a coastal city with 2.5 million inhabitants in the inner city and since 1945 it has been one of the most important ports in China. Today, it is strongly industrialized with heavy industry and chemical companies, shipbuilding, car production, etc. Investigations of four sites located in the inner city and covering soils of an industrial site, the old city, the modern centre and a coastal construction site resulted exclusively in anthropogenic soils with former excavation processes and refilling activities. The natural parent material was found at a depth between 36 and 205 cm below the surface. Above the surface mainly construction rubble such as brick, mortar, gypsum and lime was identified, influencing soil properties considerably. In detail, calcium content chemically determined as calcite had accumulated combined with enhanced pH values. In the anthropogenic layers pH values between 7.0 and 8.1 were found, causing low heavy metal mobility.

The heterogeneity of the soil profiles led to the differentiation of many soil horizons with a few centimetres in thickness only. In general, the heterogeneity of urban soils refers to both the horizontal and vertical direction. Clear texture and composition differences from horizon to horizon were good indicators of the urban soil heterogeneity. Furthermore, the soil colour revealed varicoloured mixtures. Coal remains enhanced the carbon content in some horizons, so that high C values were detected in the subsoil horizons, whilst the typical decrease in carbon content with increasing depth which is well-known for natural soils was not confirmed.

The horizons consisting of anthropogenic artifacts tended to reveal higher heavy metal concentrations than an additionally analyzed natural soil and the parent material below. The soil profile in the old town of Qingdao consisted of a material enriched with gypsum, coal and construction rubble (depth 12–45 cm) with enhanced values for lead (53–128 mg $kg^{-1}$) and zinc (136–429 mg $kg^{-1}$). Moreover, pollutants like barium, which are usually not of concern in urban soil research, indicated accumulated concentration as well, reaching values between 704 and 1,420 mg $kg^{-1}$. Another soil profile that was investigated in the city centre and contained different kinds of building rubble, including tiles, had barium values from 1,000 to 1,620 mg $kg^{-1}$. It can be summarized that construction rubble may play an important role in pollutant interpretation (Norra et al. 2008).

In Lodz (Poland) with its 1,000,000 inhabitants several profiles in lawns near streets and parks were dug, sampled and analyzed. The deeper soil horizons were partly altered by construction activity. Exact percentages of construction debris were not mentioned in the study. Nevertheless, some high heavy metal concentrations in deeper layers, which were mainly recognizable in the city profiles and to a less extent in the suburban area, revealed the contamination source with regard to deposits of distinct materials, including construction debris (Table 5.6). The superficial layers (0–5 cm) were more highly polluted, especially in street areas. This was caused by traffic and dust deposition, among other factors. Some maximum values for Pb and Zn in deeper layers, however, cannot be explained by this factor and seemed to be typical results from man-made soils containing technogenic substrates (Czarnowska and Chojnicki 2000).

The heavy metal results yielded in the Russian capital, Moscow, with its 10,500,000 inhabitants (Table 5.7), confirm the tendencies already mentioned before. The topsoils to a depth of about 30 cm exhibited strongly enhanced concentrations exceeding the background values. Considering the depth of 30–52 cm in profile no. 2 the subsoils also had values higher than the background values of Moscow. The urban impact, particularly the man-made soil disturbance as well as the addition of artefacts, may provide the reason. It should be noted that the background values themselves varied considerably depending upon parent material and texture. For example, the Pb values ranged from 6 (podsolic soils)–20 mg $kg^{-1}$ (loess chernozems) (Stroganova et al. 1998).

The research project dealing with the sites in Moscow was based on heavy metal analysis using 1 n $HNO_3$ extraction (total concentration). Additionally, metals were detected with 1 n HCl, symbolizising the mobile forms of heavy metals in a better way. On average, the percentage of mobile forms related to the total concentration showed approximately 20% (Cr), 25% (Co), 40% (Cu, Ni, Zn) and 45% (Pb) respectively. Thus, the total concentration does not sufficiently explain the pathways endangering human health, plants, and groundwater (Stroganova et al. 1998) (see Section 6.2.1).

Generally speaking, in the presence of construction rubble urban soils reveal increased heavy metal concentration independent of the research location. The famous Mall in Washington DC, USA, with a population of 580,000, exhibited higher values in comparison with nearby undisturbed areas. For example, the Pb values of the upper metre ranged from 101 to 210 mg $kg^{-1}$ (Craul 1992).

The soil profiles sometimes contain residual foundation walls and floors, utility traces and tunnels and underground storage tanks filled with problematical liquids such as gasoline. Accordingly, the soil conditions are less favourable. The tanks left in place can be partially damaged or destroyed causing losses of the contaminated liquids (Craul 1992).

**Table 5.6** Heavy metal contents (mg $kg^{-1}$) of soils in Lodz, Poland (Data from Czarnowska and Chojnicki 2000)

| Depth | Cd | | Cu | | Pb | | Zn | |
|---|---|---|---|---|---|---|---|---|
| (cm) | Mean | Range | Mean | Range | Mean | Range | Mean | Range |
| **Lawns near streets in central part of the city (27 profiles)** | | | | | | | | |
| 0–5 | **1.1** | 0.5–1.8 | **59** | 17–138 | **110** | 40–409 | **319** | 88–774 |
| 5–10 | **0.9** | 0.2–2.0 | **40** | 7–108 | **77** | 17–217 | **293** | 25–706 |
| 10–20 | **0.7** | 0.3–1.5 | **40** | 7–111 | **70** | 12–225 | **265** | 35–554 |
| 90–125 | **0.3** | 0.1–0.9 | **13** | 2–82 | **30** | 3–200 | **75** | 12–424 |
| **Parks in central part of the city (19 profiles)** | | | | | | | | |
| 0–5 | **1.0** | 0.5–2.4 | **30** | 9–76 | **62** | 40–146 | **210** | 86–312 |
| 5–10 | **0.9** | 0.2–2.0 | **29** | 8–91 | **55** | 21–144 | **184** | 53–300 |
| 10–20 | **0.7** | 0.2–1.0 | **18** | 7–87 | **44** | 11–100 | **147** | 80–306 |
| 90–125 | **0.2** | <0.1–0.5 | **9** | 3–58 | **20** | 5–66 | **47** | 15–126 |
| **Suburban area (17 profiles)** | | | | | | | | |
| 0–5 | **0.5** | 0.1–1.1 | **16** | 2–52 | **32** | 4–63 | **115** | 18–336 |
| 5–10 | **0.4** | 0.1–1.0 | **14** | 2–50 | **35** | 6–114 | **112** | 12–324 |
| 10–20 | **0.4** | 0.1–1.0 | **13** | 2–48 | **26** | 3–70 | **93** | 16–344 |
| 90–125 | **0.2** | 0.1–0.5 | **5** | 2–20 | **10** | 2–21 | **16** | 9–35 |

Metal extraction method: concentrated acids ($HNO_3$, $H_2SO_4$, HCl)

**Table 5.7** Chemical properties in soils of residential areas in the city centre of Moscow, Russia (Data from Stroganova et al. 1998)

| | | | Base | P | K | Cd | Cu | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|
| Depth (cm) | TOC (%) | pH | saturation % | (mg $100g^{-1}$) | | (mg $kg^{-1}$) | | | |
| **Profile no. 1** | | | | | | | | | |
| 0–10 | 4.9 | 7.5 | nd | 23.2 | 40.8 | 1.3 | 82 | 600 | 734 |
| 10–20 | 4.4 | 7.5 | nd | 33.8 | 37.3 | 1.2 | 76 | 560 | 588 |
| 30–60 | 1.9 | 7.8 | nd | 29.5 | 40.8 | 0.5 | 88 | 25 | 58 |
| 60–115 | 0.7 | 8.3 | nd | 26.8 | 23.6 | 0.4 | 6 | 18 | 22 |
| **Profile no. 2** | | | | | | | | | |
| 0–15 | 10.8 | 7.4 | 97 | 20.2 | 12.4 | 1.8 | 262 | 64 | 1,000 |
| 15–30 | 11.2 | 7.6 | 97 | 33.4 | 23.3 | 1.0 | 88 | 160 | 336 |
| 30–52 | nd | nd | nd | nd | nd | 0.4 | 33 | 96 | 110 |
| **Profile no. 3** | | | | | | | | | |
| 0–10 | 8.0 | 7.6 | 96 | 36.7 | 20.7 | 0.7 | 34 | 74 | 178 |
| 10–20 | 6.0 | 7.6 | 97 | 33.8 | 14.3 | 0.6 | 38 | 85 | 148 |
| 35–50 | 1.7 | 8.2 | 99 | 1.8 | 7.0 | 0.5 | 16 | 52 | 16 |
| 90–130 | 1.8 | 8.3 | 99 | 59.0 | 6.3 | 0.5 | 21 | 37 | 30 |
| **Background values in Moscow** | | | | | | <0.1–0.24 | 8–25 | 6–20 | 28–68 |

nd = not detectable; TOC = total organic carbon
Metal extraction method: $HNO_3$

**Fig. 5.7** (**a**) Foundation-soil-mixture of an urban brownfield in Rotterdam, The Netherlands. (**b**) Demolition residues of an urban brownfield in Rotterdam, The Netherlands

Belowground structures like foundations are frequently not removed during brownfield redevelopment. Instead, former hollow structures have to be re-filled or stabilized rapidly. Occasionally, even tanks, power and phone cables and pipes are not removed (Genske 2003). Consequently, in some cases it may be problematical to distinguish between "soil" and foundation structures in situ. In areas under construction only little natural soil remains and the surface is covered with debris from buildings (demolition waste) previously occupying the site (Fig. 5.7a and b).

In European cities the influence of the Second World War has changed the urban soils. In Berlin, Germany, early urban soil investigations proved the soil alteration. After the war ruins were cleared, valuable artifacts were collected, rubble and soil were mixed, bomb craters refilled and land leveled. Afterwards, many sites remained untouched, resulting in an intensive development of vegetation starting with a first annual and perennial stage of herbaceous plants, followed by a shrub stage and finally by the establishment of urban forest. In the course of time particularly the tree species *Robinia pseudoacacia,* which was able to survive despite unfavourable soil conditions such as lack of nutrients and water, became dominant. The legume tree was able to compensate for nitrogen deficiency and gathered water from depths of 2 m, indicating drought resistance. Later on, the nitrogen supply was improved by leaf litter of the tree, resulting in a long-term humus accumulation.

The soils consisted of two thirds fine earth and one third various skeleton. In the city centre of Berlin soil profiles revealed technogenic components such as bricks, mortar, coal, slag and other artificial products. According to the findings the calcium carbonate content varied between 1.7% and 11.0% and, in addition, accelerated heavy metal concentrations were detected. For instance, the copper values indicated ranges between 30 and 2,200 mg kg$^{-1}$ and lead varied between 200 and 24,000 mg kg$^{-1}$ (Blume 1986).

The deposition and dispersion of construction debris in urbanized areas after the Second World War or other wars are surely one important source of the high percentage of construction debris in urban soils, as described in Section 4.4.

However, the main reason might be connected with the civil demolition of buildings and other built-up facilities. Up to now in most countries the recycling of construction debris has failed and more or less unplanned distribution is taking place. The amount of debris constantly produced in the urban environment is enormous. It is much more than unforgettable spectacular occurrences forever showed, happy occurrences like the breakdown of the Berlin Wall in 1989 and sad occurrences like the 9/11 terror attack against the World Trade Centre in New York in 2001.

The connection between deposited rubble and higher pollutant values is also applicable to organic compounds. Deposited soils containing construction rubble tend to enhanced values of Polycyclic Aromatic Hydrocarbons (PAH). In a number of subsoil samples with construction rubble PAH concentrations have been found that cannot be explained by atmospheric deposition. For instance, the results varied from 1.5 to 2.1 mg $kg^{-1}$ (Halle, Germany, with 230,000 inhabitants), 1.3 to 3.0 mg $kg^{-1}$ (Rostock, Germany, with 200,000 inhabitants) and 0.2 to 1.6 mg $kg^{-1}$ (Kiel, Germany, with 240,000 inhabitants) mg $kg^{-1}$ $PAH_{EPA}$. Systematic evaluation of 59 deposited soils in Germany showed the principal enhanced $PAH_{EPA}$ status of urban soils, since 33 examples ranged between 1 and 50 mg $kg^{-1}$ and even five examples ranged between 50 and 500 mg $kg^{-1}$ $PAH_{EPA}$. It should be noted that some soils located in contaminated industrial sites were included (Blume and Schleuss 1997).

The site investigated in detail in the German district of Osnabrück is a typical example for soils deposited by different technogenic substrates in unplanned manner (Table 5.8, Fig. 5.8). It was situated in the bottom of the river valley. Humic to peaty alluvial deposits cover glacial sands of several metres' thickness. Until 1871, most of this area was marshland. With the aid of drainage ditches, dewatering and reclamation of this area, which is close to the city, were attempted at an early date. From 1875, ever-increasing areas were drained and built on, first of all using remainders of the city wall and later using construction debris, dumped garbage (including municipal household garbage) and ashes from domestic fuel. Most buildings were constructed in the 1950s and 1960s. Responsibility for the elevation in the landscape or drainage lay with the individual house-owners. This resulted in a very heterogeneous substrate mixture being dumped. The whole landfill area covered 220 ha and housed 18,000 inhabitants; today it is the largest inhabited contaminated and later rehabilitated site in the whole of Germany. Since the beginning of the 1992s, the public administration has been dealing with the risk assessment and clean-up of the area (Meuser and Blume 2001). The most significant results of the risk assessment were:

- The thickness of the deposited material was between 1 and 2 m
- The substrate spectrum covered different technogenic substrates
- Partly highly increased values existed for heavy metals (in particular Cu, Pb, Zn) and Polycyclic Aromatic Hydrocarbons, depending on the substrate
- The upper aquifer was contaminated, particularly in the east of the area

The site belonged to the grounds of a school (lawn with ornamental trees). The filled humic surface soil was followed by an approximately 40 cm thick fill of soil, building debris and isolated waste components. From a depth of 60 cm, distinct

**Table 5.8** Profile description of a heterogeneously deposited urban soil in a built-up area (Data from Meuser and Blume 2001; Meuser 2006)

| | |
|---|---|
| Site | Osnabrück, Germany |
| Relief | Plane |
| Land use | Lawn |
| Vegetation | Lawn with solitary shrubs (*Carpinus betulus, Sorbus aucuparia, Ligustrum vulgare*) |
| Cut and fill | Anthropogenic deposit/landfill |
| Groundwater level | Deep (8–13 dm) |
| Depth (cm) | Description |
| 0–20 | Slightly silty sand, very low skeletal content, poor to medium humic, 10YR 3/2, low compactness, slightly granular to single grain structure, strong root penetration, low carbonate content, root and earthworm tubes, sharp transition |
| | Additions: brick, glass, ceramics (each isolated findings) |
| 20–60 | Slightly loamy sand, medium skeletal content, very slightly humic, 10 YR 5/3, high compactness, single grain structure, very poor root penetration, high content of carbonate, cracks, macropores, reduction-oxidation (dark rust colours), odor of $H_2S$ and fouling gas, distinct transition |
| | Additions: brick (3–15%), mine debris, coal, ceramics, metal, glass, rubber, synthetic material, bones, furnace release (each isolated findings), low technogenic C-content |
| 60–120 | Slightly loamy sand, high skeletal content, N 3, very high compactness, single grain structure, without roots, very high content of carbonate, containing sulphide, cracks, macropores, reduction-oxidation (black), intensive odor of $H_2S$ and fouling gas, transition distinct to diffuse |
| | Additions: brick (3–15%), mine debris (3–15%), furnace release (3–15%), wood (3–15%), glass, plant remnants, coal (isolated findings), low technogenic C-content |
| 120–170$^{+}$ | Slightly loamy sand, very low skeletal content, medium to very humic, 5YR 2.5/1, very low compactness, without roots, low carbonate content, containing sulphide, macropores, reduction-oxidation (green-grey to blue-grey), odor of $H_2S$ and fouling gas |
| | Additions: none |
| Type of soil (WRB) | Technosol/(Thionic Gleysol) |
| Parent material | Mixture of soil, building debris and garbage over humic alluvial sediments |

signs of reduction-oxidation were visible (black colouration as a result of sulphide formation, the odor of $H_2S$); the substrate spectrum now revealed organic components, too, such as plant remains and wood. The groundwater temporarily reached this horizon. The garbage mass ended at a depth of approximately 120 cm; it was also possible to see there a reduction-characterized, fossil A horizon without technogenic material.

The most significant results of the analyses were as follows (Table 5.9):

- The increased carbon content in the deeper horizons was explainable on the one hand by their humus components (root remains, wood, plant remains), and on

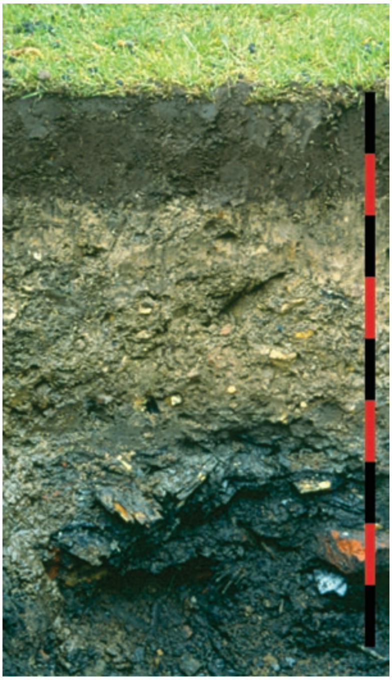

**Fig. 5.8** Heterogeneously deposited urban soil in a built-up area in Osnabrück, Germany (One measuring staff segment = 10 cm)

the other hand by technogenic components (coal, furnace release, coal mining waste). Accordingly, the C/N ratio of the depth from 20 to 120 cm emphasized the non-humus origin of the carbon.

- Strong fluctuations occurred with regard to nutrient contents in the soil, depending on the substrate.
- With the exception of surface soil, the pH values were in the low alkaline spectrum, which is typical for tipped soil consisting of building debris, garbage, and ash, respectively.
- The soil showed a slightly increased level of Polycyclic Aromatic Hydrocarbons.
- Increased lead concentrations were measured in the subsoil; furthermore, the subsoil stood out because of its high conductivity values, $CaCO_3$ content and exchangeable Ca (Meuser and Blume 2001; Meuser 2006).

### 5.4.2 Landfill Soils

Soils of deposited garbage are preferentially discussed with respect to less developed countries, where open dumpsites are visible in city areas and seem to be a disturbing factor in an urban lifestyle.

**Table 5.9** Physical and chemical properties of the heterogeneously deposited urban soil (Data from Meuser and Blume 2001; Meuser 2006)

| Depth (cm) | Skeleton (mass %) | Sand (mass %) | Silt (mass %) | Clay (mass %) | | |
|---|---|---|---|---|---|---|
| 0–20 | 2 | 77 | 17 | 6 | | |
| 20–60 | 52 | 77 | 15 | 8 | | |
| 60–120 | 58 | 72 | 21 | 8 | | |
| $120–170^{+}$ | 5 | 70 | 21 | 9 | | |
| **Depth (cm)** | **$C_t$ (mass %)** | **$C_{humus}$ (mass %)** | **$C_{technogenic}$ (mass %)** | **$N_{org.}$ (mass %)** | **C/N** | **$CaCO_3$ (mass %)** |
| 0–20 | 2.1 | 1.7 | 0.3 | 0.124 | 17 | 0.0 |
| 20–60 | 2.1 | 0.7 | 1.4 | 0.052 | 40 | 5.5 |
| 60–120 | 4.1 | 3.0 | 1.1 | 0.078 | 53 | 20.5 |
| $120–170^{+}$ | 3.6 | 2.6 | 1.0 | 0.204 | 18 | 0.5 |
| **Depth (cm)** | **pH $CaCl_2$** | **pH $H_2O$** | **P-$H_2O$ (mg $100g^{-1}$)** | **EC (dS $m^{-1}$)** | | |
| 0–20 | 5.9 | 6.5 | 0.5 | 0.06 | | |
| 20–60 | 7.4 | 8.0 | 0.4 | 0.11 | | |
| 60–120 | 7.5 | 7.6 | 0.1 | 2.27 | | |
| $120–170^{+}$ | 7.2 | 7.4 | 0.5 | 0.60 | | |
| | **Exchangeable bases (mmol$_c$ kg$^{-1}$)** | | | | | |
| **Depth (cm)** | **Ca** | **Mg** | **K** | **Na** | **$\Sigma PAH_{EPA}$ (mg kg$^{-1}$)** | |
| 0–20 | 70.2 | 3.6 | 1.1 | 0.1 | 0.6 | |
| 20–60 | 342.2 | 4.6 | 0.9 | 0.3 | 6.1 | |
| 60–120 | 672.7 | 17.6 | 4.6 | 1.1 | 8.8 | |
| $120–170^{+}$ | 250.6 | 16.5 | 2.9 | 0.5 | 2.9 | |
| **Depth (cm)** | **Cd (mg kg$^{-1}$)** | **Cr** | **Cu** | **Ni** | **Pb** | **Zn** |
| 0–20 | 0.2 | 24 | 23 | 7 | 37 | 82 |
| 20–60 | 0.2 | 26 | 39 | 19 | 78 | 176 |
| 60–120 | 0.4 | 17 | 17 | 13 | 417 | 355 |
| $120–170^{+}$ | 0.6 | 32 | 31 | 14 | 78 | 208 |

PAH = polycyclic aromatic hydrocarbons
Metal extraction method: aqua regia

Waste dumps are wide-spread particularly in countries where waste management is poorly developed. In India and Pakistan, for instance, the waste disposal can be assessed as problematical. The reasons are low collection capacity, in particular outside big cities, lack of landfill management, reduced recycle activities to street peddlers and rag-pickers, waste dispersion through animals (cattle), a low awareness in relation to waste problems, absence of garbage incineration or restricted only to private companies, hotels and hospitals and a low level of composting technology.

The waste handling is strongly related to the social conditions in which people live. Illiterate poor people living in slums in un-hygienic living conditions promote soil pollution in their own areas. Typical examples underlining the circumstances are the room heating and cooking in poor localities, where different materials are burnt in portable steel stoves. Residues of at least 15–20% are left as small pieces of coal and ash and it is an easy and common method to throw this away on unpaved areas close to the site of origin. Another example is the change and disposal of lubricating oil of motorized vehicles used for people and goods transportation on vacant land by the drivers. Furthermore, urban domestic waste is dumped, particularly in low lying areas around the municipalities. It is partly biodegradable (for example, paper and kitchen waste) but the majority of the materials disposed of are non-biodegradable (for example, plastic, metallic cans and glass), with the result that these materials remain there for a very long period of time. It has been found that in urban areas waste deposits are often in the near-surface soil horizon, which contains a lot of non-degradable waste, particularly plastic bags, hazardous industrial waste and cattle waste. Because some inhabitants are aware of the missing biodegradability, they put on fires resulting in slow burning of plastics associated with smoke and odour development in the city. Another aspect of the uncontrolled waste disposal in the areas mentioned relates to the simultaneous dumping of industrial waste. Sites, where domestic waste has already accumulated are frequently used for additional illegal disposal of industrial waste. In Pakistan, for instance, sludges of the textile industry are disposed of in low lying areas together with domestic waste. The sludge is derived from printing of various dyes on clothes. The surplus of the heavy metal oxides containing dyes after the washing process is drained out, collected and afterwards dried in open plots. After evaporation the residual material is transported to dump sites in the immediate vicinity (Panhwar 2000).

The landfill sites in India have been investigated intensively. There, dumping grounds were mainly open low-lying areas in or next to the cities. The waste was generally dumped without any segregating into biodegradable and non-biodegradable parts. The Municipal Solid Waste (MSW) came from domestic, agricultural and commercial sources and there was also construction rubble. In addition, highly toxic industrial and medical wastes illegally dumped were very often found. The rapid urbanization led to a strong growth of dumping sites, causing disposal problems for the neighbourhood. The higher temperatures characteristic of the Indian climate resulted in a higher rate of biodegradation and odor development of fouling gases. Especially (the odorless) methane is produced continuously in large quantities independent of the landfill management. Even after the waste was buried at closed sites methane generation took place permanently. The neutral pH value of

**Table 5.10** Average total concentration of heavy metals (mg $kg^{-1}$) and pH values in deposited municipal solid waste of six Indian cities (Data from Rawat et al 2008)

| City | pH | Cd | Cr | Cu | Hg | Ni | Pb |
|---|---|---|---|---|---|---|---|
| Ahmedabad | 8.7 | 2.5 | 21.5 | 34.4 | nd | 16.6 | 10.8 |
| Bangalore | 8.2 | 0.3 | 6.7 | 42.0 | 0.6 | 1.2 | 5.6 |
| Chennai | 8.0 | 0.4 | 40.4 | 45.4 | 2.1 | 6.2 | 18.5 |
| Dehradun | 7.2 | 0.3 | 9.6 | 28.4 | nd | 3.2 | 6.3 |
| Kolkata | 7.9 | 1.3 | 28.5 | 40.4 | - | 8.2 | 12.4 |
| Delhi | 7.8 | 0.6 | 23.0 | 45.0 | - | 9.5 | 6.8 |

nd = not detectable
Metal extraction method: aqua regia + HF

the waste that has been mostly analyzed improved the methanogenesis. Thus, in Indian dumpsites $CH_4$ generated ranged from 146 to 454 mg $m^{-2}$ $h^{-1}$. By comparison in Swedish dumpsites (humid, continental climate) the range was 0.5–320 mg $m^{-2}$ $h^{-1}$ (Rawat et al. 2008).

The above mentioned investigations of waste dumping sites took six agglomerations into account, e.g. Kolkata (India) with its 15,400,000 inhabitants and the Indian capital, Delhi, with its 18,000,000 inhabitants. The organic matter content indicated values between 34.9% and 47.5%, nitrogen between 0.39% and 0.54% and the C/N ratio exhibited results from 11 till 30. Table 5.10 summarizes the heavy metal concentration and pH values which were found (Rawat et al. 2008). The heavy metal pollution seemed to be negligible because the waste fragments did not reflect seriously increased values.

In Jind, India, with a population of 190,000 a waste deposit located in the middle of the city close to a hospital (Fig. 5.9) was investigated to a depth of 200 cm (Table 5.11). For the elements Cd, Cu and Zn enhanced aqua regia extractable metal values were detected. The results showed a wide range related to the different waste materials sampled. It was not possible to find a texture-related difference between the coarse fraction >2 mm and the fine earth, which were analyzed separately. The DTPA extractable concentration indicated relatively high amounts for the elements cadmium and copper. Hence, these metals tended apparently to accelerated mobile percentages. A depth gradient between the first and the second meter sampled has not been found. The pH value reacted neutrally and the electrical conductivity indicated enhanced results. This parameter was the only one showing a depth gradient with higher values below resulting from leaching of soluble salts.

The carbon content varied between 3.4% and 5.6% in association with a relatively low percentage of biodegradable organic matter. The waste differentiation showed 65.6% fine earth and 25.2% construction debris, whilst other components did not play a major role (Fig. 5.10). Organic materials were partly removed, since animals consume the edible percentages. Besides cows, pigs, dogs and birds use the wide-spread unplanned heaps of garbage in city areas, leading to an uncontrolled dispersion of the waste materials. The high percentage of fine earth results from the special situation in Indian cities (and cities in other less developed or developing countries) where the

**Fig. 5.9** Location of the waste heap in the city of Jind, India

**Table 5.11** Aqua regia (AR) and DTPA extractable heavy metal concentration (mg $kg^{-1}$), pH ($H_2O$) value, electrical conductivity EC (dS $m^{-1}$) and carbon content (%) of deposited waste material (depth 0–2 m) in the city of Jind, India (Data unpublished [data from a current research project of the Haryana Agricultural University Hisar, India and the University of Applied Sciences Osnabrück, Germany])

| | | 0–200 cm | | | |
|---|---|---|---|---|---|
| | | Fraction >2 mm | | Fraction <2 mm | |
| | | Mean | Range | Mean | Range |
| Cd | AR | **3.6** | 0.5–7.9 | **4.2** | 0.4–11.0 |
| Cr | AR | **49** | 39–55 | **46** | 36–54 |
| Cu | AR | **325** | 50–971 | **173** | 36–368 |
| Ni | AR | **28** | 25–29 | **27** | 22–29 |
| Pb | AR | **137** | 48–234 | **125** | 31–176 |
| Zn | AR | **341** | 190–551 | **320** | 143–618 |
| Cd | DTPA | **1.0** | 0.1–2.3 | **0.9** | 0.1–2.5 |
| Cr | DTPA | **0.25** | 0.2–0.3 | **0.2** | 0.1–0.3 |
| Cu | DTPA | **16.6** | 14.1–22.2 | **13.6** | 11.4–16.7 |
| Ni | DTPA | **0.9** | 0.3–1.1 | **0.6** | 0.2–0.9 |
| Pb | DTPA | **12.3** | 7.3–15.8 | **9.2** | 4.0–13.1 |
| Zn | DTPA | **9.6** | 8.9–10.4 | **9.4** | 8.1–10.4 |
| pH | | **7.45** | 7.4–7.5 | **7.5** | 7.4–7.6 |
| EC | | **1.22** | 0.82–1.63 | **1.45** | 0.71–1.66 |
| C | | **5.2** | 4.3–5.6 | **4.2** | 3.4–5.1 |

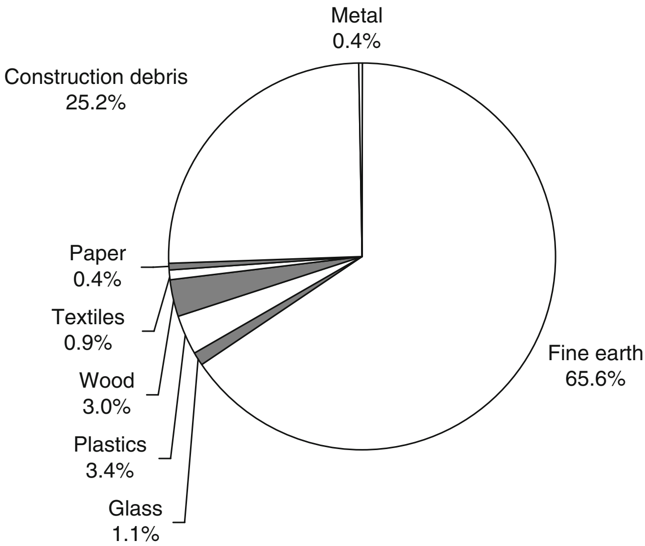


**Fig. 5.10** Composition of a waste heap in the city of Jind, India (Data unpublished [data from a current research project of the Haryana Agricultural University Hisar, India and the University of Applied Sciences Osnabrück, Germany])

inhabitants sweep away the waste around their residents on unpaved areas, so that a lot of loose soil material is combined with the waste particles.

Because of the high cost of mineral fertilizer in the capital of Burkina Faso, Ouagadougou, with a population of 1,400,000, the inhabitants used urban waste to improve soil conditions (increasing soil depth from 15–20 to 30–40 cm, increasing humus content from 0.9% to 2.5%, optimized potassium and phosphorus supply) for horticultural purposes (e.g. growing tomatoes, onions, lettuces, peppers); coarse elements such as bottles, plastic and batteries were not sorted, causing contamination of the degradable parts (see Section 4.3.2). Such materials represented 3.7–5.0% of the soil matrix. The main reason for the waste application was the presence of Leptosols covering 32% of the whole city and usually considered unsuitable for crop production. Because of the increasing population in the city the unfavourable soils were used for crop and particularly vegetable production. Farmers and citizens favoured the solution of amending urban waste stemming from markets, hospitals and other city areas (Thiombiano and Gnakambari 2000). Investigations undertaken in another African city, Dar es Salaam (Tanzania) with a population of approximately 3,000,000 mentioned the high plant uptake of contaminants by vegetables in suburban areas and in old dumpsites (Luilo 2000) (see Section 3.3.3).

Waste deposits are also found on urban land in many developed countries. For this reason many parks and even playgrounds were located on former landfills. Plants are negatively affected by anaerobically produced gas volatilization. The impacts remind one of waterlogged soils, since oxygen is strongly limited and nutrient uptake by roots seriously restricted (Craul 1992).

Present-day waste management prefers waste reduction, re-use, recycling and composting combined with energy recovery linked to incineration. Moreover, modernized landfill management is mostly connected with landfill gas utilization. Although a movement away from landfill is aimed at, actually in many countries of the developed world landfill is very common continuously. For instance, in the USA 57% of the waste was landfilled in 2000 and in countries in Eastern and South East Europe even higher values exist. In Bulgaria, Hungary, Lithuania, Poland, Romania and Turkey nearly 100% of the generated household waste was landfilled in 2003. In contrast, in some densely populated countries the percentage of landfilled waste was extremely low, such as Japan with approximately 6% and The Netherlands with less than 10%. In contrast to municipal solid waste, hazardous waste is recycled and recovered in developed countries to a smaller extent and the tendencies differ from country to country.

In comparison with less developed countries considerably more money is spent on waste collection and treatment in cities of developed countries. The solid waste expenditure per inhabitant was \$97 in 1994/1995 in New York, USA, \$63 in Strasbourg, France, \$48 in Toronto, Canada and \$38 in Sydney, Australia. In the same period of time the expenditure amounted to only \$3.92 in Mumbai, India, \$2.00 in Hanoi, Vietnam, \$1.46 in Dhaka, Bangladesh and \$0.66 in Accra, Ghana. In East European cities it amounted to \$13.80 (Budapest, Hungary), \$6.00 (Riga, Latvia), and \$2.37 (Bucharest, Romania) (Williams 2005).

The main reason for these tendencies might be the social circumstances existing in cities of developed countries compared with cities of less developed countries. Higher income and higher consumption lead automatically to more waste generation and consequently to the necessity for intensified waste management. In the developing countries with rapidly increasing urbanization in line with the trend of a movement from rural areas to urban areas (see Section 2.3) similar tendencies might be expected in the near future. Consequently, beneficial waste management is necessary in the short term to avoid an uncontrolled increase in landfill sites in the urban environment.

In general, the population size of a local community seems to be of importance in relation to waste generation. In Wisconsin, USA, it has been found that villages with a population of less than 2,500 people produce 0.91 kg day$^{-1}$ capita$^{-1}$, whilst towns counting between 10,000 and 30,000 people reached 1.45 day$^{-1}$ capita$^{-1}$ and cities counting more than 30,000 inhabitants reached even 1.63 kg day$^{-1}$ capita$^{-1}$ (Williams 2005).

In the European Union various directives and regulations attempt to include the member states in respect of waste flows and treatment as well as disposal. One important point for consideration is the development of a household and hazardous waste catalogue as a unified document for clear definition purposes. In the Commission Decision 2000/532/EC published in 2000 all sorts of waste were defined and named. For each category of waste a six-digit code was allocated, more than 650 categories were already listed and these will be updated progressively (Commission Decision 2000).

Another directive in the European Community refers to the treatment of disposed of waste in order to reduce waste volume significantly. The treatment involves physical processes like sorting. In particular, the biodegradable waste has to be removed before landfilling. Since this portion is responsible for the development of landfill gases such as methane and carbon dioxide, the sorting process is mainly discussed in the context of global warming. To control the contribution of the reductive gases to the atmosphere, it is planned to reduce the biodegradable percentage to 50% in 2009 and to 35% in 2016 related to the volume measured in 1995 (Council Directive 1999).

The disposal in landfills is associated with environmental disadvantages. Depositing of waste has to be feasible; attention must be paid to emissions of gases, explosion risk, effects on vegetation and discharge of leachates. The site operations are of importance because of site traffic, noise and dust emissions as well as visual impacts.

Landfill sites undergo distinct decomposition stages in the course of time (Fig. 5.11). In the first relatively short stage hydrolysis and aerobic degradation take place, leading to emission of carbon dioxide. Afterwards, in the second stage, hydrolysis is dominant, leading to the development of organic acids and a radical decrease in the pH value within the waste body. Accordingly, the mobility of heavy metals tends to rise during this stage. In the following third and fourth stages called methanogenesis the highest percentage of soil vapour is methane, while there is a tendency for the carbon dioxide concentration to decrease. The fourth stage lasts the longest period, normally stretching over several decades. Theoretically, after more or less complete degradation in reductive conditions, a fifth and last oxidative stage can occur. The more inert the composition of the waste body, the lower the methane concentration in soil vapour. In dumps consisting mainly of inert construction debris low $CH_4$ values were measured, in some cases tending towards zero, but typical values published for household waste landfills are of a higher order (average concentration 63.8 vol% $CH_4$) (Williams 2005).

| **Time** | | | | | |
|---|---|---|---|---|---|
| Stage | Aerobic | Anaerobic | | | Aerobic |
| **Gases** | | | | | |
| Oxygen | **XXX** | -- | -- | -- | **XX** |
| Carbon dioxide | **X** | **XX** | **XXX** | **XX** | **X** |
| Methane | -- | **X** | **XXX** | **XX** | **X** |
| **Leachate** | | | | | |
| Acids | **X** | **XXX** | **XX** | **X** | **X** |
| Mobile metals | **X** | **XX** | **XXX** | **X** | **X** |

-- not existent
X low concentration
XX moderate concentration
XXX high concentration

**Fig. 5.11** Landfill gases and leachate composition in relation to the degradation stages of municipal solid waste

Gas formation has a negative impact on the vegetation near dumpsites. A dumpsite in Berlin, Germany, with a height of 93 m and containing approximately 11.5 million cubic metres of household refuse was investigated in more detail. The site was partly insufficiently covered by quarry material in combination with construction rubble (thickness 30–50 cm only). It was possible for the methane gas to migrate through the cover and additionally through the sides with missing barrier systems. Young trees and shrubs were damaged at the site in spite of favourable nutrient conditions up to a distance of 50 m from the dump. The plant damage was predominantly caused by the reductive gas but, in addition, enhanced soil temperature due to the microbial decomposition of the garbage might also play a role. Yearly average temperatures of 10–15°C and periodically 45°C at a depth of 30 cm as well as up to 88°C in deeper layers can be expected (Blume 1986).

Similarly, attention must be paid to the leachate released downwards from the dumpsites, because enhanced contaminant values of the waste materials are generally expected (see Section 4.3.2). Between 13 and 35 samples from large landfill sites in different European countries were investigated, indicating high heterogeneity. Most of the parameters involved revealed higher concentrations in household landfills than in landfills consisting predominantly of inert construction debris (Table 5.12). It should be noted that the concentrations changed with time depending on the stage of decomposition (Williams 2005).

Old landfills, particularly in less developed and developing countries represent the majority of all landfills. Little is known about the material of input, no protection from gas emission and leachate occurred and no-one was informed about the hydrogeological conditions of the site before landfilling started. Moreover, old landfills are often sited in the vicinity of urban areas due to detrimental transport capacity. Accordingly, public authorities are not able to advise about natural attenuation of such landfills.

The waste deposits are often connected with the presence of toxic substances. In German cities some urban soils located on former landfills were analyzed and the results obtained were enhanced concentrations of Polycylic Aromatic Hydrocarbons (PAH). The subsoil values of a landfill site in Eckernförde, Germany, with a population of 23,000 without any influence from atmospheric input ranged from 3.6 to 34.3 mg kg$^{-1}$ $PAH_{EPA}$. If industrial residues are part of waste deposits, considerably higher values can be expected. In Stuttgart (Germany) with 600,000 inhabitants, for example, waste deposits in the subsoil of a former coking plant site reached concentrations up to 142 mg kg$^{-1}$ $PAH_{EPA}$ (Blume and Schleuss 1997).

In the following a dumpsite soil located in the Osnabrück district, Germany is introduced in detail by way of example (Table 5.13, Fig. 5.12). Here, weather-resistant, hard limestone of the Lower Middle Triassic was quarried from the end of the previous century up to the 1960s. In the 1970s and 1980s, the quarry was refilled with different substrates, including excavated earth and building debris, but above all with household and industrial waste. The refilled volume amounted to approximately 65,000 m$^3$ at a depth of 12–20 m. The distance between the base line of the deposit and the groundwater was at least 15 m. The surface was covered with approximately 30 cm of mineral soil. Until 1995, the area was used agriculturally

**Table 5.12** Composition of acetogenic and methanogenic leachate from large landfill sites with high waste input and relatively dry environment and from inert waste landfills (mg $L^{-1}$) (Data from different sources, cited in Williams 2005)

| | Household landfills | | | | Inert landfills | |
|---|---|---|---|---|---|---|
| | Acetogenic stage | | Methanogenic stage | | | |
| Parameter | Mean | Range | Mean | Range | Mean | Range |
| pH value | **6.7** | 5.1–7.8 | **7.5** | 6.8–8.2 | **8.1** | 7.7–8.8 |
| EC (dS $m^{-1}$) | **16.92** | 5.8–52.0 | **11.50** | 6.0–19.3 | **–** | – |
| DOC | **36,817** | 2,740–152,000 | **2,307** | 622–8,000 | **236** | 85–600 |
| Ammonia | **922** | 194–3,610 | **889** | 283–2,040 | **28.2** | 0.4–95 |
| Nitrate | **1.8** | <0.2–18.0 | **0.9** | 0.2–2.1 | **9.2** | <0.1–52 |
| Chloride | **1,805** | 659–4,670 | **2,074** | 570–4,710 | **372.5** | 32–1,700 |
| Sulphate | **676** | <5–1,560 | **67** | <5–322 | **211.8** | 51–330 |
| Arsenic | **0.024** | <0.001–0.148 | **0.034** | <0.001–0.485 | **–** | – |
| Cadmium | **0.02** | <0.01–0.10 | **0.015** | <0.01–0.08 | **–** | – |
| Chromium | **0.13** | <0.03–0.3 | **0.09** | <0.03–0.56 | **–** | – |
| Copper | **0.13** | 0.020–1.10 | **0.17** | <0.02–0.62 | **0.2** | <0.1–0.5 |
| Lead | **0.28** | <0.04–0.65 | **0.20** | <0.04–1.9 | **0.2** | <0.1–0.4 |
| Mercury | **0.0004** | <0.0001–0.0015 | **0.0002** | <0.0001–0.0008 | **–** | – |
| Nickel | **0.42** | <0.03–1.87 | **0.17** | <0.03–0.6 | **–** | – |
| Zinc | **17.37** | 0.09–140 | **1.14** | 0.03–6.7 | **0.6** | <0.1–2.8 |

DOC = dissolved organic carbon; EC = electrical conductivity; TOC = total organic carbon

**Table 5.13** Profile description of a typical landfill soil (Data from Meuser and Blume 2001; Meuser 2006)

| | |
|---|---|
| Site | Wallenhorst (Osnabrück district), Germany |
| Relief | Midslope |
| Land use | Meadow (unused) |
| Vegetation | Vegetation of pasture |
| Cut and fill | Anthropogenic fill/ landfill |
| Groundwater level | Extremely deep (>20 dm) |
| Depth (cm) | Description |
| 0–10 | Highly sandy loam, low skeletal content, slightly humic, 10YR 4/4, low compactness, sub-angular blocky structure, very strong root penetration, carbonate-free, root and earthworm tubes, distinct transition<br>Additions: synthetic material (isolated finding) |
| 10–30 | Highly loamy sand, low skeletal content, humus-free, 10 YR 5/8, mean compactness, sub-angular blocky and platy structure, very strong root penetration, carbonate-free, root and earthworm tubes, reduction-oxidation (light rust patches), distinct transition<br>Additions: none |
| 30–60 | Medium silty sand, high skeletal content, medium humic, 10YR 3/4, granular structure, strong root penetration, medium calcareous, root and earthworm tubes, reduction-oxidation (dark rust colours), distinct transition<br>Additions: brick (3–15%), carbonaceous shale (isolated finding), glass (bottles), synthetic material and metal (each 3–15%), ceramics, textiles, paper (isolated findings) |
| 60–75 | Medium sandy loam, medium skeletal content, slightly humic, 10YR 4/2, high compactness, sub-angular blocky structure, poor root penetration, low carbonate content, macropores, reduction-oxidation (rust colouring, bleaching), distinct transition<br>Additions: glass, synthetic material, metal, furnace release (each isolated findings) |
| 75–90 | Medium loamy sand, high skeletal content, medium humic, 5YR 3/3, single grain structure, very poor root penetration, high carbonate content, reduction-oxidation (dark rust colours), fouling gas odor, diffuse transition<br>Additions: silicate of iron slag, melting chamber granulates, furnace release/bottom ash (each 3–15%), synthetic material, ceramics, glass (each isolated findings), paper (3–15%), medium to high technogenic C-content |
| 90–140$^{+}$ | Medium loamy sand, high skeletal content, N 2, low compactness, single grain structure, without roots, high content of carbonate, reduction-oxidation (black), intensive odor of $H_2S$ and fouling gas<br>Additions: brick, furnace release/bottom ash, glass, synthetic material (each 3–15%), ceramics, paper (each isolated findings), medium technogenic C-content |
| Type of soil (WRB) | Technosol |
| Parent material | Loam fill over household and industrial waste |

**Fig. 5.12** Landfill soil profile in the district |of Osnabrück, Germany (One measuring staff segment = 10 cm)

(meadow, pasture ground for sheep). Guiding investigations showed highly reductive conditions in the garbage mass (Eh = −160 to −250 mV). It was also possible to determine reduction gases in near-surface strata. Enhanced methane levels (to 12 vol%) and very low oxygen levels (<1 vol%) occurred in the soil air. The contamination of soil at the site (soil air levels, heavy metal contents) forced the authorities to stop the area being used for agriculture. The site is situated in a predominantly agricultural catchment area. The slightly sloping area that was once used as grassland is no longer used for agriculture.

From a depth of 30 cm, massive garbage depositions showing a wide spectrum of garbage components but also building debris, slag and ashes can be detected in the soil. In particular, from a depth of 75 cm evidence of reduction predominated; increased organic waste components were discovered here, the anaerobic decomposition of which induced strong odor generation. At a depth of 60–75 cm there was an intermediate covering of mineral soil which, despite containing no considerable additions, still showed strong reductive features. The garbage deposit was covered with uncontaminated mineral soil. In the meantime, a 10 cm thick Ah horizon has been able to establish itself.

The most significant results of the analyses were as follows (Table 5.14):

- The horizons containing garbage components showed a high skeletal content, while surface and intermediate coverings had distinctly lower skeletal contents.

**Table 5.14** Physical and chemical properties of the landfill soil (Data from Meuser and Blume 2001; Meuser 2006)

| Depth (cm) | Skeleton (mass %) | Sand (mass %) | Silt (mass %) | Clay (mass %) | Bulk density (g cm$^{-3}$) | | |
|---|---|---|---|---|---|---|---|
| 0–10 | 17 | 67 | 21 | 12 | 1.61 | | |
| 10–30 | 21 | 68 | 19 | 13 | 1.85 | | |
| 30–60 | 62 | 67 | 25 | 8 | 1.24 | | |
| 60–75 | 38 | 63 | 27 | 10 | – | | |
| 75–90 | 53 | 61 | 30 | 9 | 0.81 | | |
| 90–140$^{+}$ | 51 | 66 | 26 | 8 | – | | |
| **Depth** | **TPV** | **vol% Water at pF** | | | | | |
| **(cm)** | **(vol%)** | **1.0** | **1.8** | **2.5** | **3.0** | **3.7** | **4.2** |
| 0–10 | 39 | 35 | 30 | 23 | 20 | 11 | 10 |
| 10–30 | 34 | 31 | 27 | 20 | 18 | 11 | 7 |
| 30–60 | 52 | 36 | 31 | 26 | 23 | 11 | 10 |
| 60–75 | – | – | – | – | – | – | – |
| 75–90 | 65 | 42 | 36 | 30 | 28 | 10 | 9 |
| 90–140$^{+}$ | – | – | – | – | – | – | – |
| **Depth (cm)** | **$C_t$ (mass %)** | **$C_{humus}$ (mass %)** | **$C_{technogenic}$ (mass %)** | **$N_{org.}$ (mass %)** | **C/N** | **$CaCO_3$ (mass %)** | |
| 0–10 | 1.5 | 1.5 | 0.0 | 0.135 | 11 | 0.0 | |
| 10–30 | 0.2 | 0.2 | 0.0 | 0.020 | 10 | 0.0 | |
| 30–60 | 5.3 | 2.6 | 2.7 | 0.256 | 21 | 1.6 | |
| 60–75 | 2.0 | 1.1 | 0.9 | 0.103 | 19 | 0.1 | |
| 75–90 | 14.0 | 2.7 | 11.2 | 0.389 | 36 | 3.9 | |
| 90–140$^{+}$ | 15.1 | 2.9 | 12.0 | 0.445 | 34 | 6.4 | |

(continued)

**Table 5.14** (continued)

| Depth (cm) | pH $CaCl_2$ | pH $H_2O$ | $P\text{-}H_2O$ (mg $100g^{-1}$) | EC (dS $m^{-1}$) |
|---|---|---|---|---|
| 0–10 | 5.9 | 6.4 | 1.2 | 0.10 |
| 10–30 | 5.4 | 6.1 | 0.1 | 0.04 |
| 30–60 | 7.3 | 7.6 | 0.2 | 0.30 |
| 60–75 | 7.3 | 7.7 | 0.4 | 0.17 |
| 75–90 | 7.3 | 7.5 | 0.1 | 0.59 |
| 90–140$^+$ | 7.3 | 7.4 | nd | 1.37 |

| Depth | Exchangeable bases (mmol$_c$ kg$^{-1}$) | | | | |
|---|---|---|---|---|---|
| (cm) | Ca | Mg | K | Na | $\Sigma PAH_{EPA}$ (mg kg$^{-1}$) |
| 0–10 | 66.5 | 11.1 | 4.1 | 0.2 | 0.8 |
| 10–30 | 39.4 | 8.5 | 2.8 | 0.2 | nd |
| 30–60 | 236.8 | 7.4 | 1.9 | 0.6 | 24.0 |
| 60–75 | 111.5 | 4.6 | 2.6 | 0.3 | 5.1 |
| 75–90 | 261.4 | 25.5 | 4.4 | 1.0 | 6.2 |
| 90–140$^+$ | 313.6 | 22.2 | 4.4 | 0.7 | 7.1 |

| Depth | Cd | Cr | Cu | Ni | Pb | Zn |
|---|---|---|---|---|---|---|
| (cm) | (mg kg$^{-1}$) | | | | | |
| 0–10 | 0.2 | 58 | 16 | 22 | 68 | 88 |
| 10–30 | 0.1 | 34 | 9 | 19 | 13 | 44 |
| 30–60 | 1.6 | 62 | 302 | 47 | 449 | 855 |
| 60–75 | 0.6 | 40 | 48 | 21 | 152 | 220 |
| 75–90 | 0.8 | 122 | 522 | 124 | 446 | 1,120 |
| 90–140$^+$ | 1.6 | 72 | 299 | 120 | 1,120 | 811 |

nd = not detectable
PAH = polycyclic aromatic hydrocarbons; TPV = total pore volume
Metal extraction method: aqua regia

- The available field capacity down to 90 cm reached a medium level; the covering showed middle air capacity, the deposit body high air capacity caused by the skeleton content.
- The garbage deposits contained very high nitrogen content which, despite the high level of a technogenic carbon, did not allow very wide C/N-ratios.
- The garbage mass was weakly calcareous to calcareous, while free carbonates were lacking in surface and intermediate coverings; the pH values lay in the weakly alkaline range, which is typical for a mass of garbage, and above that they were of medium acidity.
- Below a depth of 30 cm, a distinct increase in the contaminant values was also determined. The increased contaminant contents (PAH, Cr, Cu, Ni, Pb and Zn) were linked to the technogenic substrates discovered in some of the horizons, since the values in surface and intermediate coverings were distinctly lower (Meuser and Blume 2001; Meuser 2006).

### 5.4.3 *Soils of Industrial Deposits*

In the proximity of industrial plants technogenic monosubstrates generated during industrial processes are deposited in dry or moist conditions. Characteristics of slag and ash deposits from German industry sites are introduced in the following chapter.

Firstly, the characteristics of a typical slag heap will be outlined (Table 5.15, Fig. 5.13). The site belonged to a range of hills, a geological body that projected out of its surroundings, the uplift of which took place during the Upper Cretaceous and Tertiary periods. The area called Hüggel is, geologically speaking, particularly

**Table 5.15** Profile description of a typical slag heap soil (Data from Meuser and Blume 2001; Meuser 2006)

| | |
|---|---|
| Site | Hasbergen (Osnabrück district), Germany |
| Relief | Culmination |
| Land use | Forest |
| Vegetation | Primary vegetation (*Geranium robertianum*, mosses) |
| Cut and fill | Anthropogenic landfill |
| Groundwater level | Extremely deep (>20 dm) |
| Depth (cm) | Description |
| 0–2 | Coarse-grained sand, very high skeletal content, medium humic, 7.5YR 4/2, very low to low compactness, single grain structure, very strong root penetration, very high content of carbonate, root tubes, macropores, distinct transition |
| | Additions: metal works slag (60–85%) |
| $2–140^{+}$ | No fine soil, skeletal soil, humus-free, N 6, poor root penetration (depth 2–20 cm)/without roots, very high content of carbonate, macropores, intra-macropores (slag) |
| | Additions: metal works slag (monosubstrate) |
| Type of soil (WRB) | Technosol |
| Parent material | Metal works slag |

**Fig. 5.13** Slag heap soil profile in the district of Osnabrück, Germany (One measuring staff segment = 10 cm)

diverse. Close to the surface rock formations from the Coal Formation, Permian (Upper Permian), Triassic (Lower Triassic, Middle Triassic), Cretaceous and Quaternary periods were visible. The Upper Permian limestone deposits, in particular, contained iron ores and other significant mineral occurrences (sphalerite, galena, baryte). The high calcite contents also discovered at the site led to the suggestion of introducing iron ore mining and the iron and steel industry to the region. Ironworks, heavy metal works and steelworks were established locally. The slag deposits at the investigated site can probably be traced back to the period around 1860, when metal was also smelted close to the ore open-cast mining areas in smaller blast furnaces (Meuser and Blume 2001).

The soil profile contained a deposit of metal works slag which did not undergo filling at this location. The overlying soil increased with distance from the profile, where anthropogenically filled mineral layers of soil and pedogenically formed thin humic surface soil horizons existed on top, which even permitted reforesting (mainly with beeches). At the investigated site, however, despite the long development period, an initial A horizon of just a few centimetres' thickness has formed. Underneath that there was slightly weathered material of porous, highly carbonate-rich metal works slag.

The most significant results of the analyses were as follows (Table 5.16):

- The site consisted almost exclusively of skeleton; skeletal material therefore had to be flocculated and crushed for the fine soil analysis; a texture analysis was not possible.
- The slag material contained a high air capacity derived from the substrate itself.

**Table 5.16** Physical and chemical properties of the slag heap soil (Data from Meuser and Blume 2001; Meuser 2006)

| Depth (cm) | Skeleton (mass %) | Sand (mass %) | Silt (mass %) | Clay (mass %) | Bulk density (g $cm^{-3}$) | | |
|---|---|---|---|---|---|---|---|
| 0–2 | 83 | – | – | – | 1.11 | | |
| 2–140+ | 95 | – | – | – | nd | | |
| | | **vol% Water at pF** | | | | | |
| **Depth (cm)** | **TPV (vol%)** | **1.0** | **1.8** | **2.5** | **3.0** | **3.7** | **4.2** |
| 0–2 | 58 | 38 | 25 | 22 | 19 | 13 | 9 |
| 2–140+ | – | – | – | – | – | – | – |
| **Depth (cm)** | **$C_t$ (mass %)** | **$C_{humus}$ (mass %)** | **$C_{technogen}$ (mass %)** | **$N_{org.}$ (mass %)** | **C/N** | **$CaCO_3$ (mass %)** | |
| 0–2 | 2.5 | 1.6 | 0.9 | 0.028 | 89 | 7.6 | |
| 2–140+ | 2.2 | 1.4 | 0.8 | 0.015 | 147 | 6.8 | |
| **Depth (cm)** | **pH $CaCl_2$** | **pH $H_2O$** | **P-$H_2O$ (mg $100g^{-1}$)** | **EC (dS $m^{-1}$)** | | | |
| 0–2 | 11.3 | 11.4 | 0.1 | 0.49 | | | |
| 2–140+ | 11.4 | 11.5 | 0.1 | 0.78 | | | |
| | **Exchangeable bases ($mmol_c$ $kg^{-1}$)** | | | | | | |
| **Depth (cm)** | **Ca** | **Mg** | **K** | **Na** | **$\Sigma PAH_{EPA}$ (mg $kg^{-1}$)** | | |
| 0–2 | 952 | 30.2 | 0.9 | 1,5 | 0.5 | | |
| 2–140+ | 1,430 | 46.2 | 3.1 | 0.7 | nd | | |
| **Depth (cm)** | **Cd (mg $kg^{-1}$)** | **Cr** | **Cu** | **Ni** | **Pb** | **Zn** | |
| 0–2 | nd | 2,570 | 178 | 116 | 43 | 62 | |
| 2–140+ | 0.4 | 2,840 | 173 | 47 | 531 | 80 | |

nd = not detectable
PAH = polycyclic aromatic hydrocarbons; TPV = total pore volume
Metal extraction method: aqua regia

- A slight accumulation of humus was revealed, possibly because of roots also discovered at depths >2 cm; however, the measured carbon content was partially due to the artificially added (technogenic) carbon from the smelting technology.
- The pH values were in the extremely alkaline range; the material had a high content of calcium carbonate; therefore the material consisted almost exclusively of Ca ions.
- With regard to nitrogen and cationic nutrients, the site had to be designated as very low in nutrients.
- The metal works slag not only showed very high chromium content, but also increased Cu and Pb content, causing the site to be classified as contaminated by heavy metals (Meuser and Blume 2001; Meuser 2006).

Another technogenic monosubstrate of interest refers to the main component group ash. Fly ash generated by filter systems of coal-fired power stations was often deposited in wet conditions by pumping the fly ash/water suspension into lagoons that are surrounded by dams. After slow sedimentation of the sludge the lagoon started drying, in particular in warmer climates, where the evapotranspiration is high in comparison or higher than the annual precipitation. Furthermore, dry deposits took place simultaneously.

Fly ash lagoons based on lignite coal were intensively investigated near the German cities of Halle with its 230,000 inhabitants and Leipzig with its 500,000 inhabitants. The physical and chemical properties exhibited partly extreme results. The bulk density lay under 1.0 g $cm^{-3}$, the pH value varied between 6.4 and 8.6 indicating neutral to alkaline values and a high gypsum (2.3–19.0%) as well as calcium carbonate (0.7–9.8%) content was found. Gypsum ($CaSO_4 * 2H_2O$) and calcium carbonate ($CaCO_3$) caused cementation of the sludge field, which led to enormous penetration resistance during dry periods. It was hardly possible to dig the lagoon in warmer summer periods (Zikeli et al. 2002).

The carbon content revealed very high concentrations related to the technogenic percentage of the material. As seen in Table 5.17, the C content reached enhanced values in the subsoil as well independent of the kind of deposit. Especially the fine

**Table 5.17** Soil properties of lignite ash sites near Halle, Germany (Data from Zikeli et al. 2002; 2004; Machulla et al. 2004)

| | Depth | C total | $CEC_{POT}$ | Basal respiration | Cd | Cu |
|---|---|---|---|---|---|---|
| | (cm) | (%) | ($cmol_c$ $kg^{-1}$) | (µg $CO_2$-C*$g^{-1}$*$h^{-1}$) | (mg $kg^{-1}$) | (mg $kg^{-1}$) |
| Landfill (23 year old) | 0–30 | 16.7–21.9 | 86–118 | 6.5–9.7 | 1–23 | 161–224 |
| | 30–100 | 12.4–16.6 | 82–89 | 6.4–9.9 | – | 16 |
| Lagoon (19 year old) | 0–15 | 7.2–14.9 | 32–52 | 1.0–3.9 | 0.5–20 | 73–100 |
| – inflow (coarse material) | 25–100 | 2.2–9.8 | 14–29 | 1.9–4.4 | 0.2–0.3 | 51–65 |
| Lagoon (19 year old) – fine material | 0–30 | 10.8–38.6 | 51–95 | 5.9–7.6 | 1–14 | 64–86 |
| | 30–100 | 6.3–24.1 | 32–47 | 0.1–5.2 | 2–5 | 66–72 |

CEC = cation exchange capacity

material sedimented far away from the inflow of the lagoon consisted of coal residues and soot which were responsible for the technogenic carbon. Looking at the cation exchange capacity (CEC) this parameter also showed higher results, demonstrating increasing CEC with the accumulation of technogenic carbon content. Obviously, the fly ash seemed to be biologically active, since the basal respiration, a good indicator for microbiological activity, showed parallel results. In comparison with forest topsoils (0.1–1.1 μg $CO_2$-C $g^{-1}$ $h^{-1}$) and topsoils of agroecosystems (0.6–6.0 μg $CO_2$-C $g^{-1}$ $h^{-1}$) the measured values of the fly ash deposits indicated a generally high biological activity. The heavy metal content did not disturb the activity of bacteria and fungi mentioned, although the copper results usually influencing biological abundance ranged between 73 and 224 mg $kg^{-1}$ in the upper part of the profiles (see Section 4.3.2). This context is transferable to the second element of concern, cadmium, which is presented in the same publications (Zikeli et al. 2004; Machulla et al. 2004).

Ash deposits located in urban land obviously did not tend towards raised concentrations of Polycyclic Aromatic Hydrocarbons (PAH). In Halle, Germany two ash deposits were analyzed, resulting in subsoil concentrations between 0.1 and 0.3 mg $kg^{-1}$ $PAH_{EPA}$ (mixture of fly and bottom ash) as well as 0.1 and 0.5 mg $kg^{-1}$ $PAH_{EPA}$ (bottom ash), respectively (Blume and Schleuss 1997). The generation of PAH is closely connected to incomplete burning of organic material or coal. Ashes derived from coal power stations, in particular high temperature coal power stations, have not been inevitably contaminated with PAH.

The suitability of ashes for soil fertilizing is well-known (see Section 5.3.2). Pulford (1991) mentioned a total nitrogen content of 350 mg $kg^{-1}$ in pulverized fuel ash. Relatively large amounts of phosphorus were detectable in fuel ash deposits as well. Values of up to 630 mg $kg^{-1}$ for total P and 94–144 mg $kg^{-1}$ for available P were described.

Coarse bottom ash is frequently used to prepare wetland for construction purposes. In order to construct buildings in dry conditions the investigated wetland site in the German district of Osnabrück which has been presented was filled with different materials, in particular ashes (Table 5.18, Fig. 5.14). In the 1960s, it was used for horticultural purposes. It can be seen that, particularly in the gardens, ashes were applied, and, to a lesser extent, household garbage was used for fertilization as well.

The site was fallow land with nitrophile ruderal vegetation. Due to its previous horticultural use there is an approximately 30 cm thick A horizon that was low in constituent minerals. Underneath that, there were two different morphological and substrate-specific horizons; while domestic waste components were recognizable in the upper area of the profile (including the remains of a toilet), the lower horizon was composed exclusively of a mixture of bottom ashes and furnace release. From a depth of 30 cm, the profile portrayed redoximorphic features that, with increasing depth, showed increasingly visible anaerobic conditions (black colouring, $H_2S$ odor). The groundwater level fluctuated between a depth of 75–130 cm. The additive-free horizon filled with fossils and groundwater began at a depth of 110 cm. Between this and the deposited layers there was a mixed horizon, which was formed by the initial deposit of bottom ashes and furnace release (drainage) onto the waterlogged, possibly swampy surface.

**Table 5.18** Profile description of an ash containing urban soil in a built-up area (Data from Meuser and Blume 2001; Meuser 2006)

| | |
|---|---|
| Site | Osnabrück, Germany |
| Relief | Plane |
| Land use | Ruderal area |
| Vegetation | Nitrophile montane perennial herbs and shrubs (*Sambucus, Rubus*), fruit trees that have grown wild |
| Cut and fill | Anthropogenic fill/landfill |
| Groundwater level | Deep (8–13 dm) |
| Depth (cm) | Description |
| 0–30 | Sand, skeleton-free, slight to medium humic, 7.5YR 3/2, high compactness, single grain structure, strong root penetration, high content of carbonate, distinct transition |
| | Additions: brick, glass, ceramics, synthetic material, bones, charcoal (each isolated findings) |
| 30–55 | Sand, low skeletal content, slightly humic, 7.5 YR 3/3, mean to high compactness, single grain structure, poor root penetration, very high content of carbonate, macropores, reduction-oxidation (dark rust colours), diffuse transition |
| | Additions: bottom ash/furnace release, coal, ceramics, glass (each 3–15%), medium technogenic C-content |
| 55–75 | Sand, medium skeletal content, humus-free, 7.5YR 2.5/3, high compactness, single grain structure, strong root penetration, very high content of carbonate, macropores, reduction-oxidation (dark rust colours), $H_2S$ odor, distinct transition |
| | Additions: bottom ash/furnace release (monosubstrate), high technogenic C-content |
| 75–110 | Silt, medium skeletal content, medium humic, 10YR 2/2, mean compactness, sub-angular blocky structure, without roots, medium content of carbonate, reduction-oxidation (rust colouring, black), $H_2S$ odor, diffuse transition |
| | Additions: bottom ash/furnace release (60–85%) |
| 110–140$^+$ | Silty sand, skeleton-free, very strongly humic, 7.5YR 2.5/1, low to medium compactness, sub-angular blocky structure, without roots, carbonate-free, reduction-oxidation (black), $H_2S$ odor |
| | Additions: none |
| Type of soil (WRB) | Technosol |
| Parent material | Mixture of soil, ashes and garbage over humic alluvial deposits |

The most significant results of the analyses were as follows (Table 5.19):

- Skeletal contents and consequently air capacities were high until the fossil gleyic horizon in 110 cm depth was reached; nevertheless high to very high available field capacity was recognizable.
- The total pore volume was very high and bulk density low due to the almost moor-like substrate in the gleyic and peaty horizon; the bulk density of the deposited area above was also only approximately 1.0 g/cm$^3$.
- The carbon contents in the deposited horizons were almost exclusively due to the technogenic C-levels (bottom ash, furnace release); in the peaty subsoil, the

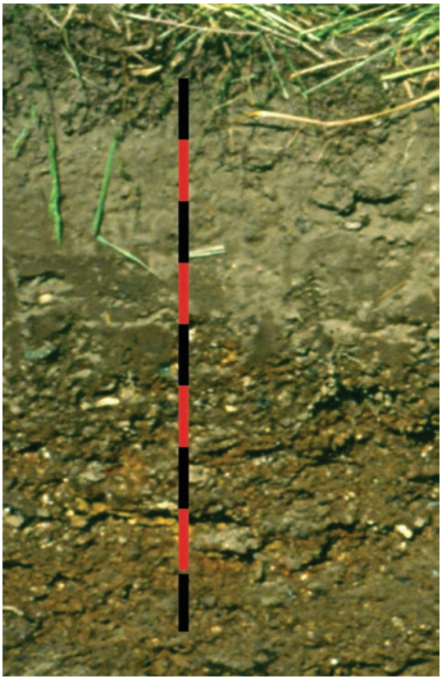

**Fig. 5.14** Bottom ash containing urban soil in a built-up area in Osnabrück, Germany (One measuring staff segment = 10 cm)

opposite conditions prevailed; here the humic carbon was dominant; as expected, the increased technogenic carbon content was linked to a wide C/N-ratio.

- The site was characterized by relatively high nitrogen level and the soil was conspicuously well supplied in the upper A horizon (horticultural use, fertilization).
- The pH values lay in the weakly alkaline range, as expected for deposited areas with ashes and garbage additions.
- The soil revealed an increased level of Polycyclic Aromatic Hydrocarbons.
- The site was also contaminated by heavy metals; at a depth of 30–110 cm, greatly increased values for Cd, Cu and Pb were determined. These can be partly explained by the large amount of furnace release (Meuser and Blume 2001; Meuser 2006).

## *5.4.4 Mining Soils*

### 5.4.4.1 Coal Mining

Waste from coal mines is usually deposited in heaps close to the shafts where coal is extracted (see Section 3.5.3). Extreme amounts of coal mining waste have to be deposited in coal extracting countries, changing the landscape to great extent.

**Table 5.19** Physical and chemical properties of an urban soil containing ash (Data from Meuser and Blume 2001; Meuser 2006)

| Depth (cm) | Skeleton (mass %) | Sand (mass %) | Silt (mass %) | Clay (mass %) | Bulk density (g $cm^{-3}$) | | |
|---|---|---|---|---|---|---|---|
| 0–30 | 18 | 74 | 20 | 7 | 1.40 | | |
| 30–55 | 63 | 76 | 16 | 8 | – | | |
| 55–75 | 55 | 65 | 30 | 6 | 1.01 | | |
| 75–110 | 58 | 69 | 27 | 4 | – | | |
| $110–140^{+}$ | 2 | – | – | – | 0.26 | | |
| | | **vol% Water at pF** | | | | | |
| **Depth (cm)** | **TPV (vol%)** | **1.0** | **1.8** | **2.5** | **3.0** | **3.7** | **4.2** |
| 0–30 | 46 | 41 | 27 | 16 | 15 | 8 | 6 |
| 30–55 | – | – | – | – | – | – | – |
| 55–75 | 57 | 43 | 35 | 22 | 16 | 8 | 6 |
| 75–110 | – | – | – | – | – | – | – |
| $110–140^{+}$ | 85 | 82 | 79 | 69 | 60 | 16 | 16 |
| **Depth (cm)** | **$C_t$ (mass %)** | **$C_{humus}$ (mass %)** | **$C_{technogenic}$ (mass %)** | **$N_{org.}$ (mass %)** | **C/N** | **$CaCO_3$ (mass %)** | |
| 0–30 | 4.8 | 2.6 | 2.2 | 0.30 | 16 | 0.3 | |
| 30–55 | 18.5 | 0.3 | 18.2 | 0.30 | 62 | 6.3 | |
| 55–75 | 19.1 | 0.4 | 18.7 | 0.22 | 87 | 4.5 | |
| 75–110 | 18.2 | 12.3 | 5.9 | 1.30 | 14 | 3.2 | |
| $110–140^{+}$ | 10.1 | 8.0 | 2.1 | 0.59 | 17 | 0.3 | |
| **Depth (cm)** | **pH $CaCl_2$** | **pH $H_2O$** | **P-$H_2O$ (mg $100g^{-1}$)** | **EC (dS $m^{-1}$)** | | | |
| 0–30 | 6.4 | 6.9 | 2.6 | 0.05 | | | |
| 30–55 | 7.4 | 7.9 | 0.2 | 0.10 | | | |
| 55–75 | 7.5 | 8.0 | 0.2 | 0.14 | | | |
| 75–110 | 7.1 | 7.3 | 0.6 | 0.21 | | | |
| $110–140^{+}$ | 5.9 | 6.1 | 0.5 | 0.34 | | | |

| | Exchangeable bases ($mmol_c\ kg^{-1}$) | | | | | |
|---|---|---|---|---|---|---|
| **Depth (cm)** | **Ca** | **Mg** | **K** | **Na** | **$\Sigma PAH_{EPA}$ ($mg\ kg^{-1}$)** | |
| 0–30 | 116 | 4.4 | 3.2 | 0.3 | 12.1 | |
| 30–55 | 266 | 5.5 | 2.5 | 0.6 | 5.8 | |
| 55–75 | 254 | 4.4 | 2.4 | 0.5 | 14.3 | |
| 75–110 | 491 | 2.1 | 1.7 | 0.7 | 0.4 | |
| 110–140$^{+}$ | 241 | 3.4 | 9.0 | 0.2 | 0.2 | |
| **Depth (cm)** | **Cd ($mg\ kg^{-1}$)** | **Cr** | **Cu** | **Ni** | **Pb** | **Zn** |
| 0–30 | 1.5 | 22 | 193 | 19 | 188 | 309 |
| 30–55 | 1.6 | 22 | 416 | 17 | 1,190 | 375 |
| 55–75 | 5.3 | 58 | 612 | 102 | 408 | 478 |
| 75–110 | 4.1 | 51 | 156 | 97 | 968 | 390 |
| 110–140$^{+}$ | – | – | – | – | – | – |

PAH = polycyclic aromatic hydrocarbons; TPV = total pore volume
Metal extraction method: aqua regia

In South Africa, for instance, approximately 330 million tonnes of coal mining waste are produced annually. In turn, household waste only amounts to about 15 million tonnes $a^{-1}$ and industrial waste to about 12 million tonnes $a^{-1}$ in the same country (University of Texas 2006).

Results from nutrient analyses have been listed by Pulford 1991. The total nitrogen content of fresh strip-mine soil ranged between 730 and 6,900 mg $kg^{-1}$ in US coal fields, between 1,000 and 8,000 mg $kg^{-1}$ in coal mine waste of Scotland (United Kingdom) and between 460 and 7,000 mg $kg^{-1}$ in Polish coalfields. The nitrogen supply for vegetation, however, has to be described as detrimental, since the nitrogen content is derived from coal residues and subsequently mainly inert. Legumes growing on coal mine waste may enhance the nitrogen content, and therefore they are often applied in mining heap rehabilitation. On colliery spoil white clover fixed 90–167 kg N $ha^{-1}$ $year^{-1}$. Other authors, cited in Bullock and Gregory 1991 stated values of 49 kg N $ha^{-1}$ $year^{-1}$ (white clover on china clay waste) and 72 kg N $ha^{-1}$ $year^{-1}$ (lupin on china clay waste). Reforested coal mine heaps exhibited 115 kg N $ha^{-1}$ $year^{-1}$ on sites on which the tree *Robinia pseudoacacia* was grown.

The phosphorus content of coal mine waste like china clay was generally deficient and much of the P was unavailable. The results were caused by the low pH value typical for coal mine waste. The phosphorus unavailability increased with decreasing pH. The relation can be explained by precipitation of iron and aluminium phosphates under acid conditions (Pulford 1991).

Pyrite containing coal mining waste tends towards strong acidification after depositing. Based upon oxidation of pyrite and creation of $H_2SO_4$, the pH value decreases rapidly and clearly (see Section 6.3.3). In Butyl, Brazil, with a population of 20,000, the deposited overburden consisting of kaolinitic sediments, smectitic shales and coal residues revealed pH values between 4.1 and 7.7 already 2 years after deposit, while the values in a 5-year-old heap ranged from 1.9 to 2.9 and in a 15-year-old heap from 2.8 and 4.9 (Kämpf 2000).

The site investigated in the German district of Osnabrück (Table 5.20, Fig. 5.15) was an island-like, tectonic up-warp from the Coal Formation period, with strata on top of it stemming from the Triassic, Jurassic and Cretaceous periods. The carbonate rock consisted of sandstones, shales and coal seams, which, due to the heating up of volcanic material that could not reach the surface, led to a high degree of coalification (anthracite coal). The valuable coal was already mined underground in the year 1727 and completed in 1871. The overburden material that was generated while sinking the neighbouring shaft building was dumped at this site, leading to a heap shaped like a flat-topped hill. Coal mining ended with the closure of the colliery in 1898. It can therefore be assumed that the site has remained untouched to a large extent since then. In the course of succession, it was possible for high-quality woodland vegetation to establish itself (Meuser and Blume 2001).

After the dumping of the coal mining waste finished a thick A horizon was able to establish itself on the flat-topped mount in the course of time, assisted by developing vegetation (succession flora). In addition to the humus formation,

**Table 5.20** Profile description of a typical coal mining heap (Data from Meuser and Blume 2001; Meuser 2006)

| | |
|---|---|
| Site | Osnabrück, Germany |
| Relief | Culmination |
| Land use | Coal mining waste heap |
| Vegetation | Forest (*Betula, Fraxinus*), rich in bushes (climax stage succession) |
| Cut and fill | Anthropogenic deposit/dump |
| Groundwater level | Extremely deep (>20 dm) |
| Depth (cm) | Description |
| 0–20 | Medium silty sand, medium skeletal content, medium humic, 10YR 3/1, low compactness, granular structure, strong root penetration, carbonate-free, earthworm and root tubes, sharp transition |
| | Additions: brick (3–15%), ceramics and glass (isolated findings), coal mining waste/burnt coal mining waste/carbonaceous shale (3–15%), high technogenic C-content |
| 20–30 | Medium loamy sand, very high skeletal content, N 3, very low compactness, strong root penetration, carbonate-free, root tubes, macropores, distinct transition |
| | Additions: coal mining waste (3–15%), carbonaceous shale and burnt coal mining waste (each 15–40%), high technogenic C-content |
| 30–120$^{+}$ | (Medium loamy sand), skeletal soil, very slightly humic, N 2 or 10YR 4/4, mean root penetration, carbonate-free, containing sulphide |
| | Additions: coal mining waste (3–15%), carbonaceous shale/coal (15–40%), burnt coal mining waste (40–60%), very high technogenic C-content |
| Type of soil (WRB) | Technosol/(Skeletic Umbrisol) |
| Parent material | Coal mining waste |

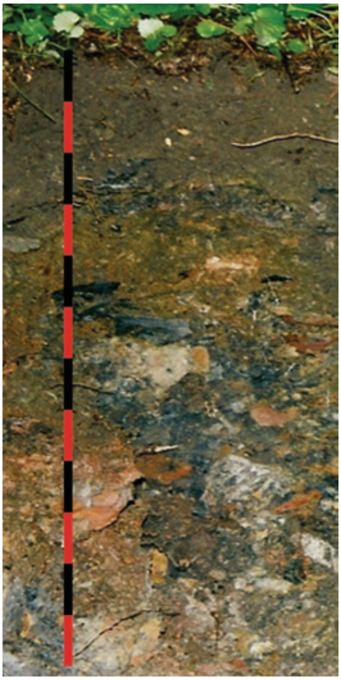

**Fig. 5.15** Soil profile of a coal mining heap in Osnabrück, Germany (One measuring staff segment = 10 cm)

weathering of the coal mining waste was a further visible feature of the pedogenesis; the mine dump material in the subsoil had already strongly weathered physically and chemically. Pyrite oxidation took place. Apart from the grey and coal-striated black coal mining waste, the profile also contained red-coloured burnt material, a product of exothermic pyrite oxidation that may have caused coal fires.

The most significant results of the analyses were as follows (Table 5.21):

- From a depth of 20 cm, the site had a substrate-related high skeletal content; the skeleton was petrographically composed of sandstone, siltstone as well as carbonaceous shale and anthracite coal.
- The high carbon content led to a high level of technogenic carbon which, on the other hand, despite the equally increased N-contents, induces a wide C/N-ratio; the total carbon content reached approximately 30% in the black coloured subsoil.
- The coal mining waste had acquired a relatively high cationic nutrient potential (K, Mg) that was connected with the petrography.
- As a consequence of pyrite weathering ($H_2SO_4$ formation), low pH values occurred.
- The industrially exploited surroundings of the site (bituminous tarworks nearby) led to an accumulation of Polycyclic Aromatic Hydrocarbons (PAH) via atmospheric input, as the depth gradient of the soil analyses showed.
- Coal mining waste from carbonic anthracite coal at this location showed a geogenic-related accumulation of lead, a phenomenon that has not been observed in other coal mining areas in Germany (Meuser and Blume 2001; Meuser 2006).

Coal mining waste was not only deposited in size limited and relatively high heaps, there are examples of widespread mining waste deposits in mining areas. For instance, Huainan mining area, one of the biggest coal bases in China with a 100-year mining history and an output of 100 million tonnes of coal annually, was most intensively used during the past 50 years, leading to a huge subsidence area of approximately 100 $km^2$ with a water surface of approximately 13 $km^2$ (see Section 6.1.6). Coal mining waste was used for reclamation purposes of the subsided land. The material was deposited, levelled and ultimately overlaid by humic soil in order to enable an agricultural re-use of the former devastated territory (Makowsky et al. 2009). Subsequently, large areas may consist of coal gangue in the Huainan area. Investigations of the used coal gangue showed generally relatively low heavy metal concentrations (Table 5.22). Compared with the values of the soils in the immediate vicinity of the anthropogenic fills the differences between these results and the analyzed coal mining waste were negligible (Cui et al. 2004; Xu et al. 2007a). Locally, it is assumed that a lot of coal mining waste has been deposited in China without exact knowledge about the area of deposit, because apart from the registered 18,000–28,000 mines approximately 200,000 Artisanal and Small Scale Mines (ASM) are in use additionally (InfoMine 2007).

**Table 5.21** Physical and chemical properties of the coal mining waste soil (Data from Meuser and Blume 2001; Meuser 2006)

| Depth (cm) | Skeleton (mass %) | Sand (mass %) | Silt (mass %) | Clay (mass %) | Bulk density (g $cm^{-3}$) | | |
|---|---|---|---|---|---|---|---|
| 0–20 | 33 | 64 | 30 | 6 | 1.15 | | |
| 20–30 | 75 | 59 | 32 | 8 | 1.40 | | |
| 30–120+ | 73 | 66 | 26 | 8 | – | | |
| | | **vol % Water at pF** | | | | | |
| **Depth (cm)** | **TPV (vol%)** | **1.0** | **1.8** | **2.5** | **3.0** | **3.7** | **4.2** |
| 0–20 | 53 | 37 | 30 | 22 | 17 | 10 | 10 |
| 20–30 | – | – | – | – | – | – | – |
| 30–120+ | 42 | 26 | 19 | 14 | 13 | 12 | 10 |
| **Depth (cm)** | **$C_t$ (mass %)** | **$C_{humus}$ (mass %)** | **$C_{technogenic}$ (mass %)** | **$N_{org.}$ (mass %)** | **C/N** | **$CaCO_3$ (mass %)** | |
| 0–20 | 11.4 | 3.3 | 8.1 | 0.283 | 40 | 0.0 | |
| 20–30 | 20.4 | 1.9 | 18.5 | 0.255 | 80 | 0.0 | |
| r30–120+ | 14.2 | 1.5 | 12.7 | 0.234 | 61 | 0.0 | |
| b30–120+ | 29.7 | 3.6 | 26.1 | 0.332 | 89 | 0.0 | |
| **Depth (cm)** | **pH $CaCl_2$** | **pH $H_2O$** | **P-$H_2O$ (mg $100g^{-1}$)** | **EC (dS $m^{-1}$)** | | | |
| 0–20 | 4.2 | 4.8 | 0.9 | 0.10 | | | |
| 20–30 | 4.2 | 4.8 | 0.1 | 0.05 | | | |
| r30–120+ | 3.5 | 4.1 | nd | 0.05 | | | |
| b30–120+ | 3.5 | 4.1 | 0.1 | 0.06 | | | |

(continued)

**Table 5.21** (continued)

| | Exchangeable bases ($mmol_c\ kg^{-1}$) | | | | |
|---|---|---|---|---|---|
| **Depth (cm)** | **Ca** | **Mg** | **K** | **Na** | **$\Sigma PAH_{EPA}$ ($mg\ kg^{-1}$)** |
| 0–20 | 49.5 | 6.4 | 5.3 | 0.4 | 13.0 |
| 20–30 | 48.0 | 8.4 | 5.5 | 0.3 | 9.9 |
| r30–120$^+$ | 12.9 | 2.3 | 4.6 | 0.4 | 0.9 |
| b30–120$^+$ | 16.8 | 3.8 | 4.0 | 0.2 | nd |

| **Depth (cm)** | **Cd ($mg\ kg^{-1}$)** | **Cr** | **Cu** | **Ni** | **Pb** | **Zn** |
|---|---|---|---|---|---|---|
| 0–20 | 1.0 | 46 | 51 | 36 | 219 | 270 |
| 20–30 | 0.2 | 33 | 42 | 26 | 266 | 127 |
| r30–120$^+$ | 0.1 | 80 | 24 | 16 | 254 | 38 |
| b30–120$^+$ | nd | 80 | 50 | 17 | 173 | 43 |

nd = not detectable
Depth 30–120$^+$ cm red coloured (r) and black coloured (b)
PAH = polycyclic aromatic hydrocarbons; TPV = total pore volume
Metal extraction method: aqua regia

**Table 5.22** Heavy metal content of coal gangue and reclaimed soils (mg $kg^{-1}$) in Huainan mining area, China (mean values, standard deviation) (Data from Cui et al. 2004; Xu et al. 2007a)

| Location of coal mine | Material | Co | Cr | Cu | Ni | Pb | Sn | Zn |
|---|---|---|---|---|---|---|---|---|
| Xin Zhuangzi | Coal gangue | 7.3 ± 1.4 | 27.2 ± 4.8 | 14.5 ± 5.2 | 10.0 ± 3.6 | 16.8 ± 0.9 | 2.1 ± 0.6 | 31.8 ± 4.1 |
| | Soil near mine | 27.0 ± 4.7 | 32.0 ± 6.0 | 11.4 ± 1.6 | 11.0 ± 2.0 | 26.0 ± 4.0 | 2.5 ± 0.4 | 31.0 ± 4.0 |
| Huainan district | Coal gangue | 10.5 ± 5.1 | 50.7 ± 13.8 | 42.1 ± 21.2 | 211.1 ± 160.5 | 32.2 ± 11.2 | 7.5 ± 6.1 | 58.5 ± 29.2 |
| Da Tong (abandoned mine, n = 4) | Soil near gangue dumps | 22.2 ± 2.5 | 71.7 ± 6.0 | 75.4 ± 5.7 | 39.7 ± 1.5 | 40.3 ± 3.9 | 3.8 ± 0.6 | 128.2 ± 62.2 |
| Jiu Longgang (abandoned mine, n = 9) | Soil near gangue dumps | 22.5 ± 2.4 | 66.2 ± 5.5 | 56.3 ± 19.4 | 39.4 ± 16.6 | 27.9 ± 3.3 | 3.3 ± 1.0 | 73.6 ± 16.0 |
| Xin Zhuangzi (old mine, n = 6) | Soil near gangue dumps | 25.0 ± 2.4 | 59.1 ± 14.8 | 32.1 ± 3.7 | 27.8 ± 4.6 | 30.9 ± 6.3 | 3.0 ± 1.2 | 67.9 ± 28.6 |
| Pan Ji (new mine, n = 5) | Soil near gangue dumps | 22.3 ± 1.5 | 66.2 ± 12.9 | 34.2 ± 2.3 | 26.9 ± 2.8 | 23.8 ± 3.2 | 2.9 ± 0.6 | 77.7 ± 18.5 |
| Background value Huainan | Soil near gangue dumps | 10.7 | 64.9 | 24.2 | 25.7 | 30.8 | uk | 80.8 |

uk = unknown
Sampled depth: 0–30 cm
Metal extraction method: $HNO_3$, total concentration

Since the distance between the basis of the refilled material and the groundwater level is very low or even not existent, attention was also paid to the groundwater quality. In comparison with background values of the Yangtze headwater region the elements As, Cd, Cu, Hg, Pb, and Zn showed higher values. But only mercury was decisively higher by two to three orders of magnitude (Xu et al. 2007b). The coal gangue did not pollute the subsided water area but the accumulating pollution of the material releasing heavy metals should probably be considered in future. Because the kind of coal in this mining area lacks pyrite, it was not possible to find the process of acidification. The pH value of the groundwater ranged between 8.0 and 8.7, indicating alkaline reaction. Consequently, groundwater contamination might probably not be expected, since pH value and total concentration of heavy metals seemed to be beneficial.

#### 5.4.4.2 Ore Mining

The worldwide prospecting is also associated with the demand of modernized industry for high quality raw material that was increasingly a necessity for goods production. Important factors determining the ore quality are the ore grade meaning the concentration of a metal in an orebody, possible by-products enabling the extraction of several metals simultaneously, characters like the grain size distribution which, however, can be easily manipulated by man, and undesirable substances which are able to reduce the usage of the extracted ore (e.g. unwanted As and Hg in copper ores). In conjunction with the extraction procedure important factors are the size and shape of the orebody affecting the workability, the mining waste-to-ore ratio and, ultimately, the geographical location with reference to the ore transport by rail and ship (Evans 2005).

Environmentally, the ore extractions cause a number of disadvantages affecting the soil medium predominantly. Open-cast mining is strongly linked to damage to land, opening up large-scale devastated territories that cannot be backfilled completely. The other point of consideration refers to the release of toxic substances during extraction operations. Apart from the metal-rich dust deposited in vicinity of the mine, some harmful agents will be used for extraction purposes.

For instance, gold extraction in the Witwatersrand goldfields, South Africa is connected with a number of environmental, particularly soil problems. Because gold is resistant to most chemical substances, cyanides, mercury and chlorine were exclusively used for the extraction process until 1991. With the help of sodium cyanide solution gold was extracted. Cyanides were used for gold extraction in mineral processing plants leading to major contamination of groundwater and surface water as a result of the cyanidation process.

Alternatively, the so-called amalgamation process was used. A mixture of water and pulverized ore was poured on an Hg covered surface, enabling the creation of an Hg-Au compound. After distillation of mercury it was possible to isolate gold.

In order to improve gold quality (target concentration 99.5% pure gold) chlorine gas was added to trigger off a material reaction, leading to the volatilization of disturbing additional metals. Hence, in high quality gold manufacturing toxic substances were applied over a long period of time, leading to accumulation in soils and waterbodies. The wastewater and sludges from solution mining caused environmental damages in streams for 150 years (Evans 2005).

Due to the extreme exploration depth of up to 3,900 m below the surface the extracted mass reached an enormous quantity. Gold mining may produce the highest rate of mining waste in comparison with other mining treatment. On average, the ratio between material moved over and the resource extracted is up to 500,000:1 for gold but only 8:1 for hard coal and 11:1 for lignite coal (Genske 2003). Mining waste heaps were deposited since the beginning of the gold mining in 1886 when the Witwatersrand gold field was already discovered. The steep and in part unvegetated mining heaps had high erosion and deflation potential responsible for lateral dispersion of waste material and dust in adjacent areas. Additionally, it should be noted that the first environment management schedule was established in 1991.

Around gold mines in many towns of South Africa there are large mining waste deposits measuring an average of 80 ha on the surface and 35 m in height are present. The heaps had steep slopes, which facilitated runoff, because the water infiltration capacity was low. In dry seasons deflation occurred due to the high susceptibility of the silty material to wind erosion. It is difficult for natural vegetation to establish itself, since the saline and acidic properties of the mining material did not mean favourable site conditions. Arsenic and mercury can accumulate in the mining waste according to the gold extraction method applied. In any case, electrical conductivity (3.71–11.64 dS $m^{-1}$) and chlorine (163–1,296 mg $kg^{-1}$) had clearly accumulated in some mining heaps investigated in South Africa. The increased percentage of exchangeable sodium gave the surfaces a dispersive character, increasing the runoff additionally. Another problematical aspect was the pH value. During deposition, pH was 9.9, 1 month later 7.3 and 4 years later only 2.7. This rapid development was caused by pyrite oxidation and subsequently the presence of acid mine drainage (AMD). Accordingly, drain water from heaps indicated sulphate concentrations of up to 2,400 mg $L^{-1}$ and strongly raised electrical conductivity, reaching values up to 81.5 dS $m^{-1}$. With respect to the extraction treatment mobile cyanides can be components of drain water as well. Processed water from gold plants revealed 3.8 mg $L^{-1}$ (Van Deventer 2000b).

In Europe a lot of ore mines are of historic nature, since ore mining activities decreased in the nineteenth century after successful exploration of ores overseas. The abandoned sites are characterized by derelict buildings, unvegetated or hardly vegetated piles of mine waste and tailing lagoons (see Section 5.4.5). After production stopped the areas remained mostly untouched and unrehabilitated. Especially the tailing material consisting of finely ground particles played and, in dry conditions still plays, a major role in environmental pollution. Apart from the metals extracted from ores co-products caused additional threats to humans and the environment. The most important ores of concern are called chalcopyrite ($CuFeS_2$), galena (PbS), and sphalerite (ZnS). In areas, where these mined metalliferous ores were extracted, contamination of surrounding soils, stream sediments and vegetation must generally be expected.

Two mines in Britain, one near Truro in Cornwell (Pb-Zn-Cu), closed already in 1885, the other (Wemyss mine Pb-Zn) near Aberystwyth, closed in 1908, were considered in relation to environmental pollution. Surrounding topsoils within a radius of 300 m, sediments of streams touching the mine heaps, tailing heaps and wild vegetation were sampled (Table 5.23). At Wemyss mine, for example, the soils showed increased results in comparison with the background values of Wales (mean concentration Cd: 0.3 mg $kg^{-1}$, Pb: 31 mg $kg^{-1}$, Zn: 106 mg $kg^{-1}$). However, stream sediments accumulated still more heavy metals, and the highest concentrations analyzed were referred to the tailings. In general, the order tailing > sediment > soil appeared to be true in old ore mine areas. The values measured in plant tissue were in a line with the ground results mentioned. Aerial deposition of wind blown material was responsible for the contaminated vegetation. With regard to Pb and Zn there was a tendency for the concentrations in springtime to exceed the fall results. With reference to all environmental compartments it was estimated that 53–75% of the annually transferred contamination was related to the dust deposition (see Section 3.2.2). Because of the metallic sulphide content in mine waste after deposition oxidation occurred, leading to increasing acidity. At Truro mine site, for instance, the pH value ranged between 3.4 and 4.1. Consequently, long-term solubility of the metals in mine waste and tailings might not be excluded in future. In the first instance, cadmium and zinc were associated with metal solutions, whereas lead was transferred in the form of larger particulates (Merrington and Alloway 2000).

Next to ore mining areas very high heavy metal concentrations can be expected. In the metal mining area of Cartagena (Spain) with a population of

**Table 5.23** Heavy metal concentration (mg $kg^{-1}$) of soils, sediments, tailings and vegetation (*Festuca ovina, Agrostis capillaris*) at Wemyss mine, United Kingdom (Data from Merrington and Alloway 2000)

| | Cd | | Pb | | Zn | |
|---|---|---|---|---|---|---|
| **Compartment** | **Mean** | Standard deviation | **Mean** | Standard deviation | **Mean** | Standard deviation |
| Soils (n = 22) | **1.6** | ±0.9 | **1,750** | ±1,510 | **335** | ±117 |
| Sediments (n = 10) | **2.8** | ±0.9 | **2,252** | ±1,164 | **1,310** | ±476 |
| Tailing heaps (n = 8) | **3.6** | ±0.6 | **5,590** | ±756 | **1,590** | ±452 |
| Vegetation, springtime (n = 8) | **0.4** | ±0.2 | **470** | ±326 | **166** | ±113 |
| Vegetation, fall (n = 8) | **0.5** | ±0.3 | **213** | ±198 | **123** | ±25 |

Sampled depth: to the depth of parent material (soil, tailing heaps), 0–5 cm (sediments) Metal extraction method: total concentration

220,000 metal extraction contaminated soils in the surrounding areas, exhibiting extremely high values in floodplain soils (Cd: 1–180 mg $kg^{-1}$, Cu: 26–460 mg $kg^{-1}$, Pb: 3,000–23,000 mg $kg^{-1}$, Zn 3,000–33,000 mg $kg^{-1}$) and in soils of residential areas, ranging from 1 to 11 mg $kg^{-1}$ for Cd, from 40 to 70 mg $kg^{-1}$ for Cu, from 700 to 1,700 mg $kg^{-1}$ for Pb, and from 400 to 1,400 mg $kg^{-1}$ for Zn, respectively (Chabari et al. 2005).

Soils with high salt contamination are typical for heavy metal mining. These areas consisting of bare soils are highly susceptible to wind erosion transporting salty material into surrounding catchments. The locations favour not only the occurrences of heavy metals in dust deposits, but also of salts. Therefore, dusty mixtures of heavy metals and salts can occur. This result has been found adjacent to a mining area in Cartagena, one of the driest regions in Spain. Hence, a salt concentration of up to 2% was detected in the upper portion of soils in neighbouring residential areas (Chabari et al. 2005).

An ore mining heap located in the German district of Osnabrück will be presented in detail (Table 5.24, Fig. 5.16). During the production of ore overburden that could not be smelted was deposited in the form of smaller flat-topped mine dumps called ore mining heaps. In particular, the extraction of heavy metals, lead and zinc, could not be carried out long-term at the site because of the high level of

**Table 5.24** Profile description of a typical ore mining heap soil (Data from Meuser and Blume 2001; Meuser 2006)

| | |
|---|---|
| Site | Hasbergen (Osnabrück district), Germany |
| Relief | Culmination |
| Land use | Forest |
| Vegetation | *Fagus sylvatica*, lacking in underwood |
| Cut and fill | Anthropogenic deposit/dump |
| Groundwater level | Extremely deep (>20 dm) |
| Depth (cm) | Description |
| 0–10 | Sandy-loamy silt, medium skeletal content, very humic, 5YR 3/3, low compactness, sub-angular blocky structure, root fibre, high content of carbonate, root tubes, macropores, distinct transition |
| | Additions: ore mining material (3–15%) |
| 10–20 | Very silty sand, medium to high skeletal content, very slightly humic, 7.5 YR 4/6, very low to low compactness, very strong root penetration, very high content of carbonate, root tubes, macropores, diffuse transition |
| | Additions: ore mining material (60–85%) |
| 20–70$^{+}$ | No fine soil, skeletal soil, very slightly humic, 7.5YR 4/6, mean root penetration (in joints), very high content of carbonate, macropores/joints |
| | Additions: ore mining material (monosubstrate) |
| Type of soil (WRB) | Technosol/(Rendzi-skeletic Leptosol) |
| Parent material | Ore mining material |

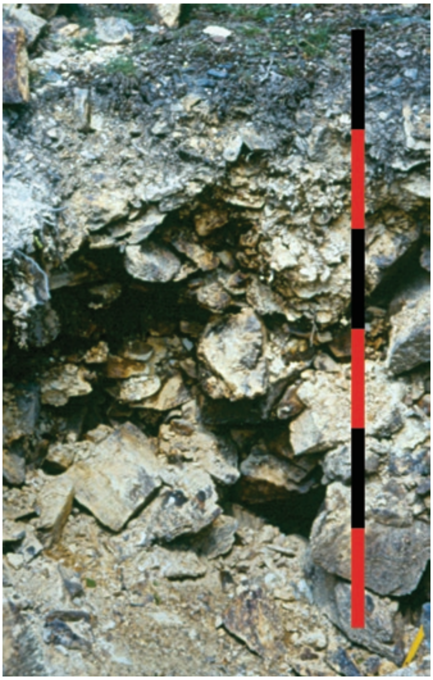

**Fig. 5.16** Soil profile of an ore mining heap in the district of Osnabrück, Germany (One measuring staff segment = 10 cm)

mine waste (barren rock). The dumps in the catchment area of the site under investigation were filled up until around 1930. In 1966 reforestation measures began at several of the dumps, principally with alders. Other dumps automatically became covered with greenery and reached the forest stage (Meuser and Blume 2001).

In the soil profile, a pedogenically created A horizon was discovered above weathered mining waste material that merges into the skeletal soil. The influence of weathering processes of the profile on the contamination is taken into consideration in Section 6.3.3. Due to the tipping, very large joints were found in the subsoil. These joints rendered site sampling with drilling sticks or driving core sounding rods impossible. The tipping material at this site also consisted of only one substrate (monosubstrate).

The most significant results of the analyses were as follows (Table 5.25):

- The skeletal content increased distinctly with increasing depth; only in the uppermost 20 cm does fine material, which is silty-sandy, exist at all, due to weathering.
- The ore mining material consisted predominantly of dolomitic limestone (aggregate for smelting), as the very high $CaCO_3$ contents and exchangeable Ca cations showed.
- The heavy metal contents in the parent material showed increased values in the strata above the parent material (Meuser and Blume 2001; Meuser 2006).

**Table 5.25** Physical and chemical properties of the ore mining heap soil (Data from Meuser and Blume 2001; Meuser 2006)

| **Depth (cm)** | **Skeleton (mass %)** | **Sand (mass %)** | **Silt (mass %)** | **Clay (mass %)** | | |
|---|---|---|---|---|---|---|
| 0–10 | 49 | 34 | 56 | 10 | | |
| 10–20 | 57 | 46 | 46 | 7 | | |
| 20–70$^+$ | 100 | nd | nd | nd | | |
| **Depth (cm)** | **$C_t$ (mass %)** | **$C_{humus}$ (mass %)** | **$C_{technogen}$ (mass %)** | **$N_{org.}$ (mass %)** | **C/N** | **$CaCO_3$ (mass %)** |
| 0–10 | 4.5 | 4.5 | 0.0 | 0.327 | 14 | 29.4 |
| 10–20 | 0.8 | 0.8 | 0.0 | 0.047 | 17 | 39.7 |
| 20–70$^+$ | 0.8 | 0.8 | 0.0 | 0.015 | 53 | 87.0 |
| **Depth (cm)** | **pH $CaCl_2$** | **pH $H_2O$** | **P-$H_2O$ (mg 100g$^{-1}$)** | **EC (dS m$^{-1}$)** | | |
| 0–10 | 7.4 | 7.7 | 0.2 | 0.17 | | |
| 10–20 | 7.6 | 8.1 | nd | 0.11 | | |
| 20–70$^+$ | 7.9 | 8.6 | nd | 0.10 | | |
| | **Exchangeable bases (mmol$_c$ kg$^{-1}$)** | | | | | |
| **Depth (cm)** | **Ca** | **Mg** | **K** | **Na** | **ΣPAH$_{EPA}$ (mg kg$^{-1}$)** | |
| 0–10 | 289.4 | 46.2 | 0.9 | 0.7 | 0.7 | |
| 10–20 | 322.2 | 14.6 | 0.2 | 0.4 | nd | |
| 20–70$^+$ | 382.9 | 15.5 | 0.1 | 0.5 | nd | |
| **Depth (cm)** | **Cd (mg kg$^{-1}$)** | **Cr** | **Cu** | **Ni** | **Pb** | **Zn** |
| 0–10 | 5.3 | 36 | 215 | 35 | 325 | 1,820 |
| 10–20 | 2.0 | 16 | 156 | 14 | 124 | 1,070 |
| 20–70$^+$ | 0.4 | 4 | 41 | 8 | 20 | 164 |

nd = not detectable
PAH = polycyclic aromatic hydrocarbons
Metal extraction method: aqua regia

In Australia, Canada, Kasachstan and South Africa uranium mining occurs for the purposes of energy and weapons. Until 1990 in the Eastern part of Germany intensive uranium mining took place as well. The landscape was excessively interrupted by open cast mines, shafts, heaps and tailings. Sludge derived from uranium extraction from rock was pumped into tailing ponds surrounded by dams. In such areas uranium contamination of adjacent soils was detectable, as was to be expected.

Uranium mining and uranium processing can cause a lot of environmental problems. For instance, in the surroundings of a uranium processing factory at Fernald, USA, high concentrations of up to 5,000 mg $kg^{-1}$ were reported and near waste combustion plants average concentrations of 500 mg $kg^{-1}$were found. But also in natural phosphatic rock uranium values between 30 and 300 mg $kg^{-1}$ were detected, leading occasionally to problems in relation to long-term fertilizing of phosphorus in cropland (see Section 3.4.1) (Lehmann 2000).

## *5.4.5 Sludge Fields*

### 5.4.5.1 Sewage Sludge Treatment Fields

As an alternative to sewage sludge application in cropland, wastewater treatment sludge is disposed of in so-called sewage treatment farms. That was common practice in developed countries until few decades ago and is currently widespread in less developed or developing countries. North of Berlin, Germany, large areas were used for this purpose between 1870 and 1990 (Fig. 5.17). The fields received up to

**Fig. 5.17** Sewage sludge treatment field in Berlin, Germany; the field is surrounded by earthen dams and a sheet-pile wall of steel

**Table 5.26** Heavy metal concentration (mg kg$^{-1}$) of 298 topsoils (0–10 cm) of a former sewage treatment farm north of Berlin, Germany (Data from Marschner and Hoffmann 2000)

| | Cd | Cr | Cu | Ni | Pb | Zn |
|---|---|---|---|---|---|---|
| **Mean** | **4.4** | **210** | **73** | **14** | **74** | **225** |
| Minimum | 0.1 | 1 | 3 | 1 | 6 | 13 |
| Maximum | 44.3 | 1,850 | 876 | 285 | 452 | 3,584 |

Metal extraction method: total concentration

**Table 5.27** Mean values of pH and element concentration (mg L$^{-1}$) in soil solution of three depths, sampled monthly from 1993 to 1997 by suction cups, in a former sewage treatment farm north of Berlin, Germany (Data from Marschner and Hoffmann 2000)

| Depth (cm) | pH | DOC | $NO_3^-$ | $SO_4^{2-}$ | Cd | Cu | Zn |
|---|---|---|---|---|---|---|---|
| 50 | 4.7 | 33 | 57 | 543 | 0.11 | 0.22 | 5.1 |
| 100 | 4.3 | 21 | 75 | 1,167 | 0.11 | 0.42 | 20.4 |
| 180 | 4.1 | 21 | 146 | 1,398 | 0.09 | 1.93 | 21.0 |

DOC = dissolved organic carbon

11,000 m a$^{-1}$ of untreated or only mechanically pre-treated sewage effluents. The fields converted to wastewater sludge fields remained under agricultural use. Afterwards, the areas were graded, ploughed and afforested. The soils accumulated heavy metals to a great extent (Table 5.26). In three depths the soil solution was sampled by ceramic suction cups. The parameters investigated showed large differences, indicating high variability in time and space. Nevertheless, the results exhibited high values for soluble metals and for some anions (Table 5.27). The subsoil to a depth of 180 cm did not show a filtering ability for mobilized metals and continuous downward movement had to be expected. The leaching of metals might occur to a depth up to 6 m, since to that depth sandy glacial till with relatively low pH values was present. In the case of the anions the leaching might continue till the groundwater table. The maximum values reached 978 mg L$^{-1}$ for nitrate and 4,360 mg L$^{-1}$ for sulphate (Marschner and Hoffmann 2000).

#### 5.4.5.2 Resource Extraction Tailing Ponds

Sludge derived from ore extraction mining is deposited in mine tailings. After separation processes (flotation, gravity separation, solution mining) it is necessary to dispose of acid wastewater using tailing ponds or lagoons. The tailing ponds may dry quickly, developing dust accumulated from mobile heavy metals and sometimes contaminated agents like cyanides. The sludge fields create bad conditions for plants due to contamination as well as physical impacts such as high soil density and platy soil structure. The acidic sludge produced during the operation must be collected. Thus, it is pumped into sludge fields termed tailing ponds that are surrounded by dams for drying purposes. In general, the acid and metal-rich sludge occupies large areas without any concept for recycling and re-use. Because of the sinking water table in the ponds the material near the surface tends to dry rapidly, causing dust problems for the neighbouring sites, since a quick re-vegetation is impossible in such phytotoxic conditions. Apart from the vertical migration of metals into groundwater, the main problem is associated with the deflation of the dust being inhaled by humans (Genske 2003).

A copper mine tailing in Mynydd Parys near Anglesey (United Kingdom) with a population of 70,000 was analyzed. The results showed a high metal content – not only Cu. The mean values of total concentration reached 5.7 mg Cd $kg^{-1}$, 1,905 mg Cu $kg^{-1}$, 7,690 mg Pb $kg^{-1}$ and 1,036 mg Zn $kg^{-1}$ respectively. The material exhibited a sandy texture, a bare and unvegetated surface and a surprisingly high carbon content of approximately 0.6%. The pH value was only 3.5, indicating strong acidity (Khan and Jones 2008).

There follows another example which confirms the same tendencies. It is located in close vicinity to the city of Tongling (China) with its 300,000 inhabitants. This area is one of the largest non-ferrous metal bases in China consisting of many mines, especially copper mines. Due to the rapid economic development of the last 2 decades large-scale exploitation took place, resulting in a series of environmental problems. The main problem is related to the tailing ponds covering about 600 $km^2$. The ponds dried continuously but the phytotoxic metal concentration did not allow plant growth. Results from the metal analyses were as follows: arsenic 3,700–12,000 mg $kg^{-1}$, cadmium 13–57 mg $kg^{-1}$, copper 500–1,900 mg $kg^{-1}$, lead 280–850 mg $kg^{-1}$ and zinc 570–1,600 mg $kg^{-1}$ respectively. The high values were, in principle, comparable with the results presented above. High bulk density combined with lower surface stability, deficient humus and nitrogen content and the extreme pH value were further reasons for unfavourable plant establishment (Fig. 5.18). Biotic crusts grew only on the tailings as an early stage of primary succession. Subsequently, the tailing ponds might form an unsightly scene. Accordingly, it was possible for highly polluted flying sand derived from the upper layers of the former sludge fields to be transported into neighbouring sensitive areas like the residential Tongling suburbs and cropland (Sun et al. 2004).

By means of decomposition of the biotic crust soil organic matter was increased in the course of time. Litter from natural communities influenced the development of organic matter increasingly and simultaneously nutrient supply. Therefore, the biotic crust may play an important role in the ecological restoration of copper mine tailing ponds. On the other hand, the high pyrite content of the material led to $H_2SO_4$ formation based on oxidation reaction during the drying process (see Section 6.3.3). Meanwhile, oxidation of sulphides caused a decrease in pH and an increase in soluble salts with a negative impact on plant growth. While pumped sludge initially indicated pH values of approximately 7.9, after some years the acidification progressed quickly, leading to low pH results (Sun et al. 2004).

Another tailing pond was analyzed in the Chiprovtsi region, Bulgaria. Mining activity in the form of Pb-Zn-Ag, Fe-As and Fe extraction took place from 1951 until 1999. The dust emission impacted agricultural land, water bodies and residents (see Section 3.2.2). In order to assess the impact of dust deposition untilled topsoils of adjacent territories were sampled to a maximum depth 30 cm. The results are presented in Table 5.28. The contaminated soil profiles revealed enrichment

**Fig. 5.18** Bare soil surface (*left*) and soil profile (*right*) of a copper mining area in Tongling, China; the blue colour is due to high concentration of arsenic and cyanides

**Table 5.28** Heavy metal concentration (mg $kg^{-1}$) of topsoils close to an ore mine tailing pond in the Chiprovtsi region, Bulgaria (Data from Mladenova et al. 2006)

| Profile | As | Cu | Pb | Zn |
|---|---|---|---|---|
| Profile A | | | | |
| Distance to tailing ponds 100 m in main wind direction | 65–515 | 69–92 | 42–117 | 126–146 |
| Profile B | | | | |
| Alluvial terrace, 800 m distance to tailing ponds, influenced by contaminated river water | 40–330 | 58–101 | 129–445 | 160–619 |
| Maximum accepted concentrations by Bulgarian legislation (for comparison) | 25 | 255 | 80 | 330 |

Sampled depths: 0–30 cm, 0–40 cm
Metal extraction method: X-ray fluorescence spectroscopy

with arsenic and heavy metals in the upper layer, in particular in the upper 5 cm. In an alluvial terrace soil high concentrations were analyzed at a deeper level because of flooding (Mladenova et al. 2006).

Furthermore, the other parts of the food chain were taken into consideration. The transfer soil-plant revealed element-specific results. Arsenic was absorbed with low intensity. The uptake capability of grasses followed the order Zn > Cu > Pb > As. The maximal transfer coefficient plant-soil, however, only reached 0.06 (grass) and 0.09 (clover) in the case of the mobile element zinc (see Section 7.3.3). Increased values in the milk of sheep grazing at adjacent sites were detected, while results from animal excrement as an integral part of the bio-circle were relatively low (Mladenova et al. 2006).

It is possible to discover the high contamination level by taking abandoned gold mines in Europe into account. Gold was exploited until 1964 close to Limoges (France) with a population of 150,000. The minerals were associated with pyrite ($FeS_2$), arsenopyrite (FeAsS) and galena (PbS). After extraction the material was crushed and spread out by floods in decantation basins. The basins contained 0.15–5.97% arsenic and 0.01–2.09% lead. With increasing age of the lagoons both metals were reduced, but the tailing water showed countercurrent behavior. For instance, the As concentration increased from 31.5 mg $L^{-1}$ (relatively fresh inflow area) to 2,205 mg $L^{-1}$ (older vegetated part at the boundary of the basin) (Neel et al. 2000).

Oil extraction may cause detrimental soil characteristics as well. The extent of pollution from crude oil in sandy soil at different distances from oil wells at Ras Sudr, Egypt, was determined. Total Petroleum Hydrocarbons (TPH) represented 82.8% of the studied crude oil containing C11–C32 aliphatic chain lengths. The soils located in the vicinity of the oil well (50 and 200 m) were polluted areas (>200 mg $kg^{-1}$ TPH) and the soils at 800 m from the oil well were moderately polluted (50–200 mg $kg^{-1}$ TPH) as they contained 142.9 mg $kg^{-1}$ TPH on average (Abdel-Hamid et al. 2005).

Of more of interest than the soils in proximity to oil extraction wells are the refinery sludge fields close to the refinery plant. Sludges generated in several stages of the refining process for more than 40 years were dumped in lagoons near Homs (Syria) with a population of 870,000. The sludge produced during waste water treatment consisted predominantly of semi-solid material with different properties. Due to growing production some lagoons were completely filled and afterwards overflowed, contaminating the adjacent areas. After construction of new lagoons the territory covered by the lagoons reached immense extension. The sludge fields were accessible for everyone, sheep grazing occurred at the surrounding dams. Four lagoon types were investigated in detail (Table 5.29). Consistency and water content varied, the pH value indicated a slightly alkaline reaction, Total Organic Carbon (TOC) and nitrogen had accumulated and the sulphide concentration reached extremely high values due to the de-sulphurification process in the wastewater treatment facility. While the heavy metal concentration revealed moderate results (except for zinc), dramatic concentrations of organic pollutants typical for crude oil compounds were analyzed (Table 5.30). There is a tendency for old dried sludge fields to show biodegradation of organic compounds with regard to the lower values

**Table 5.29** Physico-chemical conditions of oil refinery sludge fields near Homs, Syria (Data from Anonymous, 2004, unpublished)

| Consistency | Water content (%) | TOC (%) | N (%) | C/N | Sulphide-S (mg $kg^{-1}$) | pH |
|---|---|---|---|---|---|---|
| New fresh sludge | 86.5 | 34.9 | 1.6 | 23 | 2,004 | 8.4 |
| Lagoon (fluid) | 44.4 | 28.6 | 0.5 | 68 | 3,957 | 8.3 |
| Lagoon (sludgy) | 42.9 | 24.2 | 0.4 | 76 | 2,802 | 8.2 |
| Dry sludge | 3.3 | 28.2 | 0.3 | 110 | 724 | 7.9 |

TOC = total organic carbon

**Table 5.30** Contamination of oil refinery sludge fields near Homs, Syria (Data from Anonymous 2004, unpublished)

| | Free oily phase | TPH | BTEX | $PAH_{EPA}$ | Cu | Hg | Ni | Zn |
|---|---|---|---|---|---|---|---|---|
| | (%) | (mg kg$^{-1}$) | | | | | | |
| New sludge | 8.5 | 190,900 | 4,290 | 549 | 94 | 0.7 | 57 | 349 |
| Lagoon (fluid) | 4.7 | 238,260 | 327 | 189 | 187 | 0.7 | 90 | 763 |
| Lagoon (sludgy) | 0 | 111,320 | 405 | 121 | 109 | 1.4 | 60 | 406 |
| Dry sludge | 0 | 151,030 | <1 | 205 | 30 | 0.4 | 52 | 737 |

BTEX = benzene, toluene, ethyl benzene, xylene; PAH = polycyclic aromatic hydrocarbons; TPH = total petroleum hydrocarbons

for BTEX aromatics well-known for favourable biodegradation rates. It was possible to deduce a similar development for Total Petroleum Hydrocarbons and Polycyclic Aromatic Hydrocarbons (Anonymus 2004).

#### 5.4.5.3 Dredged Harbour Sediment Fields

Continuous dredging operations occur in harbours, water courses and rivers to maintain shipping. The dredged material consists of sludges and it is mostly not feasible to determine the origin of these, because the suspended particles are able to move over long distances before sedimentation. Results from the dredged sludges in the harbour catchments of the Polish cities Szczecin (500,000 inhabitants) and Swinoujscie (40,000 inhabitants) are now introduced. The river Oder transported suspended matter of organic and mineral origin to the harbour areas. In relation to various factories, intensive shipping and detrimental waterworks in this region it was possible for highly contaminated sludges to be sedimented, exhibiting, in particular, increased values for heavy metals, mineral oils, Polychlorinated Biphenyls (PCB) and pesticides. To maintain ship transportation in the period between 1971 and 1986 about 3.3 million cubic metres have been dredged and pumped onto adjacent abandoned land that was surrounded by dams. The sedimentation did not occur homogeneously and mineral fractions were able to sediment much faster, leading to sandy material deposits close to the input tubes and generally below ground. In contrast, the organic sludge percentages sedimented predominantly in the upper parts of the sludge fields and the outlying areas. The sludge fields covered an area of more than 1,000 ha with differing thicknesses up to 6 m. The material showed a platy structure and reminded one of marshland soils at the seaside. It contained a lot of shell residues causing an alkaline reaction. A relationship between organic matter content and heavy metal concentration was found (Table 5.31). Another important aspect of the research study referred to the dependence of the location on the heavy metal content. The contamination level increased with proximity to the intensively used industrial port of Szczecin (Table 5.32) (Kollender-Szych et al. 2008).

**Table 5.31** Average heavy metal concentration (mg kg$^{-1}$) of dredged sludges taken from the harbour area Szczecin – Swinoujscie, Poland divided into different organic matter content classes (Data from Kollender-Szych et al. 2008)

| Organic matter | Cd | Cr | Cu | Hg | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|
| Class (%) | (mg kg$^{-1}$) | | | | | | |
| 0–5 (n = 24) | 0.8 | 4.5 | 16.9 | 0.2 | 4.1 | 29 | 93 |
| >5–10 (n = 11) | 4.5 | 9.9 | 61.8 | 0.7 | 14.0 | 89 | 452 |
| >10–20 (n = 29) | 10.1 | 20.4 | 129.3 | 1.4 | 25.9 | 191 | 933 |
| >20 (n = 24) | 11.0 | 17.3 | 143.5 | 1.8 | 31.8 | 202 | 1,168 |

Metal extraction method: total concentration

**Table 5.32** Average concentration of organic matter content (%) and heavy metals (mg kg$^{-1}$) of dredged sludges taken from different harbours in the Szczecin catchments, Poland (Data from Kollender-Szych et al. 2008)

| Parameter | Port | | | |
|---|---|---|---|---|
| | Szczecin (n = 40) | Police (n = 11) | Trzebiez (n = 3) | Swinoujscie (n = 7) |
| | →→ increasing distance to industrial port of Szczecin →→ | | | |
| Organic matter | 14.2 | 15.5 | 9.6 | 1.8 |
| Cd | 9.8 | 7.4 | 3.8 | 0.4 |
| Cr | 18.5 | 8.1 | 14.4 | 3.2 |
| Cu | 131.9 | 52.1 | 45.1 | 5.9 |
| Hg | 1.3 | 1.1 | 0.6 | 0.1 |
| Ni | 24.8 | 16.9 | 16.6 | 3.7 |
| Pb | 176 | 113 | 108 | 11 |
| Zn | 896 | 417 | 592 | 41 |

Metal extraction method: total concentration

Since 1980 sludge dredged from the river Elbe and the harbour area in Hamburg (Germany) with a population of 1,800,000 has been disposed of in disposal sites which are later used for different purposes such as allotments and residential areas. The port of Hamburg is influenced by tides and during high tide approximately 50 million cubic metres of water flows into the harbour basin. Simultaneously, the river Elbe transports great amounts of water over a distance of more than 1,000 km from the source region to the estuary at the North Sea. Consequently, suspended materials can sediment in the harbour basins. For this reason, the port authority has to dredge between 3 and 4 million cubic metres of suspended sediments annually and of this about 1.2 million cubic metres are treated and disposed of ashore. In the meantime 181 sludge disposal fields have been registered, covering 5,800 ha or 7% of the city area of Hamburg. Depending on the sedimentation conditions the material consisted of sand and organic coastal marsh sediments. The latter was predominantly found in the harbour basins, where the water flow rate is extremely low. The sandy dredged material was mainly used for the establishment of new residential and industrial areas, while the organic material served as ground for agriculture and garden land because of its high available water capacity as well as beneficial

**Table 5.33** Metal ranges (mg $kg^{-1}$) from dredged material sludge fields (depth 0–20 cm, disposal in the 1980s) in Hamburg, Germany, divided into sandy and organic fraction (Data from Nebelsiek et al. 2005)

| Material | | As | Cd | Cr | Cu | Hg | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|
| Sandy (n = 70) | Maximum | 21 | 1.8 | 78 | 41 | 0.3 | 50 | 64 | 161 |
| | Minimum | 5 | 0.1 | 6 | 8 | <0.1 | 7 | 14 | 34 |
| Organic (n = 240) | Maximum | 289 | 24.8 | 244 | 1,309 | 26.9 | 114 | 1,009 | 3,123 |
| | Minimum | 10 | 0.6 | 5 | 24 | 0.2 | 6 | 36 | 89 |

Metal extraction method: aqua regia

nutrient status. Intensive investigations of the sludge fields exhibited the enormous contamination potential, particularly with regard to the material dredged between 1930 and 1990. We were able to find increased values for metals such as arsenic, cadmium, copper, mercury and zinc and, additionally, for Petroleum Hydrocarbons, Polycyclic Aromatic Hydrocarbons (PAH), PCB and PCDD/F (Polychlorinated Dibenzodioxins and –furans). Moreover, metal-organic compounds like tributyle tin derived from the ship paintings revealed high concentrations. In Table 5.33 metal values are presented – these are divided into the sandy and organic fractions of the dredged material. In summary, the organic material indicated much higher concentrations than the sand due to the higher cationic sorption capacity of the organic coastal marsh sediments (see Section 4.3.1) (Nebelsiek et al. 2005).

Homogeneous dredged sediment deposits in the north of France were chemically investigated, indicating that the studied material is of a complex nature, because it consisted of numerous reactive substances such as sulphides, oxides and organic matter. During dry periods the water table lowered and the sludge fields dried. Thus, it was exposed to oxidation of sulphides leading to $H_2SO_4$ formation and, subsequently, the pH value decreased. In contrast, in wet periods the hydromorphic state dominated, the conditions were reductive and hydroxides might dissolute. Both processes can produce enhanced heavy metal mobility, since the acid conditions during dry periods of time cause higher heavy metal mobility due to proton exchange, and the hydroxides usually capable of heavy metal occlusion release the metals after their dissolution (see Sections 6.2.1 and 6.2.2). In particular, the elements Cd and Zn, which showed high accumulation (Cd: 237 mg $kg^{-1}$, Zn: 6,049 mg $kg^{-1}$ average content), tended towards high mobility in the sludge field area. Besides, in the case of zinc the high available content resulted from the progressive weathering of minerals containing Zn, which already increased the exchangeable fraction 5 years after the disposal (Lions et al. 2007).

#### 5.4.5.4 Industrial Sludge Fields

Local sludge fields are sometimes results from specific industrial processes. Hence, they are located in proximity to the industrial plot and pumped into fields which are fenced in and observed by security. Nevertheless, the soil contamination is present and particularly in abandoned industrial areas with brownfield character, it is possible to enter the former fenced field areas, causing risk to human health and the environment. After a long period of time the sludge fields are dried, enabling people to walk on them. Some industrial sludge fields will be presented in order to demonstrate the extreme properties analyzed. Fly ash sludge fields are presented in Section 5.4.3. Some illustrations about the different sludge field types are presented in Fig. 5.19a–c.

In Bochum (Germany) with its 380,000 inhabitants a blast furnace sludge field that had not been used for several years was considered and established vegetation adapted to the problematical site conditions such as birches, willows and some eutrophic herbs. Physically, the pore volume was extremely high, ranging from 75 to 86 vol% (median 83 vol%). The bulk density ranged from 0.39 to 0.71 g $cm^{-3}$ (median 0.46). The texture consisted of 8% sand, 70% silt and 22% clay. Chemically, the soil revealed a median pH ($CaCl_2$) value of 7.0, $CaCO_3$ content of 14.7% and a carbon content of 13.7%, predominantly due to the inorganic and technogenic carbon fraction. But in the first instance this site appeared to be interesting with regard to the heavy metal concentrations. On average, the zinc content measured reached 27,000 mg $kg^{-1}$ and the lead content 7,000 mg $kg^{-1}$. Simultaneously, the available metal values ($NH_4$ acetate extraction) were also enhanced (Pb: 3,800 mg $kg^{-1}$, Zn: 5,100 mg $kg^{-1}$). Plant growth, however, seemed to be possible, since the nutrient capacity of the site was apparently beneficial. Nevertheless, leaves and roots of the analyzed woody plants indicated raised tissue concentrations. For instance, the roots of the birch accumulated 1,247 mg $kg^{-1}$ dry matter (DM) for Zn

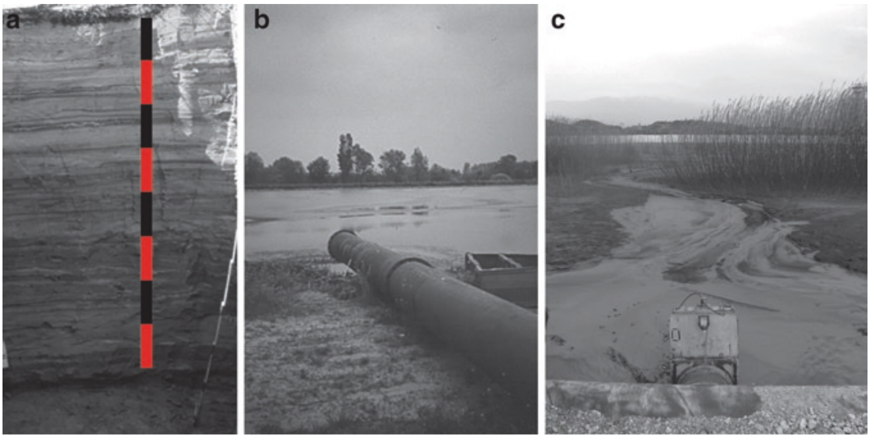


**Fig. 5.19** (**a**) Examples for sludge field types: Soil profile of a heavy metal extraction tailing pond (solution mining) in vicinity of Dresden, Germany; thin layers of sedimented material are visible, oxidized in the upper part of the profile, in reductive conditions below. (**b**) Examples for sludge field types: Inflow of a dredged harbour sediment sludge field in Szczecin, Poland. (**c**) Examples for sludge field types: Inflow of a fly ash sludge field in Pernik, Bulgaria (One measuring staff segment = 10 cm)

**Table 5.34** Profile description of a sludge field soil (Data from Meuser and Blume 2001; Meuser 2006)

| | |
|---|---|
| Site | Osnabrück, Germany |
| Relief | Plane |
| Land use | Former disposal area |
| Vegetation | Hydrophile vegetation (*Phragmites, Equisetum*), mosses, incipient bush-growth (*Betula, Salix*) |
| Cut and fill | Hydraulic ground fill |
| Dump water level | Shallow to medium (4 dm) |
| Depth (cm) | Description |
| 0–40 | Sand, very low skeletal content, very slightly humic (0–2 cm), 2.5YR 4/1, very low compactness, single grain structure, strong root penetration, carbonate-free, root tubes, cracks, macropores (rust patches), distinct transition |
| | Additions: sludge from sandstone washing (monosubstrate) |
| 40–50⁺ | Sandy silt, skeleton-free, humus-free, 2.5YR 3/1, very low compactness, strata structure, indifferent root penetration (aerenchyma), carbonate-free, root tubes, reduction-oxidation (light rust patches) |
| | Additions: sludge from sandstone washing (monosubstrate) |
| Type of soil (WRB) | Technosol/(Eutric Fluvisol) |
| Parent material | Sediments from sandstone washing |

and 205 mg kg$^{-1}$ DM for Pb. Zinc showed a stronger tendency to mobilize in plants, because the leaf contents amounted to 548 mg kg$^{-1}$ DM, whereas the maximum Pb values were 4 mg kg$^{-1}$ DM (Mansfeldt et al. 2000).

Extreme properties have been found in an industrial sludge field in the Osnabrück district, Germany as well (Table 5.34). However, this example is not associated with enhanced contaminant values. In turn, it shows the artificial character without soil contamination. Man-made soils can be completely artificial but not necessarily highly contaminated. The site of concern was located close to a sandstone quarry. The material was used for ashlars, crushed stone (road construction, railroad ballast) and as an aggregate for the concrete and asphalt industry. Processing was carried out with crushers, classifier sieves and washing facilities. In 1972, a central crushing plant with downstream washing facilities was put into operation. The settling ponds for the slurries generated by the washing of the stones were constructed. The sandy slurries produced by the washing processes consisting of an attrition technique and a hydrocyclone classifier were poured into the settling ponds. The site's disposal area was used for discharging until approximately 20 years ago and has been neglected ever since then.

The disposal area, which consisted of sediments from sandstone washing, was subject to natural succession and has now reached the stage of incipient bushgrowth. Pedogenically, an initial A horizon with a thickness of a few centimetres in parts has already been detected. The dump's own water level, which did not correspond to that of the groundwater, fluctuates several decimetres per year. When the profile

**Table 5.35** Physical and chemical properties of the sludge field soil (Data from Meuser and Blume 2001; Meuser 2006)

| Depth (cm) | Skeleton (mass %) | Sand (mass %) | Silt (mass %) | Clay (mass %) | Bulk density (g $cm^{-3}$) |
|---|---|---|---|---|---|
| 0–40 | 1 | 87 | 11 | 2 | 1.30 |
| 40–50$^{+}$ | 0 | 40 | 55 | 5 | – |

| | | vol% Water at pF | | | | | |
|---|---|---|---|---|---|---|---|
| Depth (cm) | TPV (vol%) | 1.0 | 1.8 | 2.5 | 3.0 | 3.7 | 4.2 |
| 0–40 | 52 | 41 | 36 | 10 | 6 | 3 | 1 |
| 40–50$^{+}$ | – | – | – | – | – | – | – |

| Depth (cm) | $C_t$ (mass %) | $C_{humus}$ (mass %) | $C_{technogenic}$ (mass %) | $N_{org.}$ (mass %) | C/N | $CaCO_3$ (mass %) |
|---|---|---|---|---|---|---|
| 0–40 | 0.3 | 0.3 | 0.0 | 0.026 | 12 | 0.0 |
| 40–50$^{+}$ | 0.5 | 0.5 | 0.0 | 0.030 | 17 | 0.0 |

| Depth (cm) | pH $CaCl_2$ | pH $H_2O$ | P-$H_2O$ (mg $100g^{-1}$) | EC (dS $m^{-1}$) |
|---|---|---|---|---|
| 0–40 | 6.5 | 7.1 | nd | 0.04 |
| 40–50$^{+}$ | 6.6 | 7.2 | nd | 0.06 |

| | Exchangeable bases ($mmol_c$ $kg^{-1}$) | | | | |
|---|---|---|---|---|---|
| Depth (cm) | Ca | Mg | K | Na | $\Sigma PAH_{EPA}$ (mg $kg^{-1}$) |
| 0–40 | 7.8 | 4.1 | 0.7 | 0.5 | nd |
| 40–50$^{+}$ | 14.1 | 3.3 | 0.8 | 0.1 | nd |

| Depth (cm) | Cd (mg $kg^{-1}$) | Cr | Cu | Ni | Pb | Zn |
|---|---|---|---|---|---|---|
| 0–40 | 0.1 | 19 | 9 | 14 | 16 | 65 |
| 40–50$^{+}$ | 0.1 | 26 | 21 | 20 | 20 | 64 |

nd = not detectable
PAH = polycyclic aromatic hydrocarbons; TPV = total petroleum hydrocarbons
Metal extraction method: aqua regia

was surveyed it was at a depth of approximately 40 cm. Several plant species at the site had root systems containing aerenchyma.

The most significant results of the analyses were as follows (Table 5.35):

- The upper 40 cm showed a high content of large pores and a low level of middle pores.
- Due to the early stage of succession, the sandy sludge and, below this, the sandy-silty sludge in the upper 40 cm still revealed a very low C and N content.
- The existing matrix led to low nutrient conditions, which revealed slightly higher values from a depth of 40 cm, due to the change of substrate.
- The soils achieved almost neutral pH values, and were classified as rich in base.
- There was definitely no contamination of the soil by heavy metals or Polycyclic Aromatic Hydrocarbons (PAH) (Meuser and Blume 2001, Meuser 2006).

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# Chapter 6
# Contamination Influencing Soil Properties

**Abstract** While the previous chapters focused mainly on the contamination potential of technogenic substrates and anthropogenic soils, this chapter will include the mobility-determining soil properties. In particular, the impact of physical characteristics present in urban environments such as sealing, erosion, deflation and the enrichment of the skeletal fraction are described in the context of the contaminated soil problem. In relation to the heavy metals their mobility in different anthropogenic deposits is estimated, involving a number of extraction methods like $NH_4$ acetate, EDTA, etc. Chemical parameters such as pH value, carbon content and binding agents are connected with the mobility assessment. The nutrient status of urban soils, which sometimes becomes relevant with regard to contamination, is discussed in detail. Furthermore, the influence of pedogenesis is treated, since pedogenic processes can alter contaminant potential and mobility. Again, this chapter covers many examples from all over the world.

**Keywords** Compaction • Contaminant mobility • Nutrient capacity • Pedogenesis • Physical properties • Sealing

## 6.1 Physical Properties

### *6.1.1 Sealing*

Based on the processing of aerial photographs the sealed surface amounted to 80–95% in the city centre of Moscow, Russia, with a population of 10,500,000. In industrial zones 80% was reported, some industrial plots exhibited only 10% unsealed vegetated surface. In modern residential city areas the degree of sealing was about 60% and in residential areas of peripheral suburbs 20–30%. New districts in the city were less sealed (41%) than older ones (51%) (Stroganova et al. 1998).

In Germany systematic investigation of the degree of sealing in different cities revealed the influence of urban structures. In summary, block building districts


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typical for the city centres varied from 70% to 100% sealed surface. The results of the other urban structures were 36–70% (terraced house district), 20–60% (areas with one-family houses and exclusive residences), 50–100% (industrial sites), 1–40% (parks) and 40–100% (road surfaces), respectively (Wessolek 2005).

In the northern part of Germany a method was established to determine the degree of sealing by analysing detailed digital cartographic data, thereby putting the statistical material related to the development of built-up areas and land use for infrastructure on a reliable footing. For this reason, it was feasible to calculate the average degree of sealing for a number of land-use types present in city areas (Table 6.1). The highest degree was associated with industrial and commercial uses, followed by traffic and residential areas. Areas for recreational purposes and urban agriculture had low percentages of sealed surfaces. It should be taken into account that, for instance, traffic areas indicating a degree of sealing of 64% only consist of both a completely sealed surface (road lanes) and roadside green belts (Dahlmann et al. 2001).

If the pollutant quantity exceeds the filter and buffer capacity of urban soils, groundwater becomes seriously endangered. For this reason, sealed areas can be assessed positively. Consequently, restriction to groundwater recharge should be related to deterioration of groundwater quality. In the urban environment sealed sites covering highly contaminated soils can sometimes improve the situation with regard to the risk to human health.

Usually an asphalt covering is placed on a levelled and compacted layer consisting of cancerous materials like concrete and slag. The asphalt layer and underlying load-bearing layer are tightly bound. It must be assumed that the material below the sealed surface is highly-modified and may indicate extreme

**Table 6.1** Degree of sealing according to digital cartographic data in the northern part of Germany (Federal State Lower Saxony) (Data from Dahlmann et al 2001)

| Land-use type | Areas of investigation | Degree of sealing (%) |
|---|---|---|
| Residential area (buildings and open spaces) | 17,457 | 35 |
| Commercial area (buildings and open spaces) | 875 | 76 |
| Industrial area (buildings and open spaces) | 1,282 | 80 |
| Mixed-use area (buildings and open spaces) | 333 | 62 |
| Public facility (buildings and open spaces) | 516 | 42 |
| Utility area (buildings and open spaces) | 254 | 70 |
| Traffic area (buildings and open spaces) | 1,033 | 64 |
| Square | 111 | 58 |
| Footpath | 1,372 | 15 |
| Sports ground | 105 | 8 |
| Garden area, allotment | 138 | 8 |
| Cemetery | 16 | 4 |
| Abandoned land, unused | 252 | 4 |
| Cropland | 143 | 0 |
| Pasture | 81 | 0 |

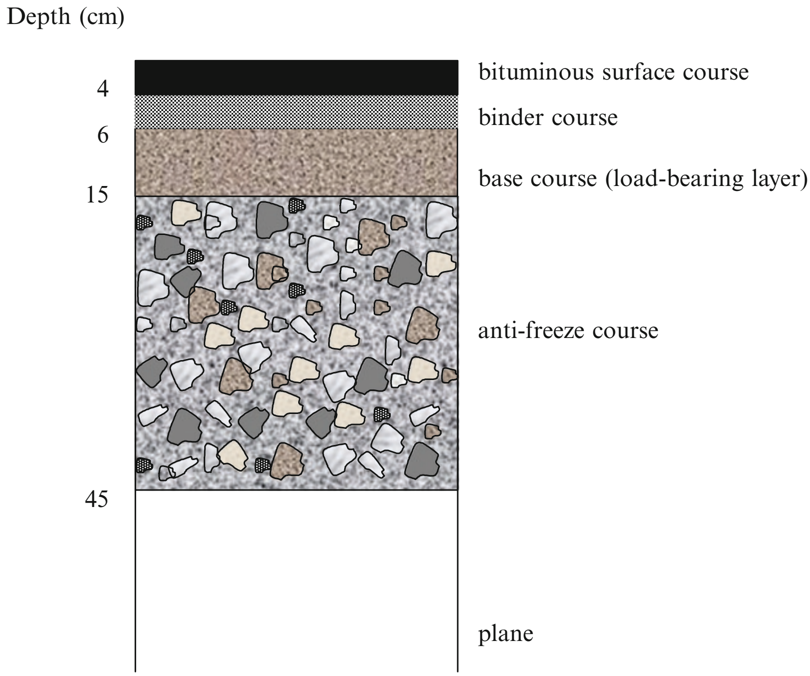



**Fig. 6.1** Standardized road construction design

properties compared with natural soils. The material used for road construction layers can be highly contaminated, as reported for roads in French cities (see Section 3.3.1). A standardized construction road design including potentially contaminated technogenic substrates is introduced in Fig. 6.1.

The humic topsoil has been removed previously but nevertheless the ground level is raised, since sandy and compacted layers were constructed in the first instance. Consequently, soils covered with asphalt do not permit contaminant input by water percolation as well as by dust deposition. In Moscow, Russia the heavy metal values in soils completely sealed with asphalt or concrete were much lower than in the directly adjacent soils. The covering may protect soils from pollutant input, preventing infiltration of rainfall into soil. Investigations of two profiles in the city centre resulted in a Pb concentration varying from 185 to 245 mg $kg^{-1}$ in unsealed anthropogenic soils, while the neighbouring sealed soil indicated 3–12 mg $kg^{-1}$ only (both analyzed to a depth of 40 cm). The findings, however, did not indicate that this was a general tendency due to the material lying below the pavement often being highly contaminated (Stroganova et al. 1998).

There are a lot of detrimental effects of sealing. Sealed surfaces alter the microclimate (reduced wind speed, humidity and ventilation, increase in air temperature), have an impact on biotopes (vegetation reduction or even complete loss, habitat fragmentation, spreading of species adapted to warm temperatures) and lead to a disturbance of the water balance (raising of runoff, reduction of water infiltration and altered evapotranspiration). Some further factors play an important role with reference to soil contamination.

Water infiltration and gaseous diffusion into the subsoil is strongly reduced, covering the soil with impermeable material like asphalt. However, infiltration can happen through asphalt layers due to cracks caused by heavy traffic and weathering processes, e.g. changes due to freezing and thawing. Preferential flow underneath was frequently discovered (Wessolek 2005).

The infiltration rate of sealed surfaces depends on the kind of material applied (Fig. 6.2a–e). Most of the materials showed relatively low total pore volume and water capacity. The available water capacity of concrete blocks amounted to 1.0 vol%, bricks amounted to 1.2 vol%, gum 4.2 vol% and asphalt only <0.5 vol%.


**Fig. 6.2** (**a**) Sealing materials: small slabs with narrow joints (0.5 cm) and reduced water infiltration rate. (**b**) Sealing materials: small slabs with wide joints (2 cm) and high water infiltration rate. (**c**) Sealing materials: honeycomb-type paving stone with high water infiltration rate. (**d**) Porous perforated sealing material with decreasing water infiltration rate in the course of time because of blocking and clogging by dust. (**e**) Water-bounded cover with high water infiltration rate

Accordingly, the total pore volume revealed 5.1 vol% (concrete), 22.2 vol% (bricks), 27.6 vol% (gum) and 21.6 vol% (asphalt), respectively. Such materials showed a delayed runoff on warm summer days, since the precipitation water evaporated quickly. In wintertime, however, the runoff was not considerably influenced by sealing materials. Porous sealing materials applied increasingly in cities exhibited higher water percolation during the first few years but road dust, weathered materials and clayey and humic particles deposited may fill the small cracks, resulting in a decrease in water infiltration in the course of time. Hence, the infiltration decreases in the long run, since the fine pores are blocked and clogged due to dust deposition.

With reference to the percentage of humus in joints and pores of sealing materials it has been found that microbial biomass and organic matter indicated enhanced values. Some areas even exceeded the values found in ploughed topsoils of cropland. It can therefore be concluded that the biological activity in cracks and voids might reveal enhanced values. Unfortunately, it is not yet clear whether the increased biomass and activity is accompanied by increased biodegradation of organic pollutants present.

On footpaths and sports fields water-bound covers are frequently constructed. The water infiltration of this type of sealing material depends on the grain size distribution in the cover and in the load-bearing layer below. The infiltration rate is high, if sandy and skeletal materials had been used (Wessolek 2005). In Table 6.2 the drain coefficient of different sealed surfaces is presented.

If water can percolate through porous and weathered covers, the soil beneath may remain moist for a prolonged time since water is not able to evaporate. Microorganisms can still live under the cover and maintain the gas exchange between atmosphere and soil vapour. The natural ventilation however is limited. Therefore, the soil might become waterlogged, promoting higher humidity in adjacent basements and cellars. High humidity in rooms accelerated the development of pathogens and fungi, causing human health problems (Stroganova et al. 1998). In dry climates the covered soil tended to remain permanently dry (Craul 1992).

**Table 6.2** Average drain coefficients (water infiltration divided by precipitation minus evaporation) of different urban areas (Data from Wessolek 2005)

| Type of area | | Drain coefficient |
|---|---|---|
| Buildings | Roof | 0.1 |
| | Gravelly roof | 0.5 |
| | Green roof with intensive planting cover | 0.7 |
| | Green roof with extensive planting cover | 0.5 |
| Roads, footpaths, sports fields, squares | Concrete pavement | 0.3 |
| | Water-bounded area | 0.5 |
| | Pavement with joints >15% | 0.4 |
| | Synthetic sports area | 0.4 |
| | Lawn sports area | 0.7 |
| Open spaces | Park with vegetation, gravel and clinker paving | >0.9 |
| | Garden with water-bounded cover | >0.9 |

It should be taken into account that even low intensity precipitation rates of <10 mm $h^{-1}$ may cause runoff in city areas. The runoff from sealed surfaces is frequently contaminated due to long-term dust deposition before the precipitation occurred influenced adjacent sites (see Section 3.2.2). High pollutant concentrations were reported in nearby topsoils of parks and gardens, since rainwater infiltration into unsealed neighbouring sites had happened. The topsoil contamination might be additionally caused by dust-covered leaves of trees and shrubs falling to the ground in autumn and creating polluted litter.

Moreover, in residential areas of developed countries, where a lot of people own dogs and cats, the input of contaminants in adjacent open space resulting from runoff from sealed sites stems from dog or cat urine and excrement as well. During rainfall large quantities of the problematical residues may be discharged into the neighbouring areas. In this way sensitive uses such as playgrounds, sand-pits and gardens are burdened by pathogens, impacting children's health (Wessolek 2005).

### 6.1.2 Erosion and Deflation

In urban areas the erosion and deflation potential can be much higher than in rural areas. For instance, it was measured that the erosion of a construction site with bare soils in metropolitan Washington D.C. (USA) with its 580,000 inhabitants reached maximum values of 218 t per acre and year. In general, where construction activities are present and most of the vegetation has been removed, the erosion rates are very high. Typical phases of construction leading to a high erosion potential are site preparation, building demolition, soil excavation and cut and fill operations and, ultimately, back-filling and soil handling operations. The humic topsoil is usually stripped and stockpiled during the construction phases. Therefore, the soil below remains unprotected throughout the whole period of construction. Consequently, it is not surprising that in urban construction areas the highest erosion rates were measured, followed by residential areas. In contrast, forest, pasture as well as public and commercial sites revealed a low erosion rate of 0.2–0.3 t per acre (Fig. 6.3) (Craul 1992).

The enhanced erosion and deflation rate in urban areas was particularly caused in heavily used urban areas such as parks, playgrounds and ball fields. Here, the vegetation cover was mostly destroyed and the soil below compacted. Both factors contributed to the susceptibility to erosion. A bare soil was not able to absorb the stress, resulting in a compacted surface layer and often crust creation near the surface. Such situations were usually found on ball fields and playgrounds used intensively or, in other words, concentrated foot activity (Fig. 6.4). In overused hilly areas accelerated erosion quickly occurred, creating deep rill and gully erosion. These erosion lines worsened the erosion potential during heavy rain considerably (Craul 1992).

In countries where roads in urban areas outside city centres are untarred, deflation may occur. This situation is very common in less developed countries, as we

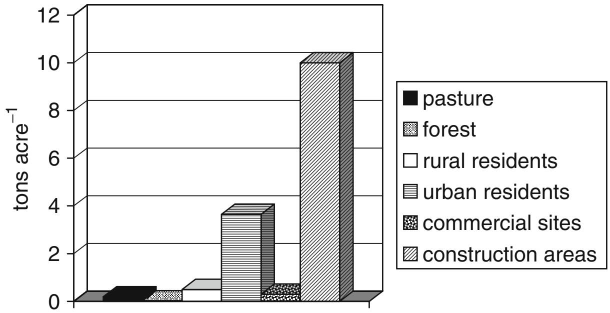



**Fig. 6.3** Sediment yield from June to September based on erosion in distinct landscapes in Lutes Run, USA (Data from Craul 1992)

**Fig. 6.4** Bare soils, development of surface crust and erosion rills resulting from overuse by playing children

can find, for example, in Africa (Fig. 6.5). In the capital of Cameroon, Yaounde, with 1,700,000 inhabitants, the negative effects of untarred motorways were observed. In the rainy season untarred roads caused mud development and slippery road surfaces. Tyres of heavy trucks and buses were responsible for a surface with deeply dug rills and gullies. Roads were unpassable for long periods. Eroded material was transported to canalization like water pipes, resulting in water pollution and destruction of pipes. Road depression occurred, affecting building walls and bridges along the roads. In the dry season unavoidable dust developed leading to deflation of contaminated dust caused by vehicles. Apart from construction areas, that were built incompletely and that let recognize bare soil surfaces, the

**Fig. 6.5** Untarred road in Kenya close to Mombasa provoking deflation in dry season and slippery surfaces in rainy season

untarred roads caused dust distribution around cities and villages to a great extent (Takam 2000).

### *6.1.3 Compaction*

Soil handling in the urban environment results mostly in compaction processes rather than loosening. In the case of sandy soils low in humus the weakly structured aggregates tend to break down (disaggregation) and in the following packing and replacement processes this material exhibits increased bulk density, since the sand grains are closely packed. Conversely, loamy and clayey soils are resistant to damage when dry, but they begin smearing and tend to compaction, if they are placed in wet conditions. Very high compaction must be expected, if the soil is so moist that the plastic limit is exceeded (Craul 1992).

Furthermore, stockpiling can cause compaction as well. The reasons are the pressure of the overburden and, in the case of handling material with low humus content, damage to soil aggregates.

Compacted heaps of stockpiled material revealed anaerobic conditions downwards, reducing the biological activity. In sandy soil heaps the boundary between aerobic and anaerobic zone was at approximately 2 m, in silty heaps at 1.3 m and in clayey heaps at even 0.3 m. Ammonia had accumulated, because the nitrifying by bacteria was hindered. The anaerobic conditions, however, are temporary, meaning original conditions of the material after re-spreading. The earthworm population was also restricted and the surviving population reached sometimes only 3% of the estimated numbers. In particular, in the first 1–2 years the abundance of earthworms decreased considerably, followed by a slow recovery process later on. It has been found, that stockpiled heaps indicated bulk densities from 1.58 to 1.64 g $cm^{-3}$ as compared to 1.12–1.15 g $cm^{-3}$ for adjacent undisturbed land. Accordingly, deposited and worked soils have reduced water permeability, reaching values less than $10^{-6}$ m $s^{-1}$. In general, after re-spreading the detrimental soil conditions might partly remain (Rimmer 1991; Craul 1992).

Usually valuable humic topsoil is stockpiled during excavation and construction activities and re-spread later on. Soil originating from arable land outside the cities and transported to construction and rehabilitation sites is stockpiled simultaneously. Both materials should be covered by sheets during stockpiling over long periods of time. However, in a large number of cases this necessity is not adhered to. Therefore, humic topsoil remains unprotected and is exposed to atmospheric input as well as waste removal for a long time. Apart from compaction and hardening, stockpiled humic soil heaps of a certain length and height can be contaminated. After re-spreading and levelling it can not be excluded that the material used for reconstitution exhibits already increased pollutant values.

Vehicle transport is another reason for soil compaction. During the transport the material is influenced by vibrational waves compacting the material. The material, which is located streetside afterwards, showed a high degree of compaction, particularly after a long period of depositing time typical for construction sites (Craul 1992).

Compaction by equipment traffic contributes especially to the problem. Earthscrapers and rubber-tired wheel loaders have higher compaction pressure than crawler-mounted equipment. But the latter is not used very often, since a separate truck must transport it to the site, a cost-intensive and relatively long-term process. Conversely, earthscrapers have the advantage that they can be transported to the construction site without any problems.

After the construction period has been finished, the kind of utilization might influence soil compaction. Intensively used areas like lawns in urban parks revealed enhanced bulk densities caused by both the construction process and overuse afterwards. It has been reported that values between 1.4 and 2.3 g $cm^{-3}$ (open parkland in Washington DC, USA) and 1.52 and 1.96 g $cm^{-3}$ (Central Park, New York, USA) were measured, usually exceeding values of agricultural areas. In Hong Kong, China, with a population of 7,300,000, two thirds of sampled soils of residential areas exceeded 1.6 g $cm^{-3}$ (Jim 1998). Accordingly, bulk densities of more than 1.6 g $cm^{-3}$ indicated strongly reduced air capacity in the subsoil, impacting the plant growth negatively (Mullins 1991).

In Moscow, Russia, the highest bulk density was found in playgrounds (1.86 g $cm^{-3}$). In lawns the bulk density was typically low (<1.2 g $cm^{-3}$), whereas strongly compacted trampled patches crossing the lawns approached 1.47 g $cm^{-3}$ (Stroganova et al. 1998). In Rostov-on-Don, Russia, with its 1,000,000 inhabitants, the maximum values were obtained in industrial and central parts of the city with high anthropogenic effects. In top horizons the maximum value of the bulk density was 1.38 g $cm^{-3}$. The value was increased, as a rule, in the bottom part of the soil profile and it reached

1.5–1.6 g cm$^{-3}$. In general, the bulk density was attributed to human-made soils and the soil-like substratum. A degree of soil hardness precisely correlated with bulk density of the soils as a whole (Morozov and Bezuglova 2005).

With regard to the classification of bulk density according to living conditions of ornamental shrubs and trees it is concluded that a bulk density >1.6 g cm$^{-3}$ indicates poor conditions and >1.8 g cm$^{-3}$ very poor conditions. In contrast, moderate conditions allow values ranging from 1.4 to 1.6 g cm$^{-3}$, good conditions 1.2–1.4 g cm$^{-3}$ and optimum root penetration is possible, exhibiting values below 1.2 g cm$^{-3}$ (Stroganova et al. 1998). In soils with very high bulk density, however, roots were able to grow in old root channels and cracks that were created by weathering processes.

Related to the living conditions of plants, compacted sites in built-up areas where constant pedestrian and vehicular traffic takes place often did not allow vegetation to establish itself. Therefore, overcrowded city areas with intensively used sports and leisure areas showed mostly bare soils and the residual vegetation was generally poor in species (Wessolek 2005).

The water infiltration is strongly affected by compaction. The limited total pore space and the low amount of macropores may reduce the possibility of infiltration considerably. Increased bulk density and reduced total pore space are normally linked to each other. Finally, the reduction of total pore space usually means a limited water holding capacity, too, since both parameters are in close agreement. Simultaneously, the gas exchange between atmosphere and soil is reduced or partly stopped and limited gaseous diffusion can occur. In particular, the oxygen influx is often decreased.

Table 6.3 presents some infiltration rates of urban soils in cities of the USA. Obviously, land-use type determined the water infiltration and permeability (Craul 1992). The results corresponded to investigations in Moscow, Russia, where the infiltration rate of pathways and uncompacted plots amounted to 4.8–21 cm h$^{-1}$ (Stroganova et al. 1998).

If the precipitation exceeds the infiltration rate, we can expect water ponding or enhanced runoff, a situation frequently observed in urban areas, especially on construction sites. Compacted layers as present on sidewalk areas and overused

**Table 6.3** Infiltration rate (cm h$^{-1}$) of different urban sites in cities of the USA (Data from different authors, cited in Craul 1992)

| Utilization | Infiltration rate (cm h$^{-1}$) |
|---|---|
| Abandoned pineapple field | 179.6 |
| Grubbed urban land | 75.7 |
| School yard | 66.8 |
| Golf course | 61.7 |
| Sidewalk area | 6.6 |
| Swimming pool area | 4.8 |
| Baseball field | 19.6 |
| Area with large slabs (60 × 60 × 7 cm) | 0.5 |
| Area with small slabs (30 × 30 × 5 cm) | 0.9 |
| Area with perforated bricks (20 × 20 × 10 cm) | 7.5 |
| Square bricks (10 × 10 × 10 cm) | 5.5 |

territories (as was to be expected in outdoor swimming pool areas) revealed the lowest infiltration rates, apart from sealed surfaces.

In subsoils the water percolation depends on soil permeability (hydraulic conductivity), level of the groundwater table, presence or absence of stagnation, topography (slope gradient and slope position) and bedrock permeability beneath shallow soils. The water percolation occurs in deposited soils mainly in voids resulting from technogenic ingredients like bulky construction and household waste (see Section 4.2.3). Furthermore, biologically determined factors may be the reason why water flows downwards, particularly earthworm channels and channels left by previous root penetration. It has been found that in loamy soils the water percolation was strongly associated with biopores, measuring a diameter of 1–6 mm in contrast to sandy soils, in which the biopores did not play a major role. Hence, the calculation of saturated water permeability should not be reduced to the parameters bulk density and texture. On the other hand, many biopores did not automatically mean a rise in water infiltration, since each biopore was not able to take part in water percolation caused by pore discontinuity (Dornauf and Burghardt 2000).

The infiltration capacity is strongly reduced in mining heaps as well, because they are deposited and mechanically compacted. Since 1782 hard coal mining has taken place in the Mecsek Mountains near Pecs, Hungary, with 160,000 inhabitants, leading to the construction of numerous coal mining heaps in the surroundings. Some spoil tips have been observed with reference to water infiltration capacity and chemical parameters. In the Nagbanyaret tip belonging to the generation of the 1950s heaps when municipal waste and mining waste had been deposited together, in this case in addition to waste from a leather factory, the water infiltration rate reached values between <0.01 cm $h^{-1}$ (clay cover) and 0.13 cm $h^{-1}$ (coarse debris cover). A further heap with a clay covering showed the same result of <0.01 cm $h^{-1}$, while a third heap with spontaneous reclamation and an embryonic soil covering reached 0.02 cm $h^{-1}$. All values must be as considered as relatively low (Czigany et al. 2000).

Comparable results have been yielded in coal mining areas in other parts of the world. In Butia (Brazil), with a population of 20,000, mining heaps of different ages were physically determined. The material showed irregular distribution of particle size and a blocky structure. The bulk density of the mechanically compacted heaps exceeded mostly 1.5 g $cm^{-3}$ (range: 1.31–2.03 g $cm^{-3}$), and, by analogy, a low water infiltration rate was found, lying usually below 0.01 cm $h^{-1}$, with maximum values of 0.09 cm $h^{-1}$ in the subsoil (Kämpf 2000).

In spite of this, water was, in principle, able to percolate through the heap downwards and laterally, as observed in the Hungarian example of Pecs. Because of missing results from collected water underneath the heaps water percolating laterally upstream and downstream was analyzed. The chemical properties tended to be strongly altered, since the water inflow ranged between 68 and 72 mg $L^{-1}$ sulphate but the outflow between an astonishing 6,800 and 7,150 mg $L^{-1}$ sulphate (Czigany et al. 2000).

Compaction may reduce water infiltration and percolation within the profile, but it will not completely prevent leachate of hazardous substances. Thus, in spite of high bulk density and a restricted water infiltration rate, the contaminant pathway to the groundwater may continuously remain and require that ongoing attention is paid to it.

### 6.1.4 Skeleton Enrichment

The presence of technogenic components in deposited soils may raise the gravel and stone content, since most of the technogenic substrates are coarse-grained. Consequently, high skeleton contents of industrial and urban soils as present in the Ruhr area, Germany, with 5,300,000 inhabitants, have been reported by Burghardt (1994). Forty-eight percent of the investigated soils indicated a stone content between 2 and 10 vol%, 14% between 11 and 30 vol% and 17% between 31 and 75 vol%.

In principle, in the presence of construction rubble physical properties tend to alter considerably. Systematic physical analyses of a German urban soil project dealing with a number of anthropogenic soils in different locations exhibited the differences between deposited soils without construction debris and with approximately 20% of construction debris (Table 6.4). In soil horizons containing sand, loamy sand and sandy loam the differences are more or less marginal with reference to air capacity and available water capacity. Horizons with loam and sandy clay loam tended to show a higher air capacity and available water capacity in the presence of construction rubble. In contrast, soil layers consisting of the texture classes silt and silt loam clearly exceeded the results from the rubble-containing soils related to the available water capacity. In the case of clayey horizons (silty clay, silty clay loam, clay loam), however, it was difficult to distinguish the air capacity, because the ranges were extreme due to the swelling-shrinkage processes in soils of these texture classes. The available water capacity did not reveal clear differences (Blume and Schleuss 1997).

Tree pits along the sides of roads show extremely physical conditions in association with the high amount of technogenic findings, as observed in Hong Kong (China). The high skeleton content caused reduced soil moisture and consequently water stress to the trees in warmer and drier seasons as they are well-known for the humid summer in the Chinese metropolitan. On average, the available water capacity (AWC) of 34 analysed soils alongside city roads was reduced to 17.6 vol%, the air capacity (AC) to 7.9 vol%. The minimum values reached 7.4 vol% (AWC) and 3.3 vol% (AC) only. After tree plantation the physical properties improved to average values of 19.2 vol% (AWC) and 7.9 vol% (AC) for 17 backfill soils. It was possible to achieve better conditions, if rootball soils from different nurseries were taken. The average results of a further 17 investigated sites exhibited 26.9 vol% (AWC) and 9.8 vol% (AC) (Jim and Ng 2000).

**Table 6.4** Air capacity and available water capacity AWC (vol%) of anthropogenic horizons (n = 197, sampling depths 0–150 cm) with approximately 20% construction debris and without artefacts in relation to distinct texture classes (Data from Blume and Schleuss 1997)

| | Anthropogenic horizons without artefacts | | Anthropogenic horizons with 20% construction debris | |
|---|---|---|---|---|
| Texture | Air capacity | AWC | Air capacity | AWC |
| Sand, loamy sand, sandy loam | 17–23 | 14–21 | 19–24 | 15–18 |
| Loam, sandy clay loam | 7–14 | 8–15 | 13–16 | 15–17 |
| Silt, silt loam | 2–23 | 27–39 | 9–13 | 11–15 |
| Silty clay, silty clay loam, clay loam | 0–23 | 11–16 | 8–11 | 12–15 |

Soils with porous technogenic substrates exhibited two pore systems. On the one hand pores between the single grains offered pores of different size determining water available, water capacity and air capacity (inter-pores), while on the other hand porous technogenic substrates such as brick and slag contained their own pore system additionally (intra-pores). Consequently, the pore system should be termed a dual system. The quantity of moisture stored in the skeletal fraction increased the water supply and air capacity significantly. For instance, the intra-pore volume of mortar revealed 29.5%, whereas in the case of red brick stones average values of 41.7 vol% were detected. But it is assumed that the water stored in the technogenic substrates is partly available to plants only (Burghardt 1994).

In summary, construction debris (mainly brick and mortar), coke as well as some kinds of bottom ashes have a high amount of medium pores contributing to the available water capacity, while most of the slag types (e.g. blast furnace slag, steel-works slag) enhance the air capacity with respect to many coarse pores included. Heavy metal slag and most of the refuse particles contain few pores only. Because the connection between both pore systems is limited, the active flow of soluble chemicals is supposed to be reduced, since flow active inter-pores and flow inactive intra-pores are not very compatible with each other.

Detailed analyses of the physical properties of two deposited soils containing construction debris showed the impact on porosity of some artefacts. The first soil included red coloured and yellow coloured bricks. Bricks indicated a specific gravity of 2.63–2.66 g $cm^{-3}$, a total pore volume of 34–36 vol%, an available water capacity of 14–17 vol% and an air capacity of 5 to 9 vol%. Similar results were found at the second soil profile containing red and yellow bricks as well, indicating 2.65–2.72 g $cm^{-3}$ (specific gravity), 33–38 vol% (total pore volume), 14–20 vol% (AWC) and 1–3 vol% (air capacity), respectively. Analogously, the values for mortar in both soils varied between 2.4 and 2.62 g $cm^{-3}$ (specific gravity) as well as 31 and 36 vol% (total pore volume). In conclusion, construction debris such as brick and mortar might improve the living conditions for plant roots, since they supply additional available water, but they do not support the air capacity, since most of the pores belong to the medium pores mainly responsible for water supply (Blume and Schleuss 1997).

If an exchange between percolating water of the inter-pores and the water stored in intra-pores takes place, the immobile intra-pores can serve as a pollutant source or as a sink influencing the transport of hazardous substances. In particular, technogenic substrates with open pores may take part in this water exchange. Other technogenic substrates with pores, which have no outlets at the substrate surface and accordingly do not influence the concentration of the flowing water, may react differently. Thus, if the water percolation in deposited soils uses voids and cracks in a line with preferential water flow, the pollutant leaching will depend on the type of technogenic substrates present. In conclusion, there are distinct factors impacting contaminant leaching in deposited soils. Physically, the intra-pores can delay downward percolation of water and soluble pollutants. Chemically, they can serve as an additional source for hazardous substances in the case of interaction between both pore systems (Bädjer and Burghardt 2000).

### 6.1.5 Altered Groundwater Table

As a result of the high percentage of sealed surfaces in urban areas the average groundwater recharge was reduced and totalled 10–20% of the annual precipitation in Berlin, the capital of Germany, with 3,400,000 inhabitants. In natural landscapes of the Berlin periphery 23–24% (arable land) and 14–21% (woodland) were measured. The difference related to the yearly results was surprisingly low, because in natural landscapes transpiration by vegetation played an important role, restricting the groundwater recharge, in contrast to the urbanized area where vegetation is removed to a great extent (Wessolek 2005). Alteration of groundwater recharge might influence contaminant release into the aquifer, in particular in relation to water soluble compounds such as nitrate, sulphate, phenols, etc.

Due to sealing the groundwater table is lowered in a number of metropolitan areas. On the other hand, the opposite effect of a rising groundwater table was discovered in urban areas as well.

The decline of heavy industry, the development of new surface water supplies and the leaking of water pipes are the reasons for rising groundwater tables in a lot of cities in developed countries. In the case of widespread cessation of groundwater extraction a new hydrological problem occurred. One impressive example for this is the city of London, the capital of the United Kingdom, with a population of 13,200,000. The water table rose from 80 m in 1950, 50 m in 1995 up to 20 m below the surface in 2005. The development created different structural damage, which can be observed in buildings, tunnels and sewers. The re-swelling clayey underground was one of the main problems with reference to the rising water table. The Public Authority was afraid of flooding of the subway, located at a depth between 20 and 40 m below the surface, and other sub-surface structures such as cellars and deep basements normally associated with high-rise buildings. Consequently, underground facilities were partly redecorated and waterproofed or even rebuilt. In order to react to the danger to the subway tunnels permanent water pumps were installed to compensate for the rising water table, which was calculated as being 2 m year$^{-1}$. The other point of view related to the rising groundwater table was the possible direct contact between the contaminated subsoil of former industrially used areas and the water table, since pollutants can reach the groundwater rapidly after soaking out of the contaminated soil. In the meantime, in other British cities such as Birmingham, Glasgow, Liverpool and Nottingham it was possible to discover comparable tendencies (Evans 2005).

### 6.1.6 Subsidence

In coal mining areas subsidence can occur during and after extracting operations. When seams are exploited and the stabilizing, mostly wooden, construction starts failing, the emptied seams collapse and the overburden begins to fill the voids and holes abruptly. Consequently, the overlying strata subside from the bottom to the top until visible damage to buildings and bridges can be observed (Figs. 6.6 and 6.7).

In mining areas, dewatered aquifers resulted sometimes in abrupt sinkhole development, as observed, for instance, in Transvaal, South Africa. In Huaibei (China) with its 1,010,000 inhabitants, one of the most important coal mining territories in China, several subsided areas arose and the total subsided area reached a size of more than 100 km$^2$. It has been found that the subsided area will add 0.11 km$^2$, when 10,000 t coal are mined. The subsidence led to a soil surface collapse of up to 8 m. Since some mines are abandoned and hence it was possible to end

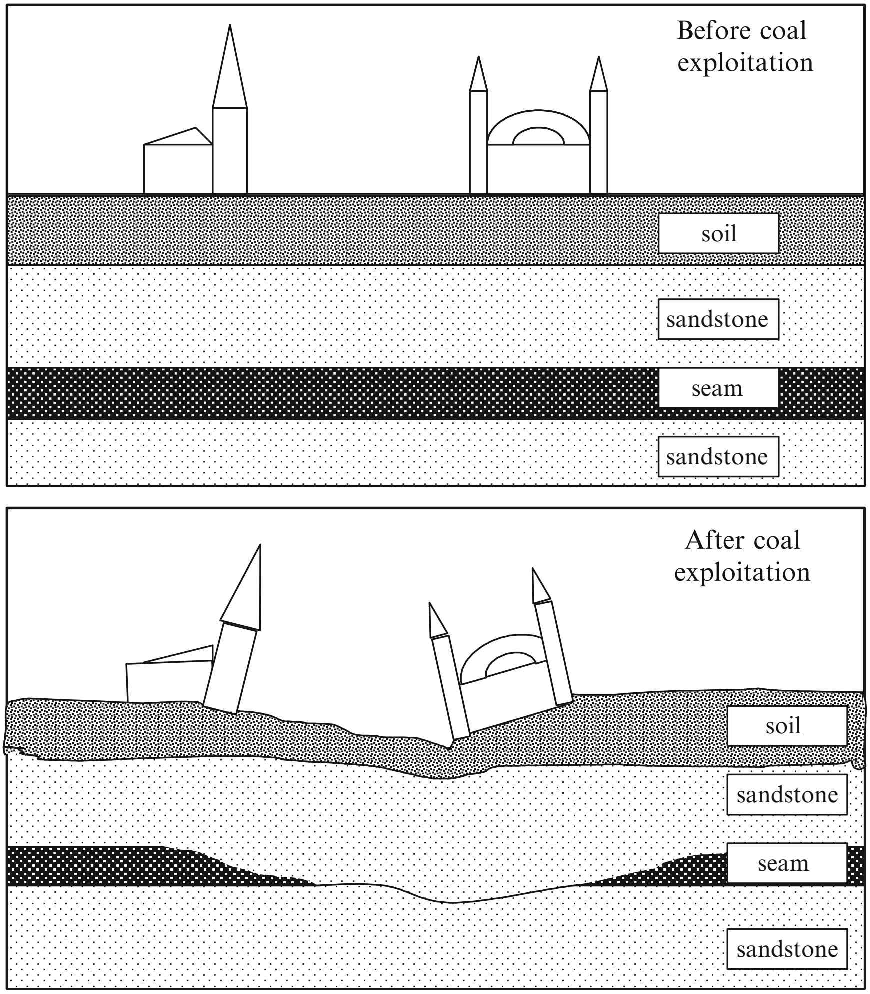



**Fig. 6.6** Subsidence processes caused by emptied seams

**Fig. 6.7** Building damage caused by subsidence of a coal mining area in Gelsenkirchen, Germany

groundwater pumping, the groundwater table raised continuously, so that the distance between land surface and rising groundwater level decreased in the last decade. In the meantime, in large areas the groundwater table exceeded the land surface, creating artificial lakes aboveground (Figs. 6.8 and 6.9). The entire landscape changed, agriculturally used sites were transformed into water bodies serving as aquatic parks and fishponds. In fact, farmers became fishermen in the course of time (Xu and Yan 2007). Ecologically, the creation of lakes without an overlaying of groundwater protective soil meant disadvantages. Industrial dust deposition and effluents were able to get into direct contact with the water body, residues of agrochemicals were discharged into the water area. Accordingly, some pollutants showed higher values in the artificial water bodies than in natural headwater regions, e.g. As, Cd, Cu, Pb, Se, Zn and mercury were higher by two to three orders of magnitude (Xu et al. 2007).

In some city areas, where basically water consumption is supposed to be high, groundwater abstraction occurred, causing compaction of the overlying increasingly dewatered unsaturated zone in fine-grained rocks. The hydrologically determined compaction in combination with the natural pressure of the overburden and the artificial load of the buildings was able to cause subsidence. Subsidence in urban environments is responsible for damage to buildings, roads and bridges, damage to sewers, water pipes, buried cables and other underground facilities, changes in the gradients of canals and flooding of low-lying areas including coastal areas. Features like damage to constructed facilities were observed in the capital of Mexico, Mexico City, with a population of 8,600,000, where approximately 9 m of subsidence has occurred (Evans 2005).

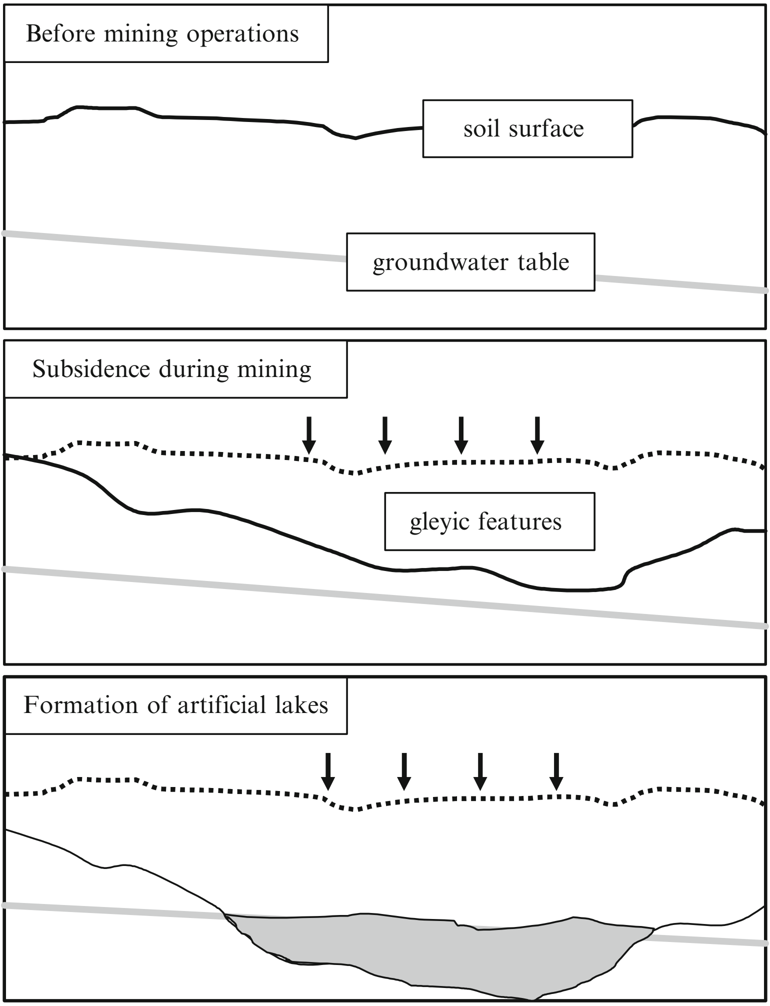



**Fig. 6.8** Development of artificial lakes in mining areas due to massive subsidence

## 6.2 Chemical Properties

### *6.2.1 Total Concentration: Methodological Aspects*

Most of heavy metal analyses are based on total concentration performed by aqua regia or $HNO_3$ extraction means. A good estimate of the more easily soluble fractions is probably more important from an environmental viewpoint than the determination of the total content. This is well known for soils in agricultural and horticultural environment.

**Fig. 6.9** Continuously progressing water logging and development of an artificial lake caused by subsidence in a coal mining area in Huaibei, China

In urban areas, soils are predominantly used for recreational and residential purposes and secondly used for food production. In particular, children are able to stay and play and are thus exposed to health risks. The risk becomes significant through direct contact, ingestion or inhalation. Taking the direct consumption of soil (hand-to-mouth behaviour) into consideration, the total concentration gets surely more importance than the soluble fractions (see Section 7.3.3).

The leachable concentration cannot be detected based only on total, aqua regia extractable concentration. Thus, to assess the risk of groundwater contamination it is not sufficient to analyze the aqua regia concentration only, since a small proportion of heavy metals are leachable (Alloway 1995). Section 6.2.4 gives information about the different binding forms in soils influencing metal mobility.

Related to the main pathways soil – plant and soil – groundwater other extraction methods appear to be more interesting. To estimate the more easily removable metals in soils and sediments complexing agents such as ethylenediaminetetraacetic acid (EDTA) or diethylenetriaminepentaacetic acid (DTPA), dilute acids like HCl or $CH_3COOH$ and neutral salts namely $CaCl_2$, $NaNO_3$, $NH_4NO_3$ and Na acetate as well as demineralised water are preferred. Furthermore, sequential extraction based, for instance, on the Community Bureau of Reference (BCR) protocol with various solutions of progressively stronger ability to dissolve metals can be a way to estimate metal contamination.

But such alternatives are not applicable to any metal in urban soils. Comparative investigations in urban topsoils of Seville (Spain) with 700,000 inhabitants showed that some metals gave results with the dilute HCl method that were very similar to the results for removable elements based on the BCR protocol but many other metals did not show any correspondence between the results obtained by both methods. Additionally, EDTA extraction revealed good correlation with the sequential or HCl methods for metals like Cd, Pb and Zn. In summary, EDTA extraction clearly overestimated the amounts of metals dissolved by both sequential or HCl extractions (Ruiz-Cortés et al. 2005). In contrast, the water-soluble fraction seemed to underestimate the heavy metal risk, since the analytic method (ratio soil: water, disturbed samples collected, short contact time applied) is not usually related to the in situ conditions.

The relatively low percentage of mobile fractions in comparison with the total heavy metal content was described in a Romanian research study. Research was carried out on urban soils of two large cities in Romania, Bucharest the capital, with 2,000,000 inhabitants, located in a plain region, and Baia Mare with a population of 140,000, located in an extra-mountainous depression. Sources of heavy metal pollution were a few metallurgical factories and motor vehicles emissions in Bucharest and a strong extractive and processing non-ferrous ore industry (flotation and smelting) in the Baia Mare area. The results of the topsoils investigated are presented in Table 6.5.

The predominant neutral – slightly alkaline reaction of the Bucharest urban soils determined a low mobilization of these chemical elements in the EDTA solution, while the predominant acid reaction of the Baia Mare urban soils determined the solubilization of medium quantities of 2.44 mg Cd $kg^{-1}$, 79 mg Cu $kg^{-1}$, 755 mg Pb $kg^{-1}$ and 138 mg Zn $kg^{-1}$ (Lacatusu et al. 2005).

A study conducted in the Ruhr area, Germany, took the EDTA extractable percentage of the total heavy metal content based on different land-use types into consideration. As shown in Table 6.6, the element cadmium revealed the highest

**Table 6.5** Total and EDTA extractable concentration of topsoils (mg $kg^{-1}$) in two Romanian cities (Data from Lacatusu et al. 2005)

| | | Cd | Cu | Pb | Zn |
|---|---|---|---|---|---|
| Bucharest | Total concentration (aqua regia) | 1.8 ± 0.5 | 32 ± 11 | 61 ± 45 | 132 ± 47 |
| | EDTA concentration | 0.6 | 9 | 24 | 37 |
| Baia Mare | Total concentration (aqua regia) | 5.2 ± 6.9 | 254 ± 222 | 1,380 ± 2,016 | 644 ± 517 |
| | EDTA concentration | 2.4 | 79 | 755 | 138 |

**Table 6.6** EDTA extractable heavy metal percentage of the total concentration (aqua regia) (%) in soils (depth 3–10 cm) with reference to different land-use types in the Ruhr area, Germany (Data from Hiller and Meuser 1998)

| Land-use type | pH status | Cd | Cu | Ni | Pb | Zn |
|---|---|---|---|---|---|---|
| Coal mining heap | acid | 24 | 16 | 16 | 18 | 9 |
| Railway embankment | neutral/alkaline | 31 | 5 | 17 | 32 | 14 |
| Collieries and coal processing industry | neutral/alkaline | 49 | 17 | 25 | 31 | 19 |
| Iron and steelworks | alkaline | 9 | 1 | 7 | 14 | 9 |
| Gardens, parks, recreational sites | neutral/alkaline | 50 | 8 | 24 | 38 | 18 |

concentration in the upper, humus-rich, part of the soil analysed by EDTA that is strongly related to the relatively mobile metal-organic complexes in neutral to alkaline soils. Zinc, in second place, demonstrated enhanced EDTA-extractable amounts. Nickel and copper had low values, but lead, usually estimated as immobile in soils, revealed surprisingly high percentages. However, it can be concluded that Pb might prefer to form metal-organic complexes in neutral to alkaline soil conditions. In summary, the order of EDTA extractable percentages was Cd > Pb > Zn > Ni > Cu.

In relation to humus-enriched anthropogenic horizons it was not possible to distinguish a clear pH influence, since the anthropogenic soils with low pH values like hard coal mining heaps were not significantly different from the sites with higher pH values like iron and steelworks soils, where the values varied from 6.5 to 8.5, in some examples even reaching 10. Only some elements tended towards reduced EDTA extractable values in acid conditions (Hiller and Meuser 1998).

Based on the sequential extraction method the fractionation of the total content of Cd, Cu, Pb and Zn of the urban soils from the two Romanian localities mentioned above highlighted the Cd predominance in the soil solution and in the exchangeable fraction, the predominance of Cu and Zn in the fractions bound by the organic matter in the Bucharest urban soils and by the organic matter and sesquioxides in the Baia Mare urban soils. The lead content was relatively uniformly distributed between the soluble fraction, the exchangeable one and the fraction bound by the organic matter (Lacatusu et al. 2005).

In general, the sequential extraction method can explain the binding capacities in urban soils in more detail. The chemical characteristics, e.g. the presence of carbonaceous substrates and iron oxides containing substances, might be identified by sequential extraction, as seen in a Polish study. The sequential metal extraction approach was conducted in association with five allotment garden soils in Inowrocaw, Poland, with a population of 80,000. The city is impacted by numerous anthropogenic sources such as the local industry, fossil fuel combustion plants and motor vehicle emissions. The lead and zinc levels were generally enhanced in this city environment, exceeding background values considerably (Table 6.7). Zinc was distributed mainly among the residual part and the carbonate form, meaning practically low availability. In contrast, lead was retained by organic matter followed by the residual percentage. At the moment, both elements showed small amounts of available forms (Dabkowska-Naskret 2000).

This good result from a horticultural point of view should not blind us to the fact that long-term processes can alter the current situation. Organic matter, for instance, is possible to convert the metal complexes and increasing acidification can take part in calcium decrease, ultimately eliminating the carbonatic metal fractions (see Section 6.3).

High values were only detectable, if the total concentration exhibited extremely high values, as at present in heavy metal mining areas. For instance, in the southern area of Morocco, which is considered a traditional mining region since antiquity and which is estimated to contain large reserves of iron, copper, zinc, silver and lead. Tailings and soils were collected from five metal mines partly abandoned

**Table 6.7** Mean concentrations (mg kg$^{-1}$) of lead and zinc analyzed in topsoils (0–5 cm) by sequential extraction method for soils of five allotments in Inowrocaw, Poland (Data from Dabkowska-Naskret 2000)

| | Pb | Zn |
|---|---|---|
| Exchangeable | 0.0 | 1.8 |
| Associated with carbonates | 0.9 | 34.6 |
| Associated with Mn oxides | 1.7 | 4.7 |
| Associated with iron oxides (amorphous) | 7.6 | 3.5 |
| Associated with iron oxides (crystalline) | 8.3 | 4.1 |
| Organic complexes | 21.8 | 15.8 |
| Residual | 35.0 | 91.0 |
| Total concentration (aqua regia) | **75** | **155** |

and partly still in activity. Soil pH was variable, ranging from very acidic (pH 2.6) to alkaline values (pH 8.0–8.8). The tailings from polymetallic mines contained very high total concentrations (aqua regia) of Cd (148–228 mg kg$^{-1}$), Cu (2,019–8,635 mg kg$^{-1}$), Pb (20,412–30,100 mg kg$^{-1}$) and Zn (38,000–108,000 mg kg$^{-1}$). The same approach was made on the aqueous extracts of solid materials in order to assess the potential availability of heavy metals. Water-extractable metal concentrations were lower but, as shown by a microbial biotest, highly toxic. Sites indicating very high toxicity by biotest showed high water soluble concentrations of Cd (2.0–2.7 mg L$^{-1}$), Cu (1.8–82 mg L$^{-1}$) and Zn (785–1,753 mg L$^{-1}$), respectively (Boularbah et al. 2005).

A study dealing with a lead and zinc mining area in Mezica (Slovenia) with a population of 4,000 was able to confirm the high mobile metal concentration, if the total concentration showed extremely high values. Meadow and garden soils (depth 0–5 cm) close to the mine were sampled and analysed indicating aqua regia extractable concentrations of 6.3–35.0 mg Cd kg$^{-1}$, 778–9,775 mg Pb kg$^{-1}$ and 315–1,505 mg Zn kg$^{-1}$. In contrast to the natural uncontaminated soils in the same catchment, where Cd was mainly bound to oxides, Pb to the organic matter, sulphides and silicates as well as Zn predominantly to silicates only, in the heavily contaminated mining soils the partitioning was generally related to both easily and sparingly soluble fractions. However, with increasing total concentration, e.g. in soils close to the smelter, the easily soluble percentage was enhanced and reached 80% (Cd), 50% (Pb) and 70% (Zn) according to the BCR sequential three-step extraction method applied (Putisek et al. 2001).

As an alternative to the estimation of the water-soluble concentration, the Dynamic Batchtest can be an example for leaching of soil columns under water-saturated conditions over 24 h. The availability factors are simultaneously measured in order to control whether equilibrium conditions exist or not. This is of importance, because preferentially pH-value and redox potential influence the heavy metal mobility (van der Sloot 1996). The shaking techniques usually applied are far away from field conditions if one considers the wide liquid-solid-ratios, mechanical stress during shaking and the process of permanent liquid and solid phase movement (Makowsky and Meuser 2008).

The example of garbage incinerator ash proves the remarkably high values for Cu, Pb and Zn (Table 5.7). Garbage incinerator ash, however, is mainly characterized by leaching of Cu and, furthermore, Cr, Ni and Zn. Despite extremely high total concentrations of Pb water-soluble Pb is not of concern, although the water-soluble content (μg $L^{-1}$) is influenced by aqua regia concentration (mg $kg^{-1}$) (Table 6.8).

Therefore, it is difficult to conclude a relationship between increasing water soluble metals and increasing aqua regia concentrations. Facing the relationship of the extraction means aqua regia and water the increasing aqua regia content does not automatically tend to be accompanied by a rising water-soluble content. As shown in Fig. 6.10, that presents simultaneous results from different technogenic materials such as construction debris, ashes, garbage as well as mining and industrial waste, the low relationship was generally found.

It has been found that the different leaching behaviour can be observed in a number of substances. For example, in a laboratory batch test the total Zn concentration of coal mining waste reached 42 mg $kg^{-1}$, whereas household waste indicated a very high level of 531 mg $kg^{-1}$. The quotient of water-soluble concentration divided by aqua regia content, however, was 1.53‰ for coal mining waste and 0.26‰ for household waste. Further substrate-related quotients (soluble concentration divided by total concentration) are presented in Table 6.9 (Makowsky 2009).

In general, the carbonate buffer might compensate for the decrease of pH in case of construction debris and ashes. The reason for the relatively high Cr mobility level of construction rubble was surely the formation of mobile chromate. Water-soluble chromium can be raised in spite of a low total concentration. The parameter shows the influence of availability factors, primarily the pH value (van der Sloot et al. 1996). Under alkaline conditions hydroxide complexes are supposed to be responsible for the mobility of some metals like Cr and As. Moreover, increased Cr mobility can be explained by anionic chromates ($Cr(OH)_3^{2-}$) at high pH- and Eh-values. In reductive conditions the chromate mobility is reduced due to the precipitation of $Cr(OH)_3^{2-}$. In general, the mobility increases at low pH values, if the metals are cationic (Alloway 1995; van der Sloot 1996).

The low pH value of coal mining waste was responsible for the moderate to high mobility of Cu and Zn in spite of low total concentration. Although the total concentration of the industrial blast furnace sludge was extremely high for some elements, the immobile character dominated because of carbonate, phosphate and oxide bindings. Ultimately, lead was, apart from dredged sediments, permanently immobile, since the pH value did not fall below pH 4.

The effect of redox potential on mobility was visible in the case of dredged lake sludge. Aerobic conditions enabled oxidation of former immobile sulphides. Consequently, acidification occurred, resulting in a high mobility of Cd, Cu, Ni and Zn. The acid pH value in combination with dissolved organic matter (DOC) was responsible for the mobility. Based on a batch leaching test using demineralised water very high quotients were found for nearly all trace elements (e.g. Cd: 285‰, Cu: 28.7‰, Ni: 174‰, Pb: 37.4‰, Zn: 331‰) (Makowsky and Meuser 2008; Makowsky 2009).

**Table 6.8** Mean values and standard deviations of heavy metals (aqua regia and water soluble concentration) as well as quotient water soluble content divided by aqua regia content for three bottom ashes (Data from Makowsky 2009)

| Substrate | Ash origin | Cd | Cr | Cu | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|
| Aqua regia content (mg $kg^{-1}$) | | | | | | | |
| Bottom ash | waste | 4.3 ± 1.4 | 84 ± 21 | 1,569 ± 341 | 69 ± 12 | 830 ± 102 | 1,649 ± 24 |
| Bottom ash | coal | 2.3 ± 0.2 | 65 ± 3 | 511 ± 32 | 89 ± 4 | 815 ± 39 | 975 ± 63 |
| Bottom ash | coal | 2.2 ± 2.6 | 40 ± 1 | 266 ± 34 | 83 ± 5 | 1,201 ± 1,228 | 526 ± 93 |
| Water soluble fraction (batchtest) (μg $L^{-1}$) | | | | | | | |
| Bottom ash | waste | nd | 28 ± 2 | 981 ± 42 | 12 ± 1 | 3 ± 2 | 69 ± 24 |
| Bottom ash | coal | nd | 1 ± 1 | 11 ± 3 | nd | 3 ± 2 | 25 ± 5 |
| Bottom ash | coal | 1 ± 1 | 8 ± 2 | 7 ± 2 | nd | nd | 24 ± 9 |
| Quotient water soluble fraction/aqua regia content (‰) | | | | | | | |
| Bottom ash | waste | <0.01 | 0.34 ± 0.09 | 0.71 ± 0.13 | 0.18 ± 0.05 | <0.01 | 0.04 ± 0.02 |
| Bottom ash | coal | <0.01 | 0.02 ± 0.02 | 0.04 ± 0.01 | <0.01 | 0.01 ± 0.01 | 0.01 ± 0.01 |
| Bottom ash | coal | 2.06 ± 1.46 | 0.36 ± 0.11 | 0.05 ± 0.02 | <0.01 | <0.01 | 0.09 ± 0.04 |

nd = not detectable

Analysis conducted by dynamic batchtest

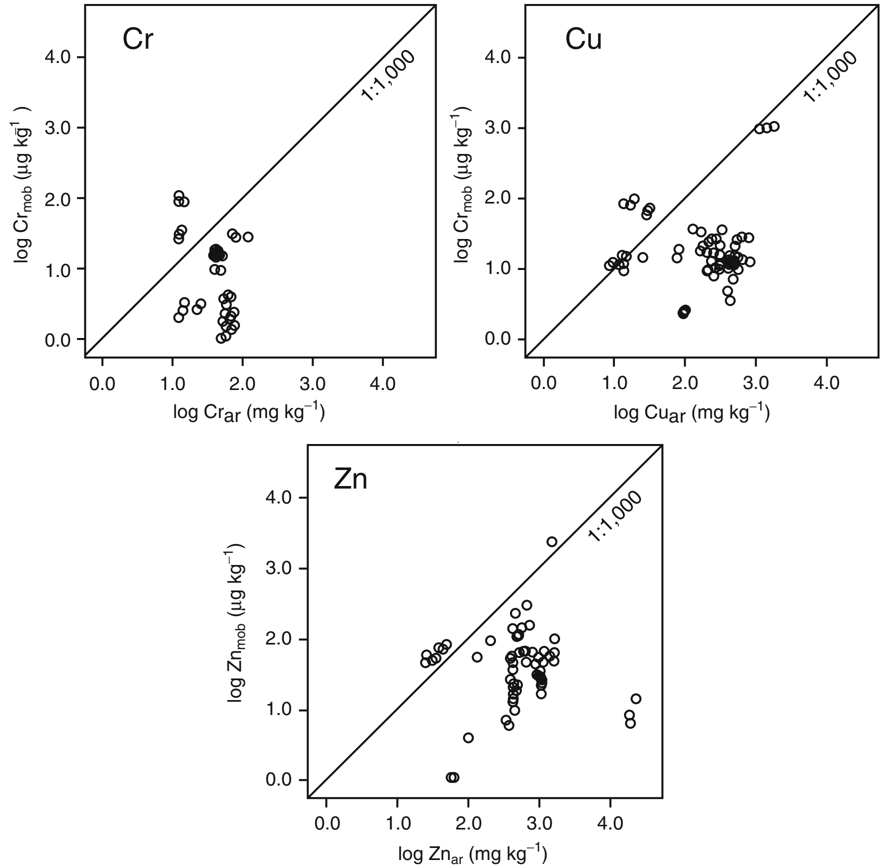



**Fig. 6.10** Scatter plot of mobile (mob) versus aqua regia (ar) extractable content of the heavy metals Cr, Cu and Zn of different technogenic substrates. Metal extraction method: water and artificial rainwater (mob), aqua regia (ar), operated by dynamic batchtest (Data from Makowsky 2009)

## *6.2.2 pH Value*

The elevated pH value of urban soils is commonly related to the presence of building rubble containing brick, cement, plaster, mortar and concrete. The alkaline substrates tend to be weathered, releasing calcium. In dry conditions the released calcium is responsible for cementation at soil surface, creating an impermeable crust (Craul 1992).

In general, the presence of alkaline substrates such as construction debris, slag and ashes in soils means raised soil pH values in urban areas. Investigations of several deposited soils in Germany, taking highly contaminated soils with techno-genic substrates into account, were presented in Table 6.9. Apart from the coal

**Table 6.9** Heavy metal mobility (quotient water soluble concentration divided by total concentration in ‰), aqua regia extractable heavy metal concentration in mg $kg^{-1}$ and pH value for different soil horizons with technogenic monosubstrates (n = 4) (Data from Makowsky 2009)

| Substrate | | Cd | Cr | Cu | Ni | Pb | Zn | pH |
|---|---|---|---|---|---|---|---|---|
| Construction debris | Mobility (‰) | 0.00 | 2.59 | 2.25 | 0.06 | 0.00 | 0.01 | 10.4 |
| | Aqua regia | nd | 12 | 31 | 9 | 25 | 101 | |
| Bottom ash | Mobility (‰) | 0.05 | 0.35 | 0.04 | 0.00 | 0.00 | 0.05 | 7.8 |
| | Aqua regia | 0.8 | 43 | 338 | 89 | 581 | 470 | |
| Coal mining waste | Mobility (‰) | 0.00 | 0.05 | 0.84 | 0.06 | 0.00 | 1.53 | 5.5 |
| | Aqua regia | nd | 19 | 13 | 11 | 62 | 42 | |
| Household waste | Mobility (‰) | 0.00 | 0.00 | 0.09 | 0.08 | 0.00 | 0.26 | 7.7 |
| | Aqua regia | 1.4 | 47 | 224 | 73 | 467 | 531 | |
| Blast furnace works | Mobility (‰) | 0.00 | 0.00 | 0.01 | 0.0 | 0.00 | 0.00 | 8.9 |
| sludge | Aqua regia | 36.5 | 66 | 100 | 0.08 | 7,512 | 20,627 | |

nd = not detectable
Analysis conducted by dynamic batchtest

mining waste, the substrates in question showed a neutral to alkaline reaction (Makowsky 2009). Moreover, the technogenic deposits topsoils are mostly influenced by the alkaline dust deposition derived from cement and concrete weathering as well as vehicle transportation in the city landscape.

In Lodz (Poland) with a population of 1,000,000 the pH($H_2O$) values were systematically measured in a field study including 63 soil profiles of lawns (beside roads) and parks to a depth of 150 cm. In summary, 46% of the soils revealed a neutral reaction (pH 6.6–7.2) and 39.5% an alkaline reaction (pH > 7.2). Weakly acid soils (pH 5.5–6.5) covered 11.5% of the surface, while acid soils (pH below 5.5) covered merely 3% of the city only (Czarnowska et al. 2000). In a number of other studies the potentially neutral to alkaline reaction was confirmed. The pH($H_2O$) values of roadside soils in Hong Kong (China) with 7,300,000 inhabitants ranged from 6.8 to 10.0 (Jim 1998), street tree pits in New York City (USA) with 23,200,000 inhabitants showed pH ($H_2O$) values of up to 8.8 (Pouyat et al. 2007) and residential soils of Ibadan (Nigeria) with 5,200,000 inhabitants exhibited pH($CaCl_2$) values from 5.1 to 7.6 (Pouyat et al. 2007).

The increased pH value associated with the presence of calcium carbonate in soil profiles can be advantageous, because the acid neutralisation capacity may increase accordingly. Investigations of 60 profiles in the Ruhr area, Germany, showed pH ($CaCl_2$) values >7.0 and calcium carbonate content of >1% in the upper 30 cm of most profiles. Calculations taking the pH value of precipitation in the area into account produced the result that the $CaCO_3$ potential will be consumed in several hundred years, before soil acidification might begin. Thus, with regard to the high acid neutralisation capacity, enhanced migration of heavy metals cannot occur in the near future (Hiller 2000).

Consequently, in soils of the Ruhr area using $NH_4NO_3$ to determine the available percentage of heavy metals, it has been found that deposited soils consisting of calcareous technogenic substrates did not leach elements like Cd, Cu, Ni and Pb downwards to a greater extent. There, natural soils of strongly acidified urban forests revealed higher metal leaching than urban technogenic soils, although the total concentration was much higher in the anthropogenic soils (Hiller and Meuser 1998).

The estimation of pH-dependent mobile heavy metal content can be conducted with different extraction solutions. One extraction method indicating a close connection to the mobility of metals is based on ammonia acetate ($NH_4$ac) agent. In summary, cadmium revealed the highest portions of mobility in most of the substrates (Table 6.10). It is followed by nickel and zinc with moderate results, while copper and particularly lead frequently did not indicate high metal mobility. With reference to the substrates the order of mobility will be designated construction rubble > household waste > sludges > ashes and coal mining waste. The differences between the substrate components should be noted, if the mobility is included in risk assessment. Chemical analyses using the Freundlich isothermes confirmed the tendencies, because the K constants for Cu and Pb were the highest ones. In a comparison of the substrate components ashes had an extreme tendency to adsorb metals, while construction debris, sludges and waste demonstrated insignificant results.

**Table 6.10** Mobile heavy metal percentage of the total concentration (aqua regia) (%) based on investigation of deposited soil horizons (n = 43, sampling depths 0–150 cm) in different cities in Germany (Data from Blume and Schleuss 1997)

| Element | Substrate | pH 3 | pH 4 | pH 5 |
|---|---|---|---|---|
| Cd | Construction debris, household waste, dredged sludges | 60 | 30 | 30 |
| | Ashes, sewage sludge | 6 | 3 | 3 |
| Cu | Independent of substrate | 6 | 3 | 3 |
| Ni | Independent of substrate | 20 | 10 | 5 |
| Pb | Independent of substrate | 6 | 3 | 3 |
| Zn | Construction debris | 100 | 50 | 10 |
| | Household waste, dredged sludges | 40 | 20 | 10 |
| | Ashes | 20 | 10 | 2 |
| | Sewage sludge | 5 | 3 | 2 |

Mobile metal extraction method: $NH_4$ acetate

The impact of the pH value on heavy metal mobility has also been observed. The increasing mobility with decreasing pH was in the line with natural soils but substrate-specific differences played an important role, too. In the presence of construction rubble and household waste the mobile percentage was frequently enhanced (Blume and Schleuss 1997).

### 6.2.3 Carbon Content and Biological Activity

In order to estimate heavy metal mobility in topsoils, apart from the pH value the humus content has to be included. The sorption behaviour can be altered in the presence of technogenic carbon (see Section 4.2.2). For instance, affinities between Polycyclic Aromatic Hydrocarbons (PAH) and soot have been reported. The $K_{OC}$, used as Freundlich coefficient considering the organic carbon distribution, was very high for thermally altered coals. In general, the order activated carbon (filter technique material), charcoal and lignite coke, sub-bituminous coal, lignite and peat were found (Table 6.11). There was a tendency for the polar constituents to decrease from the natural material to the technogenic products. The aromaticity of technogenic substrates, however, was mostly the highest. The latter seemed to be responsible for the enhanced sorption coefficient $K_{OC}$, since a positive correlation between both aromaticity and $K_{OC}$ was found. Exceptionally, the coal mining waste did not follow the tendencies described. Nevertheless, in the presence of technogenic carbon a higher sorption capacity of soils, particularly for organic pollutants, can be expected (Abelmann et al. 2005).

It is generally difficult to evaluate the mobility of organic pollutants in urban environments. The adsorption and desorption potential may depend on several factors. In particular, the quality of technogenic components present in deposited soils

**Table 6.11** Carbon content, polarity, aromaticity and sorption coefficient $K_{OC}$ for phenanthrene (Polycyclic Aromatic Hydrocarbon) of different substrates and topsoils (Data from Abelmann et al. 2005)

| | C total (%) | Polarity | Aromaticity | log $K_{OC}$ (mg $kg^{-1}$) |
|---|---|---|---|---|
| Substrates | | | | |
| Activated carbon | 89.3 | nd | nd | 6.35 |
| Lignite coke | 88.0 | 0.00 | 1.00 | 5.34 |
| Charcoal | 81.7 | 0.10 | 0.87 | 5.50 |
| Sub-bituminous coal | 55.1 | 0.43 | 0.37 | 5.12 |
| Lignite | 38.9 | 0.48 | 0.31 | 5.22 |
| Sapric Histosol (peat) | 30.7 | 1.17 | 0.16 | 5.13 |
| Topsoils | | | | |
| Luvisol | 1.1 | 1.85 | 0.12 | 4.58 |
| Chernozem | 1.7 | 1.19 | 0.27 | 4.20 |
| Regosol (lignite ash) | 18.5 | 0.34 | 0.55 | 4.74 |
| Regosol (hard coal) | 9.1 | 0.46 | 0.55 | 5.00 |
| Regosol (coal mining waste) | 14.1 | 0.34 | 0.68 | 4.15 |

nd = not detectable

may make it difficult to estimate the availability and mobility of organic contaminants. However, according to the group of Polycyclic Aromatic Hydrocarbons, it has been found that with increasing PAH concentration the mobile percentage rose simultaneously. Based on batch test experiments taking several deposited soil horizons in different localities in Germany into account, soils consisting of technogenic substrates might release PAH, particularly if construction debris containing soils have been influenced by long-term wetting (Blume and Schleuss 1997).

In relation to the humus its content is of importance with reference to the biodegradation potential of organic pollutants. The potential biodegradation in urban soils should be integrated into soil evaluation. In the Sofia – Pernik district, Bulgaria, natural soils (predominantly vertisols) and anthropogenic soils alongside roads and in a large mining centre were investigated for comparison purposes of microbiological activity that may seem to be a good indicator for the biodegradation potential. The results of natural soils and the urban soils influenced by heavy traffic were comparable. There was a tendency for technogenic urban soils of the mining area to exhibit lower values but it depended on the age of the soils, since the unvegetated, recently deposited soil was the only one with strongly reduced microflora content (Table 6.12) (Noustorova et al. 2000).

In soils in city parks with humus development biological activity potentially capable of degradation of organic pollutants was noticeable. It should be noted, however, that the biological activity is restricted to the humus-enriched upper layers. In Lviv (Ukraine) with its 740,000 inhabitants results from the upper 10 cm of several city parks and lawns conducted in June showed adequate carbon content of 1.7% and neutral pH values. In comparison with anthropogenically undisturbed forest park soils (C content 2.3%, pH($H_2O$) 4.7) some enzyme activity parameters

**Table 6.12** Average chemical and biological soil characteristics of natural and urban soils in the Sofia – Pernik district, Bulgaria (Data from Noustorova et al. 2000)

| Soil group | Depth (cm) | Total microflora ($10^3$ $g^{-1}$) | Humus (%) | Total N (%) | pH ($H_2O$) |
|---|---|---|---|---|---|
| Natural soils (n = 6) | 0–20 | 3,815 | 4.9 | 0.20 | 7.3 |
| | 20–40 | 3,937 | 3.6 | 0.14 | 7.1 |
| Urban soils (n = 6) | 0–20 | 4,626 | 3.4 | 0.13 | 7.4 |
| | 20–40 | 2,601 | 2.8 | 0.11 | 7.2 |
| Mining soils (n = 3) | 0–20 | 2,270 | 3.4 | 0.27 | 7.7 |

of the anthropogenic soils (saccharase, urease, catalase) were much higher. However, the microbial biomass carbon indicated values of 183 (forest-park), 175 (city park) and 66 (lawns) μg $g^{-1}$ and the substrate-induced respiration (SIR) 3.9 (forest-park), 3.7 (city park) and 1.4 (lawns) μg $g^{-1}$ $h^{-1}$ (Maryskevych and Shpakivska 2000).

Different soils deposited in German urban environments were assessed in relation to the microbiological parameters. The sites, including deposits consisting of rubbish, ashes, sewage sludge and mining spoil, were located in parks, rehabilitation areas and on former industrially used fallow land. Maximum microbiological carbon up to 1,141 μg $g^{-1}$ was found in the nutrient-enriched A horizons of anthropogenic soils and on sewage sludge fields within the entire profile. There the values were higher than in undisturbed natural soils. In contrast to these sites, the microbial biomass in mine material and ashes was relatively low, ranging from 158 to 225 μg $g^{-1}$. In general, the microbial carbon and other biological variables were low in deeper horizons, in particular deeper than 15 cm.

Moreover, the metabolic quotient $qCO_2$ was observed together with the dehydrogenase activity. Both parameters revealed increased values in association with sewage sludge, while construction debris, mine spoil and ashes showed low to moderate results (Table 6.13). The study indicated that some microbiological features such as microbial biomass, soil respiration as well as enzyme activity were developed to a remarkable degree. In spite of the unfavourable soil conditions like strongly alkaline pH value and salt accumulation it was obviously possible for microbes to survive. However, only humic topsoils and nutrient-enriched, deposited substrates revealed complete values comparable with natural soils (Machulla 2000). With respect to the biodegradation of organic pollutants a slow natural attenuation can be expected.

Both bacteria and fungi take part in biodegradation processes. It is well-known that in acidified forest soils the biological activity is usually oriented towards fungi species. But also in urban areas potentially pathogenic and allergenic microfungi exist, developing an enhanced risk situation. As observed in different Russian cities, the living conditions of problematical fungi with higher air and soil temperatures and alkaline soil pH values in urban environments were improved, leading to a tendency for some species like *Aspergillus, Fusarium* and unspecified dark-coloured microfungi to grow preferably. For example, there was a greater abundance of the latter in urban territory, in particular in the roadside zones. They are known

**Table 6.13** Ranges of metabolic quotient ($qCO_2$), dehydrogenase activity (DHA) and some chemical properties in deposited soil horizons in Germany (Data from Machulla 2000)

| Subtrate | $qCO_2$ (µg $CO_2$-C $mgC_{mic}^{-1}$ $g^{-1}$ $h^{-1}$) | DHA (µg TPF $g^{-1}$ $24h^{-1}$) | pH ($CaCl_2$) | Carbon (%) | C/N |
|---|---|---|---|---|---|
| Rubbish (n = 14) | 1.9–3.5 | 73–175 | 6.8–7.7 | 0.5–3.3 | 11–29 |
| Ashes (n = 7) | 1.7–14.0 | 19–39 | 6.5–7.8 | 2.9–21.0 | 63–124 |
| Mine spoil (n = 7) | 3.7–9.3 | 14–47 | 2.7–5.1 | 2.3–9.6 | 14–80 |
| Sewage sludge (n = 5) | 12.4–23.7 | 18–237 | 7.1–7.5 | 4.5–9.4 | 9–48 |

to contain melanin pigments in their cell walls, causing allergic reactions. Results revealed clear increase of potentially pathogenic microfungi in cities investigated. The average abundance of the microorganisms in Moscow research sites increased from 40% to 47% and in some smaller towns (Labytnangi, Kandalaksha, Pushchino, Serpuchov) this was between 11% and 38% (Marfenina 2000).

### *6.2.4 Texture and Binding Compounds*

The texture class influences the contamination, since fine earth can adsorb more heavy metal cations than the coarse fractions. Therefore, some technogenic substrates consisting of silt or clay such as sludges and fly ash might be more polluted (see Section 4.3.1).

Apart from organic matter content, technogenic carbon and fine earth, oxides are of importance with reference to the sorption potential of anthropogenic soils. In urban soils some iron compounds are present that are normally rarely found to a great extent. In the anthropogenic soils of the Ruhr area, Germany, magnetic iron oxides such as magnetite ($Fe_3O_4$) and maghemite ($\gamma$-$Fe_2O_3$) which might take part in heavy metal sorption were discovered. In isolated ashes up to 300 g $kg^{-1}$ were detected and anthropogenic soils of metal works in Oberhausen with 220,000 inhabitants and Duisburg with 490,000 inhabitants (Germany) resulted in values ranging from 123 to 168 g $kg^{-1}$ (Hiller and Meuser 1998). Magnetic particles are closely connected with trace elements and the correlation between magnetic susceptibility and heavy metal content was strong. Consequently, in several studies high values of magnetic reaction were found in areas with high metal concentration (see Section 3.2.2) (Kapicka et al. 2000; Magiera and Strzyszcz 2000).

In urban soils several chemical compounds such as carbonates, sulphides and phosphates are responsible for reduced solubility and availability of heavy metals (Liu et al. 2005). Due to their origin in construction rubble, slag and ashes are calcium carbonate-rich and alkaline. Carbonate bindings are suspected to be further reasons for a low solubility of heavy metals in soils. Soils indicating pH values >7.5 and free calcium produce immobile carbonatic compounds, particularly for Cd, Cu, Ni, Pb and Zn. An overview of heavy metal binding forms is shown in Table 6.14.

**Table 6.14** Overview of heavy metal binding forms

| Binding form | Soil properties | Mobility | Examples |
|---|---|---|---|
| Carbonates | $CaCO_3$-rich, pH > 7.5 | Immobile | $CdCO_3$, $Cu(OH)_2CO_3$, $NiCO_3$, $PbCO_3$, $ZnCO_3$ |
| Sulphides | Anaerobic | Immobile | CdS, $Cu_2S$, HgS, NiS, PbS, ZnS |
| Phosphates | Rich in phosphorus | Immobile | Cd-, Cu-, Ni-, Pb-, Zn-phosphates |
| Silicates | Ore-rich | Immobile | Every element |
| Oxides (occlusion) | Reductomorphic | Immobile | Every element |
| Organo-metallic complexes | Humus-rich | Mobile > pH 6.0 | Complexes of Cd, Cr, (Cu), Hg, Ci, Pb |
| Adsorptive binding | Clay-rich, humus-rich, oxides-rich | Mobile with decreasing pH | Every element |
| Miscellaneous | Accelerated soil temperature | Mobile (gaseous migration) | Hg-steam, $(CH_3)_2Hg$ |

Deposits containing household waste and sewage sludge exhibit anaerobic conditions in subsoils over long periods of time. The circumstances lead to the establishment of black-coloured and intensively smelling sulphides in soils, which are designated as further immobile binding agents and frequently related to Cd, Cu, Hg, Ni, Pb and Zn.

Phosphoric binding forms, in particular with reference to Cd, Cu, Ni, Pb and Zn, are present in humus-enriched soil horizons, e.g. deposited topsoils derived from agricultural areas, garden soils and plaggen soils, reducing metal mobility considerably, since phosphoric binding forms are also estimated to be completely immobile forms.

Furthermore, an immobile character is to be expected, if silicate bindings and occlusion into iron and manganese oxides occur. This form is not metal dependent and particularly common in ore-mining areas (silicates) and reductomorphic soils (oxides).

In contrast, adsorptive bindings can be mobilized in association with the pH value. The adsorption to clay minerals, organic matter content as well as iron and manganese oxides decreases with decreasing pH value. Generally increased metal mobility will be detectable, if the pH value falls below 6.5 but the order of pH-dependent mobility varies between the elements. With decreasing pH the metal mobility begins at <6.5 for Cd, <6.0 for Zn, <5.5 for Ni, <4.5 for Cu and Cr and <4.0 for Hg and Pb. Because of predominantly coarse-textured soils in the urban environment the adsorption potential to the mineral fraction appears to be relatively low. Adsorption to oxides might play an accelerated role due to partly magnetic oxides mentioned above.

The humus content behaves in a contradictory fashion. On the one hand, humus is responsible for a high adsorption potential of metals and organic pollutants. On the other hand, in the presence of dissolved organic matter, metal complexation occurs, enhancing the mobility of some metals. The complexes reveal high stability and mobility in the case of pH values exceeding 6.0. The stability reduces with decreasing pH, leading to complete dissolution in acid conditions. In the first instance, the elements Cd, Cr, Hg, Ni and Pb tend to form organo-metallic complexes, in the second instance Cu.

Mercury forms a specified organo-metallic complex called dimethyl-Hg ($(CH_3)_2Hg$), volatilising into the atmosphere. With increasing soil temperature the tendency of mercury to volatilise as Hg-steam or $(CH_3)_2Hg$ becomes higher.

### 6.2.5 Nutrients

The nutrient status of urban soils shows a high spatial variability. Garden soils, for instance, are mostly nutrient-rich, degraded soils of derelict land usually indicate low fertility and soils consisting of technogenic substrates vary depending on the kind of substrate present at site. The tendencies are linked to the organic matter content responsible for the nitrogen, phosphorus and sulphur supply to a great extent.

It should be mentioned that the assessment of nutrient supply in urban areas is problematical, since different extraction methods are applied, making comparison difficult. For instance, for plant available phosphorus assessment a wide range of extraction methods is used depending on the distinct soil substrates present in urban soils. However, bearing in mind the problems mentioned, tendencies can be discovered.

In China the high nutrient status of urban soils looking back on a long history of gardening has been confirmed. The urban soils in Nanjing with a population of 5,300,000 revealed total phosphorus concentrations between 1.3 and 11.4 g $kg^{-1}$, while the predominant parent material in Nanjing derived from the Holocene material of the Yangtze River only amounted to 1 g $kg^{-1}$ phosphorus. In particular, vegetable plots and school yards had enhanced values (Fig. 6.11a and b). The authors concluded that the ancient urban zones of the more than 2,400 years old city were used as vegetable gardens. Most of the garden land was transformed into residential and industrial areas during the subsequent city expansion, maintaining relatively high phosphorus content even in residential and industrial areas of today. In general, other nutrient parameters such as nitrogen and carbon (humus) reacted simultaneously. The high C/N ratio, however, measured in a number of garden soils in European cities, where coal ash was applied for improvement of the soil fertility purposes, has not been confirmed, since ash application did not normally take place. Instead, the gardeners used mainly organic manure.

The investigations in Nanjing resulted in a strongly disturbed depth gradient within soil profiles. While in agriculturally used fields the phosphorus content decreases with increasing depth, especially below approximately 30 cm, the analyzed content in the Nanjing suburbs indicated no clear tendency due to the rapid expansion process of the city (see Section 2.3) associated with continuous disturbances of the soil profiles. The disturbances (excavation, stockpiling, depositing) also led to a covering of humic fossil A horizons, indicating high phosphorus values in the subsoil (Zhang et al. 2001).

Systematically, the example of Nanjing clearly shows the urbanization process and its impact on nutrient status. Total and available P was predominantly enriched

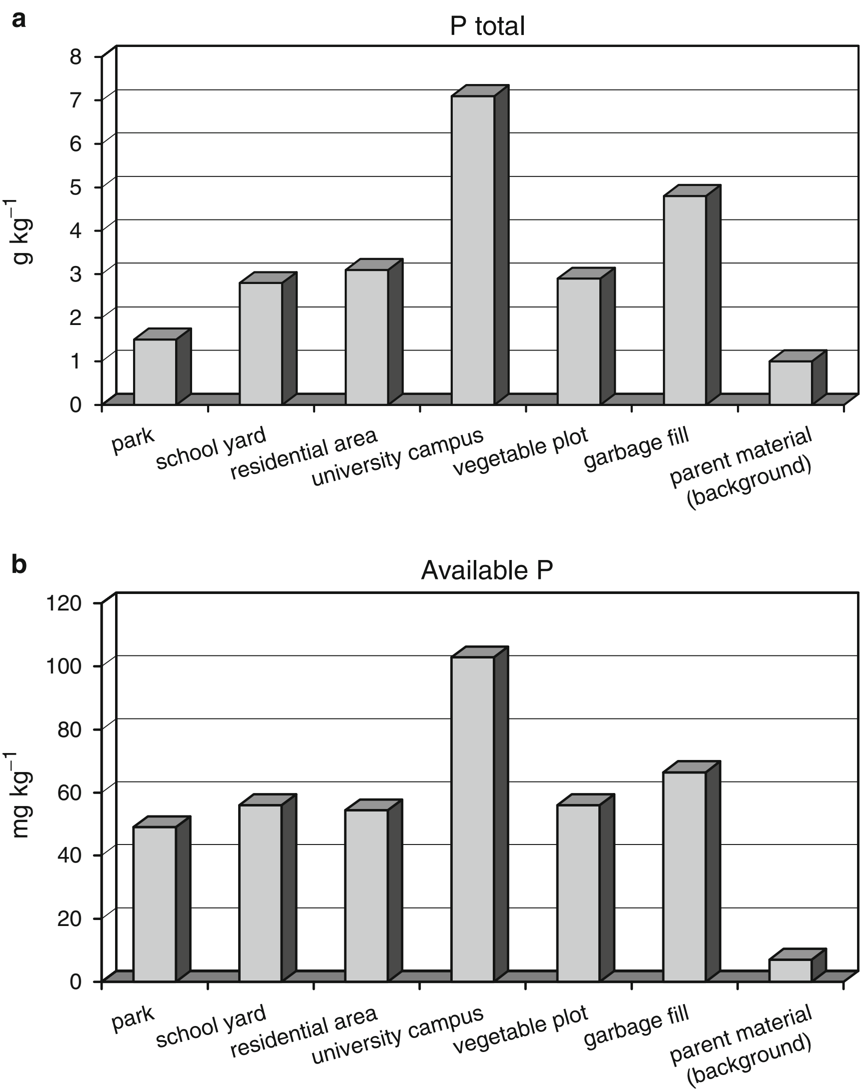



**Fig. 6.11** (**a**) Total phosphorus content in different urban soils (n = 15; depths: 0–140 cm) of Nanjing, China; extraction methods: $H_2SO_4/HClO_4$. (**b**) Available phosphorus content in different urban soils (n = 15; depths: 0–140 cm) of Nanjing, China; extraction methods: $NaHCO_3$ (Olsen) (Data from Zhang et al 2001)

in the oldest part of the city, nowadays used for residential and industrial purposes and almost completely consisting of anthropogenic deposits. It is followed by the new urban districts built-up after 1947, pedologically (deposited soils) comparable with the ancient part. In contrast, the continuously expanding suburban areas indicating both natural material and anthropogenic fills and used as parks, vegetable gardens and industrial sites had the lowest P concentrations of all. Consequently, a longer history of urbanization intensified phosphorus accumulation in soils. In particular, the total P content was more enriched in previously urbanized areas than in recently constructed areas (Yuan et al. 2007).

Another example in Zhengzhou, China, with 2,000,000 inhabitants, demonstrated the influence of urbanization on soil fertility as well. Rapid urbanization combined with transition of different functions of soil resources brought about clear problems with regard to soil quality and the environment. In the course of urbanization, a major transition of the productivity function occurred mainly from the grain field to orchard, garden and nursery. It reflected the influence of urbanization on the surrounding agricultural structure and, as a result, food cropping was converted to more economically profitable forms of agricultural land use. Consequently, a general increase in the average content of organic matter, nitrogen and potassium was observed in the suburbs from 1982 to 2003. The improvement of soil nutrients was relative to the large amount of chemical fertilizers applied every year. As for spatial variation, the distribution of nutrient content was inclined to be homogeneous, especially for available P and K, showing the impacts of agricultural planting structure (Kening et al. 2005).

In European cities the nutrient status of urban soils also reveals enhanced values. In the capital of Slovenia, Ljubljana, with 260,000 inhabitants, Seville (Spain) with 700,000 inhabitants and Torino (Italy) with 1,700,000 inhabitants Biasioli et al. (2007) sampled and analyzed topsoils up to 20 cm in depth belonging to open spaces (parks), river banks, ornamental gardens and roadsides. Most of the areas investigated have been designed by landscape planners and gardeners to establish vegetation and beautiful arrangements. Consequently, topsoil material has been used stemming from previous stockpiling during construction activities beforehand or from imported humic material. Accordingly, independent of missing continuous gardening procedures the nutrient capacity can be assessed to be optimal (Table 6.15). The mean values for total nitrogen exhibited 2.8–3.7 g $kg^{-1}$ in Ljubljana. The C/N ratio of 13 (Ljubljana), 10–11 (Seville) and 14 (Torino) would even be comparable with well-supplied agricultural areas. The pH values indicated a neutral reaction in harmony with the presence of calcium carbonate due to the alkaline additions such as mortar which we can find in urban areas in a lot of examples (see Section 4.4).

In the Ruhr area, Germany, the nutrient status was assessed in relation to the site use (see Section 2.4). Accordingly, it has been found that deposited soils with calcareous anthropogenic artefacts exhibited in the upper 10 cm a relatively low supply of phosphorus, potassium and magnesium (Table 6.16). Railway embankments, collieries and coal processing sites as well as ironworks and steelworks had been insufficiently supplied with the plant available macronutrients in question. Based upon the German standards the nutrient classes varied from very low to low. In contrast, deposited garden soils and soils of open space (parks) indicated much more nutrient capacity but in turn they showed more heterogeneity as well (Hiller and Meuser 1998).

The nutrient accumulation in garden soils observed in European and Asian urban soils is generally transferable to other regions of the world. In Ghana, for instance,

**Table 6.15** Mean values of some nutrient characteristics in three European cities (Data from Biasioli et al. 2008)

| | pH ($CaCl_2$) | | Organic carbon (g $kg^{-1}$) | | Total nitrogen | | C/N | |
|---|---|---|---|---|---|---|---|---|
| Depth (cm) | 0–10 | 10–20 | 0–10 | 10–20 | 0–10 | 10–20 | 0–10 | 10–20 |
| Ljubljana, Slovenia | 7.1 | 7.2 | 49.7 | 37.0 | 3.7 | 2.8 | 13 | 13 |
| Seville, Spain | 7.2 | 7.2 | 19.1 | 11.9 | 1.8 | 1.2 | 11 | 10 |
| Torino, Italy | 7.0 | 7.2 | 20.6 | 14.6 | 1.6 | 1.1 | 14 | 14 |

Sampling depth: 0–20 cm, land-use types: parks, ornamental gardens, roadsides, riverbanks

**Table 6.16** Nutrient status of anthropogenic soils with technogenic substrates (depth 0–10 cm) related to different uses in the Ruhr area, Germany (n = 22) (Data from Hiller and Meuser 1998)

| Uses | Total phosphorus | Plant available phosphorus | Plant available potassium | Plant available magnesium |
|---|---|---|---|---|
| Railway embankment | Low | Low | Low | Very low |
| Collieries and coal processing industry sites | Low | Very low to low | Very low to low | Very low |
| Iron and steelworks | Very low | Very low to low | Very low to low | Very low |
| Gardens, parks, recreational areas | Low to moderate | Low to moderate | Low to moderate | Very low to moderate |

Assessment:
Total phosphorus (aqua regia)
<265 very low, 265–530 low, >530–1,060 moderate, >1,060–1,550 high, >1,550 mg $kg^{-1}$ very high
Plant available phosphorus (calcium lactate acetate)
<22 very low, 22–57 low, >57–105 moderate, >105–157 high, >157 mg $kg^{-1}$ very high
Plant available potassium (calcium lactate acetate)
<33 very low, 33–75 low, >75–125 moderate, >125–183 high, >183 mg $kg^{-1}$ very high
Plant available magnesium ($CaCl_2$)
<25 very low, 25–35 low, >35–55 moderate, >55–85 high, >85 mg $kg^{-1}$ very high

a 200 km long rural-urban gradient was investigated to determine the influence of increasing urbanization on soil and geographical parameters (Drechsel and Zimmermann 2005). Three blocks at different distances to the city of Kumasi (Ghana) with a population of 1,900,000 were taken into account (Table 6.17). The block associated with the urban and peri-urban city structure (block no. 1) showed the highest percentage of vegetables grown and subsequently soils of this area were the most intensively used soils. The nutrient-rich sites were due to mulching with weed and crop as well as fertilizer application. Furthermore, cattle manure was applied, while compost use was not very common. The topsoils are well supplied with humus (1.7–3.8% Total Carbon, C/N ratio 11–14, effective Cation Exchange Capacity ($CEC_{eff}$) 5.6–11.8 $cmol_c$ $kg^{-1}$). Before cultivation, farmers burnt the area

**Table 6.17** Characteristics of the gradient urban – rural land-use in Kumasi catchment, Ghana (Data from Drechsel and Zimmermann 2005)

| | Block 1 | Block 2 | Block 3 |
|---|---|---|---|
| Inhabitants km$^{-2}$ | 147 (city of Kumasi: 650) | 84 | 44 |
| Proximity to major urban market (km) | 10–35 | 85–120 | 130–185 |
| Area covered with closed forest (%) | 2 | 18 | 25 |
| Percentage of fields used for vegetables (%) | 91 | 77 | 53 |

using the fertilizer effect of burning procedures in the line with the shifting cultivation traditionally used in African countries. Lettuce farming allowed 9–11 harvests per year there. Due to the lack of transportation the vegetables were grown close to the city market of Kumasi. The urban and peri-urban area can therefore be called an intensive vegetable production system, leading to a high nutrient level of the soils.

Table 5.8 (in Section 5.4.1) summarizes the chemical properties of some typical urban soils in Moscow, Russia. Apart from the high nutrient capacity some properties are also of importance. The pH value (measured in water) revealed a neutral to slightly alkaline reaction, indicating that the soil composition contained potentially alkaline artefacts like lime-based concrete (see Section 4.2.2). Accordingly, the base saturation was nearly 100% due to the dominant calcium cation. The base saturation did not decrease with increasing soil depth, since the exchangeable calcium related to the amount of calcium-containing inclusions present within the whole soil profiles. For comparison purposes, the base saturation in nature-like profiles of urban parks in Moscow varied between 60% and 85%. Additionally, in soils of lawns alongside roads in Moscow the exchangeable sodium content reached 5–10% due to the frequent use of de-icing salt on roads and pavements. It has been found that the percentage of sodium tended to decrease in late spring, summer and fall (see Section 3.3.1).

The carbon content of the profiles presented showed a clear tendency towards decreasing values within the profiles. However, in other profiles of the same city it was possible to detect deep humus-accumulated profiles. As profile no. 3 illustrated, a carbon content of 1.8% at a depth of more than 1 m below the surface can obviously be found (Stroganova et al. 1998).

If topsoils are influenced by dust deposition (soot) or contain substrate particles such as coal, asphalt and ashes, the carbon content will be altered considerably. This is noticeable from a wide C/N ratio (see Section 4.2.2). As illustrated in Table 6.18 dealing with soils in Halle with 230,000 inhabitants and Essen with 580,000 inhabitants (Germany), topsoil carbon, nitrogen and the C/N ratio influenced by the factors mentioned differ from neighbouring sites, where the influence did not occur.

Although properties like phosphorus, nitrogen, humus, etc. are advantageously listed as nutrient parameters, detrimental impacts have to be taken into account. Hence, nutrient parameters can take on a pollutant character, if concentrations are too high. Zhang et al. 2001 mentioned the high, in some cases extremely high

**Table 6.18** C and N content of natural topsoils depending upon atmospheric influence (coal dust deposition) in Halle and Essen, Germany (Unpublished data)

| Horizon | Depth (cm) | C total (%) | N total (%) | C/N |
|---|---|---|---|---|
| Chernozem, not anthropogenically influenced, Halle | | | | |
| A (ploughed) | 0–20 | 2.26 | 0.20 | 11 |
| Chernozem, influenced by immission of a lignite coal-fired power station, Halle | | | | |
| A (ploughed) | 0–25 | 13.86 | 0.36 | 39 |
| Podzol, not anthropogenically influenced, Essen | | | | |
| A | 0–12 | 1.45 | 0.04 | 36 |
| Podzol, influenced by immission of a hard coal-fired power station, Essen | | | | |
| A | 0–8 | 12.20 | 0.19 | 63 |

water-soluble P content of the vegetable gardens in Nanjing, China, which exceeded eutrophication level. The measured values ranged between 1.0 and 5.5 mg kg$^{-1}$ water-extractable P, while the result from natural Luvisols in the catchments was only 0.5 mg kg$^{-1}$. They concluded that percolating water and runoff from the garden areas will change groundwater and surface water quality negatively.

Analyses of total phosphorus and water-extractable phosphorus concentration in different German soils deposited confirmed the low percentage of mobile and available phosphorus in soils made of technogenic materials. In soils consisting of construction debris, household waste and lignite coal mining waste it was usually not possible to detect water-extractable phosphorus, although the total concentrations revealed values of up to more than 1,000 mg kg$^{-1}$ with regard to construction rubble and deposits containing household waste. In the presence of ashes and coal mining waste the total concentrations were much lower, indicating a maximum of 150 mg kg$^{-1}$. Exceptionally, soils derived from sewage sludge indicated very high total concentrations. The mobile percentage, however, showed the same tendencies as observed in the other technogenic substrates.

The German project dealing with deposited soils in different cities also took several anionic nutrients into consideration. The total boron concentration did not usually exceed concentrations of 10 mg kg$^{-1}$. However, the available borate percentage reached one third to a half. There was a tendency for higher values to be detected with reference to sewage sludge and dredged harbour sediment deposits. If the soil contained bottom and fly ash, the boron concentration increased drastically up to values of more than 70 mg kg$^{-1}$ combined with high available borate content.

Results relating to the element molybdenon were also found. The total molybdenon values normally lay below 1.0 mg kg$^{-1}$. Apart from sewage sludge with values higher than 1.5 mg kg$^{-1}$ clear tendencies with reference to the technogenic components were not discovered. The hot water extractable molybdate concentration fell below one third of the total concentration (Blume and Schleuss 1997).

Special attention should be paid to nitrate leaching. Systematic nutrient investigations in German cities revealed a relationship between the presence of technogenic substrates and nitrate formation. Based on incubation experiments the highest nitrate formation (>100 mg $NO_3$-N kg$^{-1}$) was determined in topsoil materials consisting of sewage sludge, dredged organic harbour sediments and household waste. Soil materials indicating mixtures of mineral soil and household waste and soils altered by compost revealed values between 50 and 100 mg $NO_3$-N $kg^{-1}$. Urban soils without organic ingredients achieved values between only 20 and 50 mg $NO_3$-N $kg^{-1}$. The nitrate formation occurred slowly and showed relatively low rates (<20 mg $NO_3$-N $kg^{-1}$) with reference to topsoil materials containing ashes and coal mining waste.

In urban environments, heavy nitrate leaching is, in most cases, not to be expected in contrast to the rural areas (see Section 4.3.2), where continuous nutrient application occurs. Some areas either intensively used as gardens or deposited with organic technogenic materials, however, might cause high nitrate leaching locally. For instance, long-term investigations of a soil consisting of sewage sludge deposits in Stuttgart, Germany, with a population of 600,000, revealed nitrate concentrations in the upper groundwater aquifer between 5 and 194 mg $L^{-1}$ and, additionally, relatively high concentrations of other mobile anions such as sulphate (120–361 mg $L^{-1}$) and chloride (22–56 mg $L^{-1}$), while the phosphorus leaching achieved very low concentrations (0.5–2.0 mg $L^{-1}$), as was to be expected (Blume and Schleuss 1997).

## 6.3 Pedogenesis

In general, the question must be asked, whether processes leading to the development of deposited soils should be considered to be pedogenic. Pedogenesis in urban landscapes is mostly confined to human intervention and the possible natural pedogenesis is drastically modified by man (Kosse 2000).

Nevertheless, pedogenesis of man-made soils can be recognized, because pedogenic features are visible. For instance, reductomorphic features in waste deposits can be discovered just behind the process of depositing and the visible features remind one of the development of natural soils in marshland, where metal sulphides as well as iron and manganese oxides pedogenically appear in the same manner. Apparently, the pedogenesis reveals parallelism in both kinds of soils, natural soils and anthropogenic soils. Parent material, climate, biota, topography and time are similar to natural soils factors of soil formation.

Theoretically, soil-forming processes start immediately when a bulldozer cuts and fills a landscape in order to level a construction site or park soil. Weathering from the top to the bottom and initial humus formation are the first pedogenic processes which take place. In the course of time soil formation continues independent of the quality of soil material deposited.

Three urban sites in France presenting different deposited soils were investigated to gain knowledge about pedological features. The first was an open landfill of paper industry by-products (5–10 years old), the second a former coking plant site reclaimed with treated industrial soil (1–2 years old) and the third was a field experimental set-up consisting of reconstituted soils made of organic products such as paper sludge and terric material such as treated industrial soil (1–2 years old).

**Table 6.19** Factors of pedogenesis in solution mining basins of different ages near Limoges, France (Data from Neel et al. 2000)

| | Zone 1 (inflow) | Zone 2 | Zone 3 |
|---|---|---|---|
| Vegetation | Green algae and bryophytes | *Cytisus scoparius*, few trees (*Betula pendula*, *Salix caprea*, *Populus tremula*) | *Equisetum telmateia*, few herbs (*Cirsium palustre, Angelica sylvestris, Phragmites australis*), trees (*Alnus glutinosa, Populus tremula*) |
| Depth of A horizon | 1–3 cm | 4–5 cm | 4–5 cm |
| Rooting depth | 3 cm | 15 cm | 100 cm |
| Organic matter content (A horizon) | 7% | 9% | 37% |
| Exchangeable bases (10–40 cm) | 6 $cmol_c\ kg^{-1}$ | 14 $cmol_c\ kg^{-1}$ | 43 $cmol_c\ kg^{-1}$ |
| Porosity (10–40 cm) | 10–15 vol% (loam) | 20–21 vol% (silt loam) | 22–25 vol% (silty clay loam) |
| pH (10–40 cm) | 2.9–3.6 | 4.9–8.2 | 6.8–8.0 |
| As (10–40 cm) (total concentration) | 5.3–6.0% | 0.4–5.3% | 0.2–0.7% |
| As mobility in tailing water | High (31.5 mg $L^{-1}$) | Extremely high (820 mg $L^{-1}$) | Extremely high (2,204 mg $L^{-1}$) |
| Pb (10–40 cm) (total concentration) | 1.1–2.1% | 0.1–1.7% | <0.1% |
| Pb mobility in tailing water | None | None | Moderate (1.4 mg $L^{-1}$) |

The results showed that even after a short period of time signs of early pedogenic evolution had been observed within the profiles, e.g. different horizons were visible and materials became structured. Moreover, spontaneous vegetation has colonised the three sites and there was some evidence of the activity of the macrofauna, the micro-organisms and the major biogeochemical cycles (water, carbon and major elements) seemed to be active (Sere et al. 2005).

In the vicinity of Limoges (France) with a population of 150,000 the influence of pedogenesis in three basins of a former lagoon used for treated residues of a gold mine was detected. There were three zones of distinct age. The first one, which was continuously in movement, belonged to the inflow next to the waste discharge pipe. In zone 2 drying had already taken place over a long time, so that it was possible to establish a few trees and a dense shrub cover. The third zone was the oldest one – this was long-term sedimented – and with intensive tree vegetation. The results demonstrated the physico-chemically determined progress of pedogenesis (Table 6.19). At a depth of 20–40 cm parameters such as porosity, organic matter content and exchangeable cations increased in the course of time and horizonation appeared. The metal concentration decreased due to weathering processes and consequently higher metal mobility in the vertical direction (Neel et al. 2000).

### 6.3.1 Humus Formation and Pedoturbation

Deposited soils might start humus accumulation in the course of time, unlike toxic substances which only reach a low level. In contaminated areas plant growth can be limited because of phytotoxicity leading to bare unvegetated soil cover. Due to the lack of plant establishment initial formation of humic topsoils would be prevented. Nonetheless, lichens and mosses may establish themselves, followed by the first grasses and herbs which reduce the area of bare soil and accelerate humus and dust accumulation. Within the next few years more and more shrubs and young trees establish themselves, deepening the root zone and enabling a humic topsoil (Brady and Weil 2008).

Soil investigations of brownfield sites in the city of Berlin that remained untouched after the Second World War and where natural succession of the vegetation occurred showed that, in the first 12 years after site levelling at undisturbed sites, 0.4 kg $m^{-2}$ $year^{-1}$ organic matter within the upper 30 cm accumulated and in the following 12 years an additional 0.2 kg $m^{-2}$ $year^{-1}$. With reference to the parameter nitrogen the values were 12 g $m^{-2}$ $year^{-1}$ in the first 12 years and 6 g $m^{-2}$ $year^{-1}$ in the next 12 years. In the upper part of the profile (0–2 cm) after approximately 12 years organic mater and nitrogen were in equilibrium and the amount did not increase any more (Blume 1986).

If toxicity to plants and soil animals does not exist, the generation of humic topsoils takes place relatively fast, as observed on abandoned brownfields. However, in city areas, e.g. parks and green buffers, litter is raked and swept away, so that the organic matter cycle is strongly interrupted, reducing biological activity and worsening the living conditions of soil animals.

In general, animals such as earthworms, ants, termites, mice and moles burrow and mix the soil permanently, influencing soil pedogenesis. The activity of the species in question is one of the pedoturbation factors known for both natural and anthropogenic soils. Apart from biologically determined pedoturbation, it also happens in wet, loamy to clayey sludge fields resulting from the swelling-shrinkage cycle. Intensive turbation can indicate disadvantageous features, because mixing processes can be responsible for the vertical distribution of contaminants, e.g. from bottom layers to the top layers and the soil surface (see Section 3.2.2).

Missing organic matter accumulation and subsequently absence of a more or less thick humic topsoil slows pedogenic soil formation, because the microbial community and plant species are not able to grow. In particular, the problem may occur in an arid climate, if water is also limited.

In the first instance, in urban road areas the situation is of importance, because the soil conditions are extreme, minimizing pedogenic processes. Investigations in Hong Kong, China, revealed the unfavourable properties by additional opening of tree pits in largely narrow pavements, where it was not possible for humus formation and physical weathering to take place. The tree growth was strongly limited due to a number of soil restrictions. The soil underneath the former paving was usually heterogeneous, containing different technogenic materials like construction debris. Accordingly, the texture was stony and coarse, the structure poor and compacted, the solum generally shallow and lithological discontinuity dominated. The root development was restricted not only by the compacted soil structure but also by the presence of buried utilities (Jim and Ng 2000).

### *6.3.2 Physical Weathering*

The most important climatic variables impacting soil formation are precipitation and temperature. Rain water percolates through the soil, transporting soluble compounds from the top to the deeper layers. The percolating water stimulates physical weathering reactions such as the freeze-thaw cycle and the shrinkage-swelling cycle.

In coal mining waste heaps physical weathering processes like the freezing-thawing and drying-wetting cycles lead to relatively rapid disintegration of the material consisting of sandstone, siltstone, clayey shale and residual coal, because the material, particularly the shale, is not resistant to weathering. In the Ruhr area, Germany, distinct heaps have been distinguished indicating that after 4 years the physically weathering depth reached only 10–15 cm, whereas after 40 years the upper 30 cm were weathered and after approximately 70 years the depth of weathering amounted to 43 cm (Fig. 6.12). Below, unweathered coarse material was discovered (Wiggering 1987).

Lack of aggregation and steep man-made slopes encourage soil loss by erosion (Fig. 6.13). Hence, soils on steep terrain have rather shallow, poorly developed soils in the upper portion of the slopes, whereas material accumulation termed colluvium occurs downslope. In most cases the depressed area downslope shows enhanced soil development and humus enrichment. In some cases material saturation and restricted aeration is visible in the lower portion of the slopes, caused by accumulation of loamy and clayey material (Brady and Weil 2008).

In association with the limited root development in urban soils the presence of skeleton percentages usually derived from technogenic components may influence the root development. A study dealing with the Ruhr area, Germany, showed that deposited soils of industrialized sites let recognize shallow root development conditions in contrast with the natural soils of the same catchment (Table 6.20). It should be noted that the anthropogenic soils containing calcareous technogenic substrates such as construction rubble and slag tended to reveal soils with a shallow rooting depth due to the limited weathering zone below (Hiller and Meuser 1998).

The low frequency of the freeze-thaw cycle in urban climates and, in particular, the higher winter temperature in cities of the northern hemisphere reduces soil aggregation and weathering. In the southern hemisphere similar tendencies are recognizable because of the higher temperatures and, consequently, fewer wet-dry cycles. Moreover, the transport of soluble compounds is reduced, leading to accumulation in the upper portion of the soil profiles (Craul 1992).

Again, in the northern hemisphere the city climate shows warmer temperatures in comparison with the surrounding rural area. In Moscow the difference between

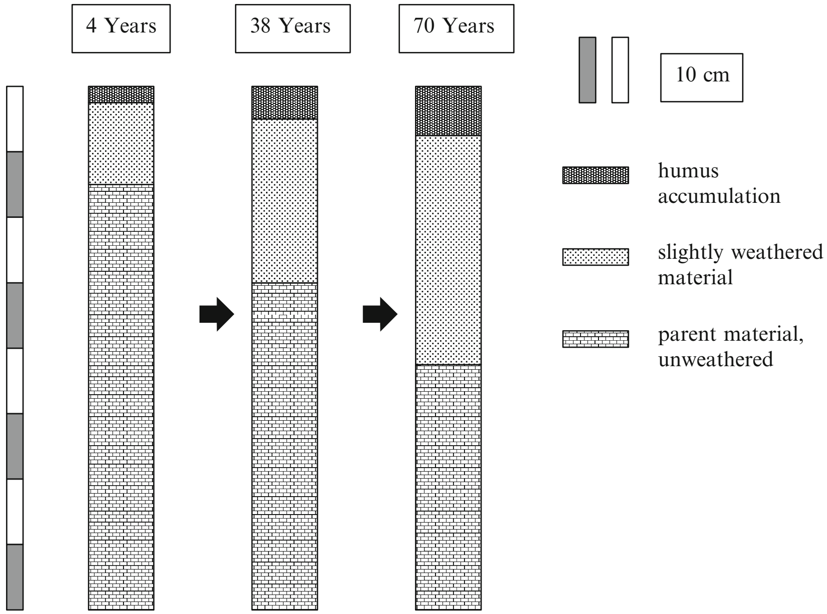



**Fig. 6.12** Weathering process of coal mining waste in the course of time

**Fig. 6.13** Erosion of a coal mining heap in Pernik, Bulgaria due to a lack of aggregation and humus formation

the city centre and the periphery of the city was 4–5°C during spring, summer and autumn. In wintertime the differences decreased to 2–4°C. However, in summertime in urban well-shaded parks the temperature was much lower than in the built-up areas, while in fall the differences tended to disappear. Moreover, precipitation and

**Table 6.20** Frequency (%) of rooting depth (cm) associated with natural soils (n = 19), carbonate-free deposited soils (n = 14) and deposited soils containing carbonate (n = 22) in the Ruhr area, Germany (Data from Hiller and Meuser 1998)

| Rooting depth | Very shallow (<10 cm) | Shallow (10–25 cm) | Moderate (>25–50 cm) | Deep (>50–100 cm) | Very deep (>100 cm) |
|---|---|---|---|---|---|
| Natural soils | 0 | 16 | 11 | 68 | 5 |
| Anthropogenic soils (carbonate content <2%) | 7 | 14 | 21 | 43 | 7 |
| Anthropogenic soils (carbonate content >2%) | 22 | 31 | 13 | 22 | 13 |

the amount of foggy days in winter showed increased values (5–10% and 20–30% respectively) to those of neighbouring rural areas (Stroganova et al. 1998). In other cities of the northern hemisphere the differences between city and rural areas were higher in relation to foggy days (+50–100%) and aerosol content (+250–1,000%). In contrast, in city climates wind speed, radiation and periods of sunshine showed lower values (10–30%, 15–20% and 5–15% respectively).

## 6.3.3 Chemical Weathering

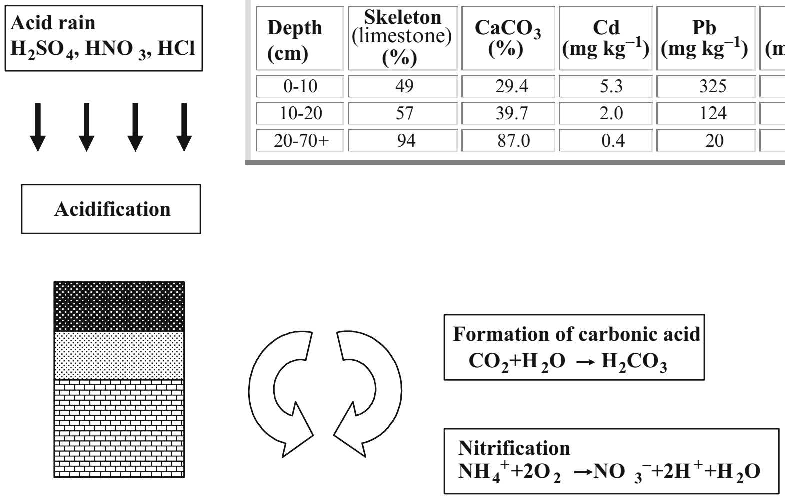
The nature of parent material frequently deposited by man influences soil characteristics. In turn, its chemical composition influences the chemical weathering processes. For example, calcareous materials like construction rubble reduce acidification, which normally occurs in soils of humid climates.

Dissolution is one important chemical weathering process observed in urban soils. Water capable of dissolving some minerals hydrates cations and anions until they are completely dissociated. In soils containing construction rubble and especially the component gypsum ($CaSO_4$) dissolution can be seen following the chemical reaction

$$CaSO_4 * 2H_2O + 2H_2O \rightarrow Ca^{2+} + SO_4{}^{2-} + 4H_2O \tag{6.1}$$

In the presence of acids chemical weathering is accelerated. Soils containing mineral and organic acids show a higher weathering rate, since acids react predominantly with soil minerals. In urban landscapes exhibiting acid precipitation due to $HNO_3$ and $H_2SO_4$ emission the uppermost horizons might be influenced. Unfortunately, acid rain is not the only source of acids existing in urban soils. The formation of carbonic acid, which is usually produced in topsoils, and nitrification, which is usually linked to humic topsoils as well, oxidation processes influence the development of acids.

In a 40-year-old deposit of calcium carbonate-containing ore mining waste the buffer capacity clearly decreased. Acidification led to reduced $CaCO_3$ content in the




| Depth (cm) | Skeleton (limestone) (%) | $CaCO_3$ (%) | Cd (mg kg$^{-1}$) | Pb (mg kg$^{-1}$) | Zn (mg kg$^{-1}$) |
|---|---|---|---|---|---|
| 0-10 | 49 | 29.4 | 5.3 | 325 | 1,820 |
| 10-20 | 57 | 39.7 | 2.0 | 124 | 1,070 |
| 20-70+ | 94 | 87.0 | 0.4 | 20 | 164 |

**Fig. 6.14** Influence on the contamination potential in connection with chemical weathering processes of an alkaline ore deposit in the district of Osnabrück, Germany

top horizon and the horizon below, accompanied by physical weathering recognizable by the lower skeleton content in the same horizons (Fig. 6.14). Consequently, soil mass decreased due to carbon dioxide volatilisation and the residual compositions remained. Accordingly, the first horizon revealed the highest Cd, Pb and Zn values, followed by the second horizon (see Section 5.4.4) (Meuser 2006).

In heaps consisting of coal mining waste the pyrite content amounts to up to 2%. In non-compacted heaps oxygen can enter, oxidizing pyrite according to the equation

$$FeS_2 + 7.5O_2 + H_2O \rightarrow Fe^{2+} + 2SO_4^{2-} + 2H^+ \tag{6.2}$$

In this way the formation of sulphuric acid ($H_2SO_4$) is caused and leachate of Acid Mine Drainage (AMD) occurs, contaminating groundwater and adjacent surface water. After depositing the pH value of coal mining heaps derived from the Carboniferous Period (predominantly hard coal) and the Tertiary Period (predominantly lignite coal) is about 7–8 but after a short period of time the pH value decreases, reaching values between 2 and 3. The rapid acidification is related to desorption of cations, including nutrients such as potassium, magnesium and calcium as well as heavy metals. The elements become mobile, reducing fertility of the heap soils and causing leachate of cationic metals (Wiggering 1987; Genske 2003).

The pyrite oxidation process occurring inside the heap is based on an exothermic reaction, emitting heat in the surrounding portion of the heaps. Because of the high amount of residual coal in association with deposited wooden struts used in seams spontaneous ignition can occur, leading occasionally to aboveground burning of the forested heap. Sometimes surface-near coal seams exposed to oxidation tend directly to cause fires. In China, in particular, so-called coal fires are widespread (Prakash 2007).

### 6.3.4 Aggregation

Most anthropogenic soils have undergone a short-term pedogenesis compared with natural soils. Accordingly, it was mostly not possible to form soil structure, reducing the types of soil structure present in urban soils. Because of the deposition character the majority of soils exhibit single-grained structural conditions. Massive structural conditions can be seen in subsoils as well, since the deposited material excavated somewhere else was not cultivated and refilled based on soil-protective preparation techniques. In the presence of loamy and clayey soil material platy structures are often created, caused by the heavy machinery used. Furthermore, a granular structure characterizes a lot of humic topsoils (A horizons). In parks and gardens the humic material was deposited after transportation from agricultural sites. Thus, the border between humic topsoil and underlying subsoil is very sharp and to a less extent influenced by pedogenic processes like pedoturbation. In summary, the amount of structural types is strongly reduced.

Most technogenic substrates contain a high percentage of calcium compounds. In dry conditions soluble $Ca(HCO_3)_2$ can lead to carbonation, in particular in the uppermost horizons. The chemical reaction

$$Ca(HCO_3)_2 \rightarrow CaCO_3 + CO_2 + H_2O \tag{6.3}$$

means calcium carbonate formation at the soil surface, resulting in a hard cemented crust. After cementation the calcium crust is impenetrable by roots.

Anthropogenic soils in urban environments can exhibit the opposite development as well. Soils alongside roads where de-icing salts containing sodium were applied show strong dispersion of aggregates. Soil aggregates are destroyed and accelerated runoff of the embankment material occurs.

### 6.3.5 Reductomorphose

In waste deposits reductive gases like methane and carbon dioxide displace the air. In an approximately 65 m high refuse heap in Berlin, Germany, 10–60% methane was discovered at a depth of 30–80 cm throughout the year. Below 80 cm the figure was 40–60%. Only close to the surface the $CH_4$ concentration was restricted to 0–10%. Microbial mineralization of organically bound sulphur was limited and reduction of iron and manganese oxides occurred. Consequently, black coloured metal sulphides developed, which can become oxidized by oxygen supply from the air in the upper parts of the deposit that was insufficiently covered. In dry conditions the oxidation of sulphides started from the top of the profiles and continued downwards.

The process of reduction is not limited only to waste deposits. Repeated infiltration of organic liquids like petrol and diesel may cause reduction of iron and manganese as well. Following this reductomorphic features become visible. Moreover, reductomorphic properties have been found in soils with leaking gas pipes and in wet deposits of wastewater sludge and dredged harbour sludge. These site-specific soils were termed reductosols (Blume 2000).

Reductomorphic features can be the result from soil human-made compaction as well. If soils are densely structured, indicating enhanced bulk density, water will not easily percolate, leading to stagnating water caused by reduced total pore space and crushed macropores. The creation of oxidation-reduction patterns becomes visible.

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# Chapter 7
# Assessment of Urban Soils

**Abstract** Contaminated urban soils can be assessed in different ways. This chapter introduces the most important approaches applied. Firstly, urban soils are classified based on the World Reference Base (WRB) and specified with reference to several international proposals. Secondly, it introduces the functional assessment of urban soils, taking particularly the functions habitat for plants, component of the water cycle as well as filter and buffer medium for metals into consideration. Solutions to quantify the functional estimation are presented. The assessment of contaminated urban soils is commonly focused on pathways endangering human health and other biota. Accordingly, all relevant pathways such as soil-human, soil-plant-human and soil-groundwater-human are described with reference to standardized exposure scenarios. Moreover, quality standards for contaminated land of a number of countries in different continents are listed, compared and discussed.

**Keywords** Contaminant pathway • Functional assessment • Quality standard • Risk assessment • Soil classification • Soil naturalness

## 7.1 Classification

With reference to the international classification system World Reference Base for Soil Resources anthropogenically influenced soils can be divided into Anthrosols and Technosols.

Anthrosols are anthropogenically altered soils through addition of organic material, household waste, irrigation and cultivation. The anthropogenic influence is mostly restricted to the upper part of the soil (topsoil). So, the pedogenically developed subsoil remains unchanged. Anthrosols are widespread and tend to be more located in the proximity of cities, towns and villages. Different types can be distinguished:

- Plaggic Anthrosols: soil with long-term deposition of plaggen sods derived from heathland, pasture and woodland, afterwards mixed with animal excrement in stables and brought back to the fields; predominantly existent in North Germany and The Netherlands (see Section 5.3.1).

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- Terric Anthrosols: soil with long-term amendment of fertilizers, compost, sand and sludge; existent all over the world.
- Irragic Anthrosols: long-term irrigated soil with slurry water as well as mineral and organic manures; predominantly existent in the Middle East and India.
- Anthraquic and Hydragic Anthrosol: soil on paddy fields with puddled layer underlayed by plough pan and redoxic subsoil due to long-term water ponding; predominantly existent in South and Southeast Asia, e.g. China, Vietnam, Indonesia.
- Hortic Anthrosol: soil with long-term fertilizing (mineral and organic manure, organic garbage), intensively cultivated; found especially in gardens and allotments in city peripheries all over the world (see Section 5.3.2).

Technosols are all kinds of man-made soils featuring deposited soils containing natural and technogenic artefacts, sealed soils and constructed soils with completely artificial layers. They exist in urban, industrial, traffic, mining and military areas in all parts of the world (WRB 2006). A subdivision of Technosol has not taken place up to now. However, some so-called qualifiers were defined, explaining Technosols in more detail. The most important are listed below:

- Ekranik: technogenic material existing within the soil profile at least 5 cm below surface and covering ≥95% of the soil surface in the horizontal direction
- Garbic: soil layer of ≥20 cm thickness within the upper 100 cm of the soil, consisting of ≥35 vol% organic garbage
- Reductic: reductomorphic features in more than 25 vol% of the upper 100 cm of the soil due to gaseous emissions such as methane and carbon dioxide
- Spolic: soil layer of ≥20 cm thickness within the upper 100 cm of the soil, consisting of ≥35 vol% industrial and mining waste
- Transportic: deposit of ≥30 cm material at the soil surface anthropogenically transported from outside of the area and altered
- Urbic: soil layer of ≥20 cm thickness within the upper 100 cm of the soil, consisting of ≥20 vol% artefacts with a percentage of ≥35 vol% construction debris

To cover all types of Anthrosols and Technosols the current system should be completed. Alternative suggestions published can be helpful for this purpose. Three interesting approaches have been made – in Germany (Burghardt 1994), Russia (Stroganowa et al. 1998) and in the Slovak Republic (Sobocka et al. 2000).

The German suggestion differentiates between several Urbic Anthrosols. The first one called *Meiktosol* indicates normally an anthropogenic alteration deeper than 40 cm and includes deep-ploughed and mixed soils without and with amendment of organic materials. Deep-ploughed soils (*Treposol*), deep-mixed soils (*Rigosol*), garden soils (*Hortiso*l) and cemetery soils (*Nekrosol*) are mentioned. The second group includes deposited soils (*Deposol*), divided into natural material (*Allolith*), technogenic material (*Technolith*) and mixtures of both materials (*Phyrolith*). Depending on the pedogenesis fresh deposits and deposited materials indicating (initial) humus formation are separated from each other. Furthermore, *Reductosol* reduced by methane, *Denusol* scalped to a depth of more than 40 cm and *Intrusol* influenced by groundwater, gases and liquids are defined.

The Russian system involves human-transformed and human-made soils. The first group covers physically transformed soils with anthropogenically altered urbic topsoils of more than 50 cm (*Urbanozem*). In the case of intensively cultivated Urbanozem in gardens and parks the term *Agrourbanozem* is used, cemetery soils are called *Nekrozem* and sealed soils *Ekranozem*. Furthermore, chemically transformed soils (*Chemozem*) are taken into consideration, divided into soils of heavily contaminated industrial and municipal areas (*Industrizem*) and soils impregnated and contaminated by liquid mineral oils (*Intruzem*). Completely human-made soils are called *Urbotechnozem*. The soil type *Replantozem* derived from rehabilitation procedures is distinguished from artificially constructed soils termed *Constructozem*.

The Slovakian suggestion is mainly focused on the topsoil characteristics. There is a differentiation between cultivated and ploughed soils with a thickness of 10–35 cm and soils ameliorated deeper than 35 cm. Apart from the cultivated soils the type *Anthrozem* is defined, which is man-made and derived from natural and technogenic material. Anthrozems with initial humus formation and rehabilitated topsoil are distinguished. Moreover, Anthrozems with contaminated topsoils are taken into account. If remnants of original diagnostic horizons are present, the soil type will be termed *Cultizem*. The complicated classification system offers a number of further varieties in line with, for instance, natural and technogenic substrata.

In summary, many details of the three approaches may overlap and can complete the World Reference Base classification system. In any case, the aspect of contamination appears to be considered in all systems introduced.

The following conclusion can be drawn in relation to Anthrosols in urban environments:

- To differentiate between regularly cultivated cropland soils and soils in the urban environment such as gardens and parks, deep-ploughed soils (deeper than 35, 40 or 50 cm) are generally separated (*Hortisol, Agrourbanozem, Anthrozem*)
- Soils of cemeteries indicating special features with regard to the kind of use are distinguished (*Nekrosol, Nekrozem*)
- Cultivated soils underlain by an impermeable basis (e.g. concrete), pipelines, etc. deserve a separation from other soil types (*Urbanozem*)
- Originally natural soils influenced by input of liquids are classified separately (*Intrusol, Intrusem*)
- Originally natural soils influenced by reductive gases like methane should be treated in the same way (*Reductosol*)
- Originally natural but heavily contaminated soils are differentiated as well (*Industrizem,* contaminated *Anthrozem*)

In relation to Technosols following aspects should be taken into consideration:

- Deposited soils (>50, >80 cm) are divided based on the kind of material deposited: natural, technogenic, mixtures of both (*Deposol, Allolith, Phyrolith, Technolith*) or on the target of depositing (*Replantozem, Constructozem*)

- Sealed soils are generally separated independent of the quality of the subsoil (*Ekranozem*)
- Deposited soils with recognizable diagnostic features of natural pedogenesis are treated distinctly (*Cultizem*)
- The degree of contamination and toxicity are further characteristics of differentiation (contaminated *Anthrozem*, contaminated toxic *Anthrozem*)

## 7.2 Functional Assessment

Based on the European Community soil functions are defined as follows:

- Production of food and biomass
- Storage, filter and transformation system
- Habitat and gene pool
- Physical and cultural environment of human beings
- Source of raw materials (European Community 2004)

In Germany for instance, the soil functions are described as

- Natural functions including habitat for human beings, animals, plants, soil organisms, component of ecological cycles related to water and nutrients and filter, buffer and biodegradation medium for contaminants
- Archival functions involving natural and human history
- And use functions with respect to agriculture and forestry, furthermore ground for human settlements and basis for raw material extraction (BBODSCHG 1998)

On the one hand ecological soil functions can be defined as aimed at the protection of soils, on the other hand soil functions are difficult to harmonise with soil protection, because they are preferentially focused on the use of soils. An example for the latter is the function of raw materials such as sand, clay, peat, limestone, sandstone, etc., mostly combined with a complete excavation and removal of the soil on-site. In contrast, soil functions like the storage, filtration and biological transformation of pollutants attenuate the contaminants and serve as a natural filter system for groundwater. They should be assessed as natural soil-protective functions. Accordingly, the habitat for humans, animals including edaphon and plants also fulfils an integral ecological function. The soil function production of agricultural and silvicultural goods belongs surely to the natural functions as well, because the soil supplies water and nutrients for all plant species, food plants and wild plants. On the other hand this function must also be assigned to the use functions with reference to the original aim of land use, agriculture as well as silviculture.

Just like every soil type, soils of urban, industrial, traffic and mine areas have several beneficial functions. They may provide plant products for food supply, taking garden, allotment and urban agriculture soils into consideration. As well, the soils can provide groundwater recharge for water, even drinking water supply, since water reserves are occasionally located in urban environments. The most important functions, however, deal with the contribution to infrastructure, aboveground (e.g. recreational site) and belowground (e.g. infrastructure pipelines, see Section 3.3.2). Moreover, the advantage of retention, decomposition and immobilization of toxic substances should be taken into account. Beneficial functions refer to the climatic effects such as buffering of temperature and humidity and dust entrapment as well. Furthermore, soils of urban environments mean prehistorical and historical archives (Lehmann and Stahr 2007).

The functions can be estimated by taking different soil properties into account. In this way it is relatively easy to describe complex functions. For instance, the habitat for plants can be defined, if single soil parameters such as available water capacity (estimated by texture, bulk density, humus content), nutrient status (e.g. estimated by cation exchange capacity) and pH value are investigated. The method is based upon a quantity of measurements in the past, where parameters have been directly determined using laboratory techniques.

Soil functions may partly depend on the same soil parameters leading to mutual dependencies between the single functions. The dependencies are one reason for the contrasting character of some functions. For instance, a nutrient-rich soil with nearly neutral pH value and high available water capacity might indicate a very positive assessment related to the soil function food production. The same soil might, however, require detrimental conditions for valuable vegetation and consequently be assessed badly as a habitat for wild plants (Lehmann et al. 2008).

In order to assess soil functions an evaluation consisting of five ranges can be recommended according to practical application. The ranges are termed very low (1), low (2), moderate (3), high (4) and very high (5).

To evaluate soil functions adequate methods have been developed, serving as a basis for city planning procedures. These methods should be efficient and useable for each area of concern. Accordingly, the applied methods should involve natural soils predominantly present on the city periphery and man-made soils mainly present in residential, industrial and traffic areas. Furthermore, the approaches should be managed at low cost. However, in urban areas with high soil heterogeneity it is necessary to implement field work. In the following the soil functions are introduced. Some important functions, namely the habitat for plants, component of the water cycle and filter and buffer function for heavy metals, are explained in detail, taking the contamination problem of urban areas into account.

For soil function evaluation purposes it makes sense to sub-divide functions into constituent functions. For example, the habitat function should be differentiated into habitat for humans, habitat for animals, and habitat for plants.

### *7.2.1 Habitat Function*

The soil function *habitat for human life* is related to the question of human health risk. If a soil does not reveal contamination or in other words, does not exceed quality standards defined (see Section 7.4.1), optimum life conditions can be expected and consequently the soil can be a basis for human life. Thus, the level of contaminants potentially endangering human health is the decisive soil parameter to be taken into consideration.

The soil function *habitat for flora and fauna* is linked to the life conditions of specialized and rare species and therefore closely connected to the improvement of biodiversity. In contrast to the floristic and faunal mapping on-site, the determination of the habitat function taking soil properties into consideration has a long-term basis. Based on Middle European locations for instance, the presence of soils with extreme properties such as very dry or wet conditions, acid or alkaline pH value, and very low cation exchange capacity plant species may survive adapted to these conditions, while most of the native species will not correspond to the soil quality present. In Table 7.1 a so-called ecogram is presented, forming a basis for functional soil assessment.

However, extreme soil properties can be found in industrialized areas as well. For example, coal mining heaps exhibit very acid pH values in combination with other abnormal characteristics (see Section 5.4.4). Another example refers to the salinity of a site. Rare halophytes can be discovered in maritime sites. If the salt content, detectable by the electrical conductivity, is high, halophytes react positively, while other species are not capable of undisturbed survival. But this approach is not only appreciable to characteristic of maritime sites, sites shaped by mining and industrial activities may occasionally indicate saline soil conditions as well. In a similar way to the salinity contaminants can determine plant species present because of different phytotoxicity the plant species indicate.

Thus, in urban environments apart from the soil properties, the naturalness of a specific location should be involved (Lehmann et al. 2008). A possible definition of the naturalness based on land use modifications is summarized in Table 7.2. To consider both life conditions of plants with reference to soil properties and naturalness of sites a combining matrix is developed in Table 7.3.

It should be noted that man-made soils really demonstrate a potentially valuable resource for the creation of species-rich native vegetation in towns and cities, partly compensating for the loss of vegetation in the countryside. Herbaceous native species can grow in artificial and poor soils. They have a high ornamental value, have a rapid growth, require low input of watering and fertilizing, preserve native plant populations and enhance wildlife habitats. They can provide an attractive roadside environment and offer physical advantages like prevention of soil erosion and stabilizing of constructed soils.

The *gene reserve of soils* is another meaningful soil function in discussion. For instance, bogs, isle-like present in some urban areas of the Northern hemisphere, contain countless seeds and may form an important gene reservoir. Excavation necessary for the subsequent construction of buildings may for ever destroy this gene pool.

### 7.2.2 Function as Component of Ecological Cycles

The soil function *component of the water cycle* has a quantitative as well as a qualitative aspect and is closely connected with the aim of groundwater protection.

**Table 7.1** Suggestion for basic evaluation of the soil function habitat for plants presented as ecogram

| pH ($CaCl_2$) value in 0–60 cm | Acid (<4.5) | Moderate (4.5–7.0) | Alkaline (>7.0) | Acid (<4.5) | Moderate (4.5–7.0) | Alkaline (>7.0) | Moderate (4.5–7.0) | Alkaline (>7.0) |
|---|---|---|---|---|---|---|---|---|
| CEC in 0–60 cm | Low (<4.0 $cmol_c$ $kg^{-1}$) | | | Moderate (4–12 $cmol_c$ $kg^{-1}$) | | | High (>12 $cmol_c$ $kg^{-1}$) | |
| Wet | 5 | 4 | 5 | 4 | 3 | 4 | 3 | 4 |
| Very moist | 4 | 3 | 4 | 4 | 3 | 4 | 2 | 3 |
| Moderately moist | 3 | 2 | 3 | 2 | 2 | 2 | 1 | 2 |
| Slightly moist | 2 | 1 | 2 | 1 | 1 | 1 | 1 | 1 |
| Slightly dry | 2 | 2 | 2 | 2 | 1 | 2 | 1 | 1 |
| Moderately dry | 4 | 3 | 4 | 3 | 3 | 4 | 2 | 3 |
| Very dry | 5 | 4 | 5 | 4 | 4 | 5 | – | – |
| Flooded and sub-hydric | 5 | 5 | 5 | 4 | 4 | 4 | 4 | 4 |
| Peat bog | 5 | 5 | – | 4 | 4 | – | – | – |
| Degraded peat bog | 3 | 3 | – | 2 | 2 | – | – | – |
| Salty (EC > 0.75 dS $m^{-1}$) | 5 | 5 | 5 | 5 | 5 | 5 | 5 | 5 |
| Phytotoxic concentration (e.g. Cu, Zn) | 5 | 5 | 5 | 5 | 5 | 5 | 5 | 5 |

Assessment ranges: 1 – very low; 2 – low; 3 – moderate; 4 – high; 5 – very high
CEC = cation exchange capacity

**Table 7.2** Suggestion for assessment of the naturalness of soils and sites

| Soil features | Land-use type (examples) | Range |
|---|---|---|
| Natural soil profiles | Extensively used woodland, biotopes (e.g. river bank, coastline, peat bog, wet meadow, xerophytic grassland), wilderness (e.g. savannah, desert land, rain forest, rocky mountains) | 5 |
| Natural soil profiles with alternative modification<br>• Ploughed topsoil<br>• Plaggic horizon <30 cm<br>• Colluvial topsoil<br>• Few artefacts<br>• Subsoil drainage<br>• Missing pesticide application (organic farming) | Extensively used cropland and horticulture, shifting cultivation, extensively used pasture and forest, near-natural park | 4 |
| Cultivated soil profiles: garden soil, cemetery soil, plaggen soil (plaggic horizon ≥30 cm), deep-ploughed soil (≥30 cm), vineyard soil<br>Deposited soil consisting mainly of natural material<br>Denuded soil <30 cm | Cropland (arable and vegetable farming, paddy field), orchard, vineyard, nursery, garden, allotment, cemetery, park (lawn, bed, woodland), golf course | 3 |
| Deposited soil profiles with alternative features<br>• Deposited horizon <60 cm<br>• Percentage of technogenic substrates <30%<br>• Percentage of sealed surface 10–60%<br>Denuded soil profiles until parent material | Suburban residential area (garden, green buffer, playground, sports field), pit of unconsolidated material, quarry | 2 |
| Deposited soil profiles with alternative features<br>• Deposited horizon ≥60 cm<br>• Percentage of technogenic substrates ≥30%<br>• Percentage of sealed surface >60%<br>• Soil compaction<br>• Reductomorphic features<br>• High contamination level | Industrial area (green buffer), central residential area (green buffer, courtyard), traffic areas (roadside green, railway line green), mining heap, landfill, sludge field | 1 |

Assessment ranges: 1 – very low; 2 – low; 3 – moderate; 4 – high; 5 – very high

**Table 7.3** Suggestion for the assessment of the soil function habitat for plants based upon ecogram results (see Table 7.1) and naturalness results (see Table 7.2)

| | Ecogram results | | | | |
|---|---|---|---|---|---|
| Naturalness results | 1 | 2 | 3 | 4 | 5 |
| 1 | 1 | 1 | 1 | 2 | 3 |
| 2 | 1 | 1 | 2 | 3 | 4 |
| 3 | 1 | 1 | 2 | 3 | 4 |
| 4 | 2 | 2 | 3 | 4 | 5 |
| 5 | 2 | 2 | 3 | 4 | 5 |

Assessment ranges: 1 – very low; 2 – low; 3 – moderate; 4 – high; 5 – very high

Soils act as a filter system associated with adsorption and desorption capability, mechanical filtration as well as microbial degradation of organic pollutants. There is no doubt that groundwater quality is dependent on the filter requirements. Moreover, if groundwater is used for drinking water purposes, the significance of the function component of the water cycle may become even more import.

In the context of prevention of run-off and erosion, landslide and flooding a quantitative aspect of the function cannot be neglected, in particular the soil infiltration capacity. In the urban environment the infiltration rate ought to be included if one remembers the frequent disastrous flood occurrences. The hydraulic conductivity of the soil profile, the water holding capacity within the profile, the groundwater table and stagnating water are of importance to enable water infiltration and, particularly, water flow downwards. Due to the expected high percentage of technogenic components the discontinuity and interruption of macropores as well as the intra-pore impact (see Section 6.1.4) should be involved. Apart from soil characteristics the infiltration rate depends on the slope gradient and the percentage of sealed surfaces. In the context of functional assessment a general decision must be made whether sealed surfaces should be implemented or not. This question is transferable to all kinds of soil functions. Because of the high percentage of sealed surfaces in urbanized areas it appears advantageous that the sealed sites generally take part in functional assessment. Furthermore, the vegetation responsible for interception losses and transpiration rates should be included in order to recognize the water really infiltrated.

In Fig. 7.1 a model expressing the soil function component of the water cycle is illustrated. The water retention is classified on the basis of the physical properties water-holding capacity and hydraulic water conductivity. The slope gradient can change the classification identified as follows:

- Slope gradient 2–9%: subtraction of one value
- Slope gradient >9–18%: subtraction of two values
- Slope gradient >18%: general categorization very low (1)

The vegetation influences the result as follows:

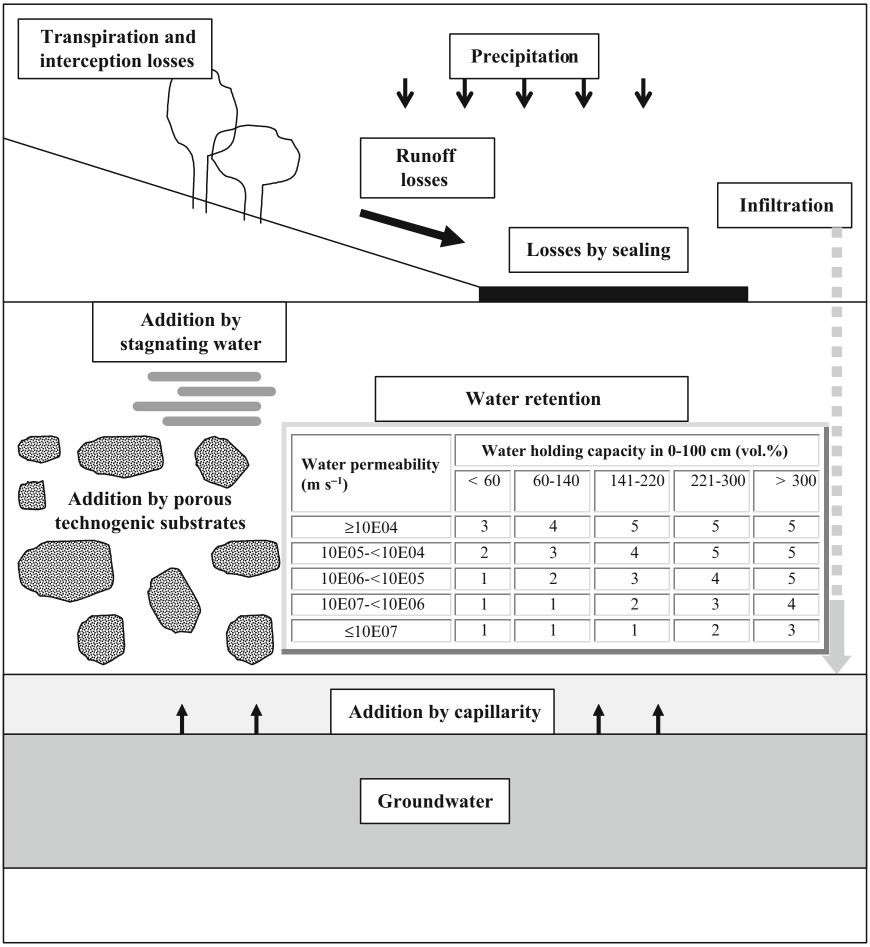
- Vegetation with high transpiration and interception rate (forest, tree planted park): subtraction of two values
- Cropland, pasture, garden, park: subtraction of one value

The influence of sealing results in:

- Porous sealing materials (e.g. water-bound cover, small slabs, perforated materials, etc.): subtraction of two values
- Unweathered materials (e.g. asphalt, concrete): general categorization very low (1)

Groundwater and stagnating water are taken into consideration in the following way:

- High average groundwater table (<40 cm): addition of two values
- Moderate average groundwater table (<80 cm): addition of one value
- Presence of stagnating water in 0–100 cm: addition of one value




| Water permeability ($m\ s^{-1}$) | Water holding capacity in 0-100 cm (vol.%) | | | | |
|---|---|---|---|---|---|
| | < 60 | 60-140 | 141-220 | 221-300 | > 300 |
| ≥10E04 | 3 | 4 | 5 | 5 | 5 |
| 10E05-<10E04 | 2 | 3 | 4 | 5 | 5 |
| 10E06-<10E05 | 1 | 2 | 3 | 4 | 5 |
| 10E07-<10E06 | 1 | 1 | 2 | 3 | 4 |
| ≤10E07 | 1 | 1 | 1 | 2 | 3 |

**Fig. 7.1** Illustration of the soil function component of the water cycle

Technogenic substrates can be involved as follows:

- Presence of porous substrates (brick, bottom ash, some slag types) with a percentage of more than 10%: addition of one value

Additionally, soils serve as a *factor in climate regulation* as well, namely the cooling effect. Due to the cooling down of air temperature by enhanced evaporation in summertime of the northern hemisphere the city climate becomes more beneficial. The function is not relevant in the temperate zone. The evaluation of the function is associated with climatic parameters like precipitation and air temperature and with soil properties like water holding capacity and capillarity in the case of sites influenced by groundwater. Heat storage by buildings, vegetation, wind speed and roughness of the soil surface are further points for consideration. Originally, however, this function is more closely connected with the environmental compartment atmosphere than with the pedosphere (Lehmann et al. 2008).

The soil function *component of the nutrient cycle* focuses on the natural fertility of the soil, which can be estimated by soil basic parameters like the cation exchange capacity. In relation to the natural fertility points of consideration are water supply, air capacity, nutrient and humus content and rooting depth. Generally speaking, the potential for root growth is the important factor that may summarize the functional requirements. Similarly to the evaluation of other functions simple soil parameters need to be considered. In contrast with the function habitat for plant species, the best quality soils enabling high biomass production are associated with the very fertile sites. The function is of importance in agriculturally used areas and less significant in woodland and ornamental gardens. The nutrient supply determines growth conditions of ornamental plants, lawns and other urban land uses as well. However, the productivity of soils in relation to food production might play a limited role in urban environments of the developed countries. In less developed countries the situation seems to be clear, because vegetable production in city areas helps people to improve their living conditions considerably, if not to survive at all (see Section 6.2.5).

### *7.2.3 Filter, Buffer and Transformation Function for Contaminants*

The soil function *mechanical filter and chemical buffer for heavy metals and organic contaminants* becomes more interesting in urban environments because of the higher contamination level. Furthermore, humans are permanently exposed to the contaminants irrespective of the pathway oral ingestion, entering the respiratory tract as well as drinking water and plant consumption (see Section 7.3.3). The filtering and buffering capacity enables soils preventing harmful substances from reaching the groundwater and the food chain.

With reference to the heavy metals, simple soil properties such as clay content and humus content may qualify the filter and buffer capacity, because they adsorb heavy metals. The pH value and binding agents like $CaCO_3$ determine metal solubility and air capacity influences mechanical filter capacity of particulated pollutants (see Sections 6.1 and 6.2). In Table 7.4 an approach is presented how heavy metal mobility can be estimated aimed at the possibility of minimizing entry into the food chain and minimizing entry of leachate into the groundwater. With reference to the adsorption potential of heavy metals soil characteristics frequently found in urban areas should be included:

- Due to the strong binding capacity black coloured, sulphide-rich reductive horizons (e.g. sludge fields, landfills) and alkaline horizons with calcerous technogenic artefacts such as concrete, mortar, slag and ash (e.g. deposited soils of residential and industrial area) reveal high binding potential in accordance with low water solubility (range 1 = very low).

**Table 7.4** Suggestion for the assessment of heavy metal adsorption potential in soil horizons; the sum of (a–d) means the final adsorption potential

| | | | | | |
|---|---|---|---|---|---|
| (a) Dependence on pH ($CaCl_2$) value | | | | | |
| pH value | <3 | 3–<4 | 4–<5 | 5–<6 | ≥6 |
| Cd, Ni, Zn | 0 | 1 | 2 | 3.5 | 5 |
| Cu, Cr, Hg, Pb | 1 | 2.5 | 4 | 5 | 5 |
| (b) Consideration of humus content | | | | | |
| Humus content (%) | <2 | 2–8 | >8–15 | ≥2 and pH ≥ 6 | |
| Cd, Ni, Zn | 0 | +0.5 | +1 | −2/0 (Zn) | |
| Cu, Cr, Hg, Pb | 0 | +1 | +1.5 | −2/−1 (Cu) | |
| (c) Consideration of texture | | | | | |
| Texture class | Gravel, sand | Loamy sand, sandy loam, loam, sandy clay loam, silt, silt loam | Silty clay, silty clay loam, clay loam, clay | Silty clay, silty clay loam, clay loam, clay and peat with swelling-shrinkage cycle | |
| Cd, Ni, Zn | 0 | +0.5 | +0.5 | −1 | |
| Cu, Cr, Hg, Pb | 0 | +0.5 | +1 | −1 | |
| (d) Consideration of Fe and Mn oxides | | | | | |
| | | Present | | Absent (reductive horizons without sulphides) | |
| Cd, Ni, Zn | | +0.5 | | −0.5 | |
| Cu, Cr, Hg, Pb | | +1 | | −0.5 | |

Assessment ranges: 1 – very low; 2 – low; 3 – moderate; 4 – high; 5 – very high

- In contrast, reductive horizons lacking in sulphides (sludge fields, deposited and compacted soils with stagnating water) tend to reduce iron and manganese oxides, so that adsorptive cations become mobile and occluded metals are released.
- Peaty soil horizons and clayey soil horizons with a swelling-shrinkage cycle exhibit preferential flow during dry periods, causing enhanced heavy metal leachate.
- While a high humus content reduces the water solubility of metals, soil horizons that are humus-rich and alkaline simultaneously tend to elevate metal leachate caused by the formation of mobile organo-metallic complexes.
- Soil horizons with a very high total concentration (for instance Cd and Hg > 10 mg $kg^{-1}$, Ni > 500 mg $kg^{-1}$, Cr, Cu, Pb and Zn > 1,000 mg $kg^{-1}$) are assumed to release a portion of the adsorbed metals, since the maximum adsorption potential can be exceeded (see Section 6.2.1).

The mechanical filtration and chemical buffer of contaminated particulate matter is shown in Table 7.5. The transport of particulate matter through soil profiles is dependent on air capacity (macropores) and potential adsorption potential defined by Cation Exchange Capacity (CEC):

**Table 7.5** Assessment of heavy metal mechanical and chemical filtration as particulate matter in soil horizons

| | Cation exchange capacity ($cmol_c$ $kg^{-1}$) | | | |
|---|---|---|---|---|
| Air capacity (vol%) | <4 | 4 – < 12 | 12 – < 30 | ≥30 |
| <2 | 3 | 4 | 5 | 5 |
| 2 – < 5 | 2 | 3 | 4 | 5 |
| 5 – < 10 | 2 | 3 | 4 | 4 |
| 10 – < 25 | 1 | 2 | 3 | 4 |
| ≥25 | 1 | 2 | 2 | 3 |

Assessment ranges: 1 – very low; 2 – low; 3 – moderate; 4 – high; 5 – very high

- The more macropores, the faster the transport downward occurs
- The lower the CEC, the higher the possibility of percolating downwards

The soil function *transformation medium for organic pollutants* depends on the living conditions for microorganisms capable of successful biodegradation. In detail, parameters like water content, nutrient supply and temperature determine the turnover of organic pollutants. In general, humus-enriched topsoils offer the best biodegradation rates. Accordingly, the evaluation of the function is mainly focused on the soil quality of humic topsoils, though the contamination can be detected within the whole soil profile. It is difficult to assess the parameters with reference to the living conditions of the edaphon. Since direct measurement of microorganisms during functional evaluation is not practice-oriented and cost-effective, basic soil parameters will often be applied for indirect estimation purposes. As a result of functional assessment of soils in a number of cities in Austria, Germany, Italy and Switzerland it was recommended that parameters such as humus content, structure (especially presence of granular structure) and pH value in the upper 30 cm, as well as groundwater level below surface, were included (Lehmann et al. 2008).

### *7.2.4 Archival Functions*

The *archival soil function* considers predominantly indications of historic land use and former soil formations in line with witnesses of historical development. This function helps us to understand the evolution and development of the earth and the mankind. A distinction is made between functions representing an archive of landscape history (e.g. plaggen soils) and those containing information about cultural history (paleontological and archaeological findings).

In some urban areas natural undisturbed soils are generally worth protecting, because they are rare and contain valuable information in the last isle-like sites at all. In order to evaluate the archival function the rarity of the total area under investigation should be taken into account. The archive of cultural history must be related to artefact findings proving the historical significance (Lehmann et al. 2008).

### *7.2.5 Use Functions*

The *use functions* are strongly linked to socio-economic factors. Soils provide ground for the erection of buildings, roads, infrastructure and recreational facilities and they can serve as sites for household, commercial and mining waste disposal. In urban areas there are numerous constructed land-use types competing with each other for the occupation of limited municipal space (see Table 2.5):

- Residential area
- Industrial and commercial area
- Special building area
- Disposal area
- Traffic area
- Recreational area

The burden of constructed areas falls heaviest on urban agricultural land and former abandoned, fallow land and, except for cities in less developed countries, on urban forest and nature reserves to a lesser extent. Water bodies are, for the most part, less included in urban expansion as well.

Finally, soils can be sources of numerous raw materials extracted in urban areas leading to the establishment of extraction and mining territories:

- Unconsolidated material pits (e.g. sand, loam, clay, marble)
- Quarries for consolidated material (e.g. sandstone, limestone, basalt, granite)
- Peat dredging areas
- Sludge dredging areas
- Open-cast mines (e.g. lignite coal, ores)

In general, the use functions can mean excavation of at least humic topsoil, complete removal of the soil, radical disturbance of soil properties as well as deposit and sealing onto the original soil surface. Consequently, establishment of use functions is, in principle, contrary to soil protective measures.

## 7.3 Pathway-Oriented Soil Assessment

### *7.3.1 Risk Assessment Scheme*

As previously explained in Sections 3–5, in the urban environment it can be expected that, in principle, enhanced contaminant values are caused by different sources. Consequently, risk assessment might play an important role, because population density is high, land uses are multifunctional and contaminant problems have to be solved to enable an urban lifestyle without any danger to human health.

In potentially contaminated city areas many questions have to be answered in order to assess contaminated land. Typical ones are:

- Do contaminant sources exist?
- What kind of toxic substances are involved?
- What is the current distribution of contaminants and are there extractable hot spots?
- Do the analytical results exceed benchmark levels and quality standards, if defined at all?
- What are the background values in comparison with analytical results?
- What are the site characteristics which influence pollutant migration in relation to soil properties, hydrogeology, land use pattern, etc.?
- What are the potential and dominant migration pathways and transport processes contributing to spread of contaminants?
- Is there soil erosion and deflation which contaminate adjacent areas?
- Are there current and future potential receptors that could be affected, especially sensitive uses and ecosystems?
- Is there an immediate life-threatening exposure present?

Systematically, the different questions should be part of the risk assessment scheme. Risk assessment takes place in order to ensure that the area of concern is fit for current use or for the intended planned use. The likely presence of pollutants and their linkages has to be involved.

The risk assessment is focused on the relationship between contaminant and receptor by a pathway. Thus, all three parts, contaminant, receptor and pathway must be present to realize a reasonable assessment. Consequently, in the context of contaminated land the three components must be taken into account, the source of contamination, the receptors and the pathways for receptors which are exposed to toxic compounds. The pathway is closely connected to the migration of the contaminants. Migration means the movement of contaminants from their source via media such as air, water and soil to the receptors. The term exposure describes the receiving of a chemical dose or the coming into contact with toxic substances by an organism. Finally, the receptor is mostly the human being, but also other organisms like fish, mammals and birds can also be called receptors (Asante-Duah 1996; Nathanail and Bardos 2004).

The receptors are the main point of view in the exposure scenario. For this reason, distinct receptor exposures should be taken into consideration. In general, the scenarios are mainly focused on human health involving direct oral ingestion, inhalation and dermal contact with contaminated soil, use of contaminated groundwater and surface water as part of the municipal supply and food and dairy products consumption. In Fig. 7.2 all relevant exposure scenarios are summarized.

The risk assessment procedure is managed by the Local Authority and aims to solve different problems. Firstly, the question whether significant harm reaches the receptor must be answered. Furthermore, the way in which one or more pollutants interact with each other, e.g. synergistic potentials, has to be examined and ultimately the possible combination of two or more pathways has to be taken into consideration. The risk assessment has to identify every pollutant that can be sufficiently toxic to the receptors. Therefore, the detection of the potential of a

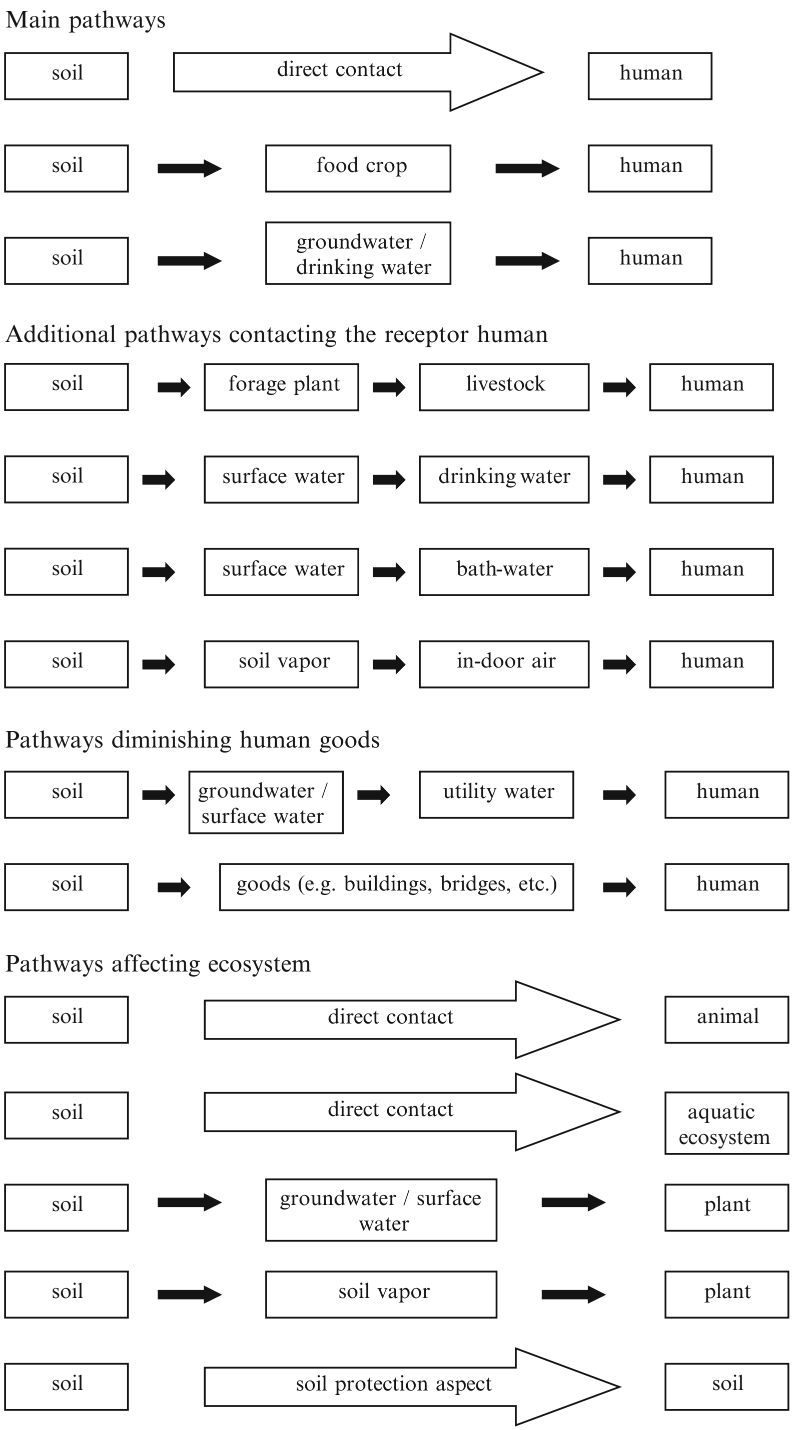



**Fig. 7.2** Overview of the relevant pathways in question

contaminant to disperse in soil and groundwater must be combined with the danger to the receptors of coming into contact with toxic substances.

In urban environments the risk assessment has to be related to the current use of the land guided by existing planning permissions. The town planning processes, however, usually lead to altered land use opportunities. For instance, if an ore mine spoil heap of an abandoned site is used for mountain bike terrain, the sensitivity of the area has changed significantly and it can be assessed as a locality like a playground, where children are now exposed to the toxic substances. In the case of a derelict industrial site that is used for residents to go for walks on, a new receptor situation has to be taken into account, e.g. danger to humans (pathway soil-human) and even dogs taking part in walking activities (pathway soil-animal). These examples may give advice to the problems present in areas for which there are no detailed planning processes, as is the case with brownfields in city areas. If a planning process is implemented, the area in question should be assessed with reference to the planned use and its possible alterations of the receptors involved.

In the absence of planning intentions the current land use is of importance. For this reason, a lot of highly contaminated city areas remain untouched, since the necessary requirements for risk assessment and following clean-up strategies are low as long as the brownfield status is present. Accordingly, many large abandoned polluted areas have never been assessed for decades. A risk assessment and possible soil clean-up without any decision on the planning does normally not occur, or, in other words, the target of multi-functionality for contaminated land is no real option to choose. However, risk assessment can be applied to enable redevelopment of formerly used, contaminated areas. It can be an economic instrument improving the land value in association with changes of ownership (Nathanail and Bardos 2004).

The risk assessment involving the target of risk reduction is called risk management. The management includes the decision how the risk can be removed effectively and in a cost-efficient way. Again, this decision depends on the land-use type present or planned. If a change to more sensitive land uses occurs, the process of risk assessment has to be repeated or at least to be completed. In the context of integrated planning processes, the risk management will be a decision-making process aimed at fitness for land use, protection of the environment as well as long-term care. All aspects of development and sustainability have to be implemented.

The management refers to decisions about remediation, land use restrictions, monitoring, etc. Attempts are made to answer questions about contaminant source reduction (e.g. soil excavation), pathway management (e.g. installation of barrier systems) and modification of the exposure of the receptors (e.g. impacts of planned future land use). The source reduction is easy to carry out in the case of localized zones, so-called point sources. Diffuse sources, where the contaminants were spread over wide area, however, are more difficult to remediate. The modified planned land use is of importance, since some exposure scenarios may disappear in modified conditions. For example, the pathway soil-plant-human would not continue, if cropland for growing food was changed into a car park area. Simultaneously, the same area must not involve the pathway soil-human (direct contact), because direct contact to the soil matrix is completely excluded.

In urban environments, however, one negative aspect appears, if the site owner requires a decision urgently, because the land has to be sold rapidly or the planned business should be started at once. Especially in relation to severe site problems, e.g. unknown pathways, insufficient data basis, chemically complicated soil contamination, etc. urgent decisions can become problematical (Nathanail and Bardos 2004).

For soil scientists the risk assessment tool should include all relevant receptors, calculated intake of contaminants and pathways. This tool should not be viewed as a black box with a number of unknown impacts. Therefore, the assessment has to consider contaminant properties like toxicity, physico-chemical properties and site concentration. Moreover, site conditions such as pH value, organic matter content, depth of contamination, current and planned land use and receptor characteristics such as soil ingestion rate and drinking water usage should be integrated in order to assess the pathways to the receptors in more detail.

### *7.3.2 Calculation of the Risk Assessment*

The Acceptable Daily Intake (ADI) estimates the maximum dose of a chemical to which a receptor can be exposed on a daily basis over a lifetime without any suffering adverse effects. It is calculated by mg $kg^{-1}$ (body weight) $day^{-1}$. It depends on the absorption dose entering the organism via the gastrointestinal tract (oral ingestion), via the lungs (inhalation) and via the skin (dermal exposure). Normally, the intake of non-carcinogens and cancerogenes is separated. In the case of cancerogene pollutants the health hazard evaluation is based not only on the formula mg $kg^{-1}$ (body weight) $day^{-1}$, a cancer slope factor (cancer potency factor) is also involved. This factor is defined as an upper bound estimate of the probability of a response to develop cancer per unit intake of a chemical over lifetime. Accordingly, it is a statistical value only and does not necessarily mean that cancer will really develop.

The ADI value must be differentiated from the term acute toxicity. The latter shows acute symptoms of poisoning derived from a chemical intake in a short period of time and it requires emergency response actions.

To calculate the exposure scenarios several factors should be taken into consideration. In any case the period over which exposure takes place, the body weight of the receptor (kg) and the contaminant concentration of the investigated media (mg $kg^{-1}$, mg $L^{-1}$, mg $m^{-3}$) are important factors. The exposure period can be calculated on different bases such as exposure time (h $day^{-1}$), frequency (day $year^{-1}$) or duration (e.g. year). It should be noted that the exposure period, for instance, can be reduced during snow cover of the soil and rainy days. Furthermore, in risk assessment attention should be paid to the following parameters:

1. With reference to the oral ingestion
   (a) The ingestion rate (mg $day^{-1}$)
   (b) Gastrointestinal absorption factor (%)
2. With reference to inhalation
   (a) Inhalation rate ($m^3$ $h^{-1}$)
   (b) Retention rate of the inhaled air (%)

3. With reference to dermal exposure
   (a) Skin surface area available for contact ($cm^2$ $event^{-1}$)
   (b) Soil loading on skin (mg $cm^{-2}$)
   (c) Skin absorption factor (%)
4. With reference to drinking water consumption
   (a) Average water ingestion rate (L $day^{-1}$)
   (b) Gastrointestinal absorption factor (%)
5. With reference to contact to surface water bodies
   (a) Contact rate (L $h^{-1}$)
   (b) Gastrointestinal absorption factor (%)
6. With reference to food consumption, in particular locally grown crops
   (a) Average food ingestion rate (mg $meal^{-1}$)
   (b) Gastrointestinal absorption factor (%)

In Table 7.6 specific parameters for exposure scenarios based upon the Environmental Protection Agency (EPA) of the United States are listed. Table 7.7 presents some parameters based on the exposure models of The Netherlands, for comparison (Brand et al. 2007). In conclusion, the definition of the parameters estimating human health risk reveals some differences, so that the ultimate politically determined quality standards might vary from country to country.

The risk assessment scenarios must be standardized and perhaps simplified, but they have to consider many factors to present reality-like conditions. Hence, the

**Table 7.6** Specific parameters used for exposure scenarios in the United States of America (Data from Asante-Duah 1996)

| Parameter | Unit | Children up to 6 years | Children 6–12 years | Adults |
|---|---|---|---|---|
| Average body weight | kg | 16 | 29 | 70 |
| Average lifetime | year | 70 | 70 | 70 |
| Average lifetime exposure | year | 5 | 6 | 58 |
| Amount of soil ingested | mg $day^{-1}$ | 200 | 100 | 50 |
| Frequency of soil contact | days $year^{-1}$ | 330 (residents, schools) | 330 (residents, schools) | 330/260 (residents/off-site work) |
| Average total skin surface | $cm^2$ | 6,980 | 10,470 | 18,150 |
| Percentage of skin area contacted by soil | % | 20 (=1,396) | 20 (=1,047) | 10 (=1,825) |
| Inhalation rate | $m^3$ $h^{-1}$ | 0.25 | 0.46 | 0.83 |
| Retention rate of inhaled air | % | 100 | 100 | 100 |
| Frequency of fugitive dust inhalation | h $day^{-1}$ (outside) | 12 (residents, schools) | 12 (residents, school) | 12/8 (residents/off-site work) |

**Table 7.7** Specific parameters used for exposure scenarios in The Netherlands (Data from Brand et al. 2007)

| Parameter | Unit | Children | Adults |
|---|---|---|---|
| Average body weight | kg | 15 | 70 |
| Average lifetime | year | 70 | 70 |
| Amount of soil ingested | mg day$^{-1}$ | 100 | 50 |
| Relative absorption factor | % | 100 | 100 |
| Skin area contacted by soil | cm$^2$ | 500/2,800 (indoor/outdoor) | 900/1,700 (indoor/outdoor) |
| Inhalation rate | m$^3$ h$^{-1}$ | 0.32 | 0.83 |
| Retention rate of inhaled dust | % | 75 | 75 |
| Frequency of fugitive dust inhalation | h day$^{-1}$ | 16/8 (indoor/outdoor) | 8/8 (indoor/outdoor) |
| Time indoors | h day$^{-1}$ | 21.14 | 22.86 |
| Time outdoors | h day$^{-1}$ | 2.86 | 1.14 |

type of land use should be included, since human behaviour is decisively dependent on the area of concern. For example, the pathway soil-crop-human may play a significant role with reference to gardens and agriculturally used areas, but need not be taken into account in the case of the land-use type nature reserve and city park. In turn, the latter, which are often located on contaminated land formerly used industrially should take part in the pathway soil-groundwater-human, while small gardens and cropland will probably not be sources of typical industrial organic pollutants and thus they will not be important in relation to the groundwater pathway.

Because of different contact times and intensiveness of use it makes sense to differentiate between residential areas (with mostly ornamental gardens), playgrounds, kitchen and vegetable gardens, agriculture, nature reserves, parks, recreation areas and sports fields, as well as buffer greens between high rise buildings, industrial plots and traffic routes, as done in the Dutch risk assessment scenarios. Playgrounds as the most sensitive use include every place where children are allowed to play such as playing fields, kindergartens, elementary school yards and, clearly, gardens. Gardens in residential areas may demonstrate both risk pathways to children through direct contact and consumption of food, if vegetables and fruits are grown. The food chain pathway has by far the most significance in the context of kitchen and vegetable gardens, where crops are grown for own consumption. Therefore, in contrast to gardens in residential areas with an abundance of ornamental plants and an expected percentage of only 10% of the yearly diet, in kitchen and vegetable gardens it is supposed that up to 50% of the yearly consumed vegetables are grown in these own gardens.

Contact time and possible soil ingestion rate are much shorter and smaller in buffer greens between industrial buildings and traffic routes and the areas are not places where children should play. Nature reserves are areas with low contact frequency as well. On the other hand, some areas such as green ones between high rise buildings, recreation areas, parks and sports fields are more intensively used. In Table 7.8 the distinct pathways are related to the urban land-use types in question.

**Table 7.8** Different pathways in association with land-use types in urban environments

| Land-use type | Examples | Direct contact (children) Oral ingestion | Inhalation | Dermal contact | Food consumption | Groundwater usage (from site-specific well) |
|---|---|---|---|---|---|---|
| Playgrounds | City playground | X | X | X | – | – |
| | Kindergarten | X | X | X | – | – |
| | School yard | X | X | X | (X) | (X) |
| Gardens | Ornamental garden | X | (X) | (X) | – | (X) |
| | Kitchen and vegetable garden | X | X | X | X | (X) |
| Vegetated area | Park | (X) | (X) | (X) | – | – |
| | Recreation area | (X) | (X) | X | – | – |
| | Outdoor swimming pool | (X) | (X) | X | – | (X) |
| | Sports field | (X) | X | X | – | – |
| | Residential area (high rise buildings) | (X) | (X) | (X) | – | – |
| | Industrial plot | – | (X) | – | – | – |
| | Traffic route | – | (X) | – | – | – |
| Nature reserves | | – | – | – | – | – |
| Urban agriculture | | – | (X) | – | X | X |
| Urban forest | | (X) | – | – | (X) | – |

X = pathway of relevance; (X) = pathway of minor relevance; – = pathway not relevant

## *7.3.3 Main Pathways*

The main pathways existing predominantly in urban areas with high population density are discussed in the following text.

### 7.3.3.1 Direct Contact

The exposure to receptors includes ingestion of soil, indoor inhalation of contaminants derived from soil vapour and outdoor dust inhalation by the respiratory tract, as well as absorption through the skin after dermal contact with contaminated soil.

The direct contact is mainly focused on the oral ingestion, because after release into the digestive tract the contaminants can be absorbed into the body. With reference to the children's behaviour the ingestion can occur during licking of the fingers or other media. It is well-known that young children up to 6 years have a significant hand-to-mouth behaviour. Some children younger than 6 years old can eat larger quantities of soil and other dirty objects up to 30 g day$^{-1}$ (so-called *Pica* behaviour) (Asante-Duah 1996; Brand et al. 2007).

In general, volatile contaminants that are not attached to the soil matrix, such as volatile organic hydrocarbons and methane, can evaporate and consequently be inhaled as gases. Apart from the gaseous contaminants in relation to the inhalative pathway the particulate matter with diameter <10 μm plays a major role, because it represents the respirable particulate percentage (see Section 3.2.2).

In contrast to the pathways mentioned above the dermal uptake through contact to contaminated soil is usually a negligible one. It is possible for some contaminants to be absorbed into the human skin. Afterwards they can be taken up by blood vessels and transferred into the interstitial part of the body. In the first instance, this problem is connected to lipophillic organic contaminants (Asante-Duah 1996; Brand et al. 2007).

#### 7.3.3.2 Food Chain Pathway

The exposure to receptors consists of consumption of crops contaminated by deposition onto the plant or by plant root uptake, ingestion of milk containing chemicals, consumption of livestock exposed by ingestion of contaminated material and consumption of aquatic animals exposed to contaminants as a result of discharging into surface water bodies.

Primarily, the plant uptake receives attention in risk assessment. The exposure to humans is linked to the concentration of contaminants in the crops and the amount consumed. Another criterion refers to the total eatable vegetation that really comes from the contaminated soil. Thus, the relation between vegetables originating from one's own garden and vegetables bought in supermarket should be taken into account. In the Dutch risk assessment model the percentage of vegetables from one's own garden is calculated at 10% for leafy crops and root crops respectively, for both children and adults (Brand et al. 2007).

Independent of the pathway of dust deposition or plant uptake by roots, only the contamination of the plant components that are going to be eaten are of interest. For instance, the contaminant accumulation in leaves of apple trees does not play a considerable role, in contrast to the apple fruits. The degree of contamination due to deposition can be strongly reduced, if the crops are properly washed. Then, the pathway is probably minimized to a great extent (Asante-Duah 1996; Brand et al. 2007).

Up to now relatively few results have been presented which deal with the plant transfer of contaminants in anthropogenically deposited soils. Deposited soils with a high percentage of technogenic materials and consequently a high contamination level can be assumed to cause enhanced plant uptake of metals. The example of a former industrially used brownfield in Essen (Germany) with a population of 580,000 indicated a considerable transfer into plant tissue of grasses. Heavy metals were accumulated to a great extent, the mean values were 1.3 mg Cd kg$^{-1}$ DM (dry matter) (background values <0.1–1.0 mg Cd kg$^{-1}$), 11.0 mg Ni kg$^{-1}$ DM (<0.1–5.0 mg

Ni kg$^{-1}$), 20.8 mg Pb kg$^{-1}$ DM (<0.1–5.0 mg Pb kg$^{-1}$) and 384 mg Zn kg$^{-1}$ DM (25–250 mg Zn kg$^{-1}$) respectively (Hiller and Meuser 1998). Apparently, plants can absorb and take up relatively high metal concentrations without any visible damage symptoms such as chlorosis, necrosis and growth degeneration. In relation to crop plants it can be concluded that such symptoms and yield decrease has not to be recognized, if contaminant accumulation in plant tissue already occurs.

A potentially high plant transfer is not only restricted to wild plants as investigated on the fallow land mentioned. Some results of contaminant accumulation in vegetables and fruits grown in urban areas are presented to help with the assessment of the food chain pathway.

In the urban environment many community gardens are established on sites which have a high heavy metal content and are accordingly unsuitable for gardening purposes (Fig. 7.3). Food quality may be reduced when metals enter garden products and are consumed by humans. Apart from the direct effect on human health, plant physiology and nutrient balance are disturbed.

The negative impact of cadmium on human health and its high mobility in soils has aroused special interest in this element. A cadmium overdose is associated with kidney diseases and, moreover, pregnant women and older people are particularly affected. Since dietary risk is scientifically linked to the intake of an average person consuming 2,000 calories daily and with a body weight of 70 kg, the calculated intake of nutrients and unwanted toxic substances like cadmium of the problematical group of people might be possibly inappropriate.

Cadmium appears to be a good indicator for the pathway soil-plant-human. In general, for assessment purposes it is necessary to choose typical garden products and to take the right portion of the plants (tuber, leaf, fruit, seed) which will be consumed into consideration. If leaves tend selectively to enhance accumulation of

**Fig. 7.3** Community garden area in New York, USA in proximity to a contaminated dumpsite

cadmium but the leaves are not intended for human consumption (e.g. carrots), it does not make sense to have a look at the leaf uptake. In the past it has been discovered that some leafy plants such as spinach and lettuce, which are grown in many countries and capable of several harvests during one season, are very good indicators for cadmium uptake, because they accumulate metals in the edible portion.

Topsoils of urban gardens, urban parks and mixed sites, where lettuce was planted, were investigated to find out dietary risk in different towns in the United States (exact locations are not published). The soils showed mean concentrations of 2.7 mg $kg^{-1}$ (mixed urban), 1.8 mg $kg^{-1}$ (urban gardens) and 0.5 mg $kg^{-1}$ (urban parks) for cadmium and 228 mg $kg^{-1}$ (mixed urban), 180 mg $kg^{-1}$ (urban gardens) and 82 mg $kg^{-1}$ (urban parks) for zinc, which is chemically similar to cadmium. The risk assessment was based upon the empirically determined Cd/Zn ratio, which should not be higher than 0.015. In all areas of investigation the ratio (0.010–0.012) did not exceed this threshold. However, at some sites revealing cadmium values above 3.0 mg $kg^{-1}$ it was possible to exceed the threshold. Sites with a mean Cd concentration of 3.45 mg $kg^{-1}$ and a mean Zn concentration of 212 mg $kg^{-1}$ exhibited a ratio of 0.016, leading to endangered human health (Scheyer 2000).

Investigations in London, the capital of the United Kingdom with 13,200,000 inhabitants and Newcastle with 270,000 inhabitants, revealed that lead concentrations in lettuce and radish increased significantly with soil values. In the case of radish it was expected that this tendency was caused by plant uptake from contaminated soil. In contrast, it was possible to remove between 40% and 80% of the Pb content in lettuce by washing. Obviously, most of the lead content resulted from foliar deposition from sources in the air. Plants with a large leaf surface can be contaminated based on airborne pollutant deposition. Preparation in the kitchen, however, might reduce this problem to a great extent. Typical plants indicating foliar deposition are lettuce and other large-leaved species such as cabbage, kale and spinach (Thornton 1991).

One hundred and five garden topsoils (0–30 cm) belonging to a coal mining area and industrially influenced environment were sampled and analyzed in North-Eastern France (Lorraine). The sampled sites were located in close proximity to chemical plants, coking plants, power stations and coal mines. Results were compared with the garden soil inventory of industrialized areas in South-Western Germany compiled 10 years previously (Crößman and Wüstemann 1992, cited in Schwartz et al. 2000) (Table 7.9). In fact, the mean values were comparable and revealed accelerated levels. One interesting aspect related to the age of the gardens considered. With increasing age the Pb and Zn concentration indicated higher values.

With respect to the German research project lettuce was collected in 32 gardens to evaluate plant uptake. Gardens which are used solely for people's own production of vegetables were chosen. In lettuce the concentration of the relatively mobile elements was 0.6 (0.2–4.7) mg $kg^{-1}$ DM for cadmium, 8.4 (5.2–52.2) mg $kg^{-1}$ DM for copper and 44 (19–103) mg $kg^{-1}$ DM for zinc. The transfer coefficient (concentration factors of Cd and Zn defined as plant concentration divided by soil concentration) was mainly influenced by soil pH value. The factors lay below 0.5 in soils with pH values higher than 6.0 and increased up to 1.6 in soils with pH values below 6.0. In the

**Table 7.9** Heavy metal concentration (mg $kg^{-1}$) in garden topsoils of 105 French and 3,624 German gardens in industrialized areas, where vegetable consumption from own plot occurs (Data from Schwartz et al. 2000)

| | Cd | Cu | Ni | Pb | Zn |
|---|---|---|---|---|---|
| | (mg $kg^{-1}$) | | | | |
| North-Eastern France | | | | | |
| Mean | **1.0** | **27** | **19** | **59** | **138** |
| Minimum | 0.2 | 4 | 4 | 1 | 37 |
| Maximum | 5.3 | 181 | 56 | 340 | 518 |
| South-Western Germany | | | | | |
| Mean | **0.5** | **24** | **14** | **65** | **151** |
| Minimum | 0.1 | 4 | 2 | 8 | 14 |
| Maximum | 7.3 | 196 | 69 | 627 | 1,035 |

Metal extraction method: aqua regia

**Table 7.10** Transfer coefficient for Cd dependent on soil characteristics; mean values from vegetables, fruits and potatoes (Data from UBA 1999)

| | pH ($CaCl_2$) < 5.5 | | | pH ($CaCl_2$) 5.5–6.5 | | | pH ($CaCl_2$) > 6.5 | | |
|---|---|---|---|---|---|---|---|---|---|
| | Carbon content (humus) | | | | | | | | |
| Clay content | <1% | 1–3% | >3% | <1% | 1–3% | >3% | <1% | 1–3% | >3% |
| <15% | 3 | 2.5 | 2 | 1.6 | 1.2 | 0.8 | 1.2 | 0.8 | 0.5 |
| 15–40% | 2.5 | 2 | 1.5 | 1.2 | 0.8 | 0.5 | 0.8 | 0.5 | 0.3 |
| >40% | 2 | 1.5 | 1 | 0.8 | 0.5 | 0.3 | 0.5 | 0.3 | 0.1 |

presence of acidic soils, it should be noted that it is necessary to include a minimizing of risk to human health, particularly if people consume mainly vegetables from their own gardens. Thus, guidelines and recommendations published by public authorities should help to inform people (Schwartz et al. 2000).

The incorporation of metals in plant tissue is strongly influenced by the soil characteristics, in particular pH value, clay content and humus content. The context is presented for the mobile element cadmium in Table 7.10. The mean transfer coefficient of 0.8, defined as total plant concentration divided by total soil concentration, ranged from 0.1 (humus-rich, clay-rich and alkaline soil) to 3.0 (opposite conditions) (UBA 1999).

In addition, a ranking in relation to the different elements and the plant organs consumed can be seen. In general, plants where the roots are eaten (e.g. celery tube, carrot) accumulate the highest amounts, followed by leafy vegetables (e.g. spinach, cabbage) and finally by plants where the fruits are consumed (e.g. tomato, bean, all species of cereals and fruits). With reference to the different elements cadmium and zinc they are designated as elements with a high accumulation rate. Nickel and copper are moderate, while arsenic, chromium, mercury and lead did not tend to accumulate to a great extent. In the published Ordinance of the European Union, which has been restricted to the two elements Cd and Pb up to now, the differences in relation to the plant organ to be consumed and to the elements listed are taken into consideration. The defined quality standards for food crops are presented in Table 7.11 (EC 2001).

**Table 7.11** Maximal permissible Cd and Pb concentration of food crops (mg kg$^{-1}$ fresh weight) in the European Union (Data from EC 2001)

| | |
|---|---|
| Cadmium | |
| Cereals (apart from wheat) | 0.1 |
| Wheat | 0.2 |
| Vegetable, fruits (apart from leafy and root vegetable, celery tube) | 0.05 |
| Leafy vegetable, celery tube, mushrooms | 0.2 |
| Root vegetable, potato | 0.1 |
| Lead | |
| Cereals | 0.2 |
| Vegetable, potato (apart from cabbage, leafy vegetable, mushrooms) | 0.1 |
| Cabbage, leafy vegetable, mushrooms | 0.3 |
| Fruits (apart from soft fruits) | 0.1 |
| Soft fruits | 0.2 |

**Table 7.12** Heavy metal concentration of 357 allotments, garden topsoils (mg kg$^{-1}$) and taproots of red beets (mg kg$^{-1}$ DM) in Poznan, Poland (Data from Grzebisz et al. 2000)

| | Cd | Cu | Pb | Zn |
|---|---|---|---|---|
| Garden topsoils | | | | |
| Mean | **0.9** | **20** | **36** | **125** |
| Minimum | <0.1 | 2 | 8 | 17 |
| Maximum | 4.7 | 256 | 266 | 1,067 |
| Red beets | | | | |
| Mean | **0.5** | **9.8** | **4.8** | **57.4** |
| Minimum | 0.1 | 2.8 | 0.9 | 8.9 |
| Maximum | 1.5 | 74.0 | 16.9 | 166.2 |

Metal extraction method (soil): aqua regia

Allotments in Poznan, Poland with a population of 640,000 located in close proximity to traffic routes and elsewhere and covering up to 4.2% of the city area were investigated in relation to the soil-plant pathway. Taproots of red beets give information about the endangering of consumers. It has been found that the plants accumulated the four analyzed metals and, except for copper, the tissue concentration exceeded in a lot of cases the limit values for consumption established by the Polish Ministry of Health and Social Care (Cd: 0.6 mg kg$^{-1}$ DM, Pb: 4.0 mg kg$^{-1}$ DM, Zn: 80.0 mg kg$^{-1}$ DM) (Table 7.12). The lead accumulation seemed to be relatively moderate. The reason for this was the sampling sites, which were partly chosen far away from busy roadways. Although the zinc concentration in garden soils lay occasionally above 200 mg kg$^{-1}$, the analyzed plants (apart from red beets wheat was also included) did not reveal any growth disturbances. So, a zinc value higher than 200 mg kg$^{-1}$, frequently described as a critical threshold, was obviously tolerable for crops (Grzebisz et al. 2000).

The soils in industrialized areas in France, Germany, Poland, the United Kingdom and the USA were predominantly affected by atmospheric pollutant deposition. A further input of contaminants affecting garden soils with vegetable plots results from the application of city effluent, a common practice in poorer countries deficient in wastewater treatment plants.

In Faisalabad (Pakistan) with 2,800,000 inhabitants a large number of wastewater channels, which are connected to each other, carry city effluent. This dirty water is used for field irrigation purposes. The cadmium concentration analyzed ranged from 0.01 to 0.04 mg $L^{-1}$ and showed an alkaline reaction (pH 7.7). Irrigated soils indicated values in the upper 15 cm between 0.25 and 0.34 DTPA (diethylenetriaminepentaacetic acid) extractable cadmium; total concentrations are unknown. The concentrations were under the phytotoxic level for most of the crops. Generally speaking, it was not possible to discover visible plant damage. Vegetables grown on irrigated soils in urban areas were analyzed. It was possible to distinguish higher values in leaves than in fruits. There was no detectable yield decline. On the contrary, soils with raised Cd concentrations tended to accumulate more cadmium without any yield depression. In addition, obvious adverse effects on human health were unknown in areas where people consume vegetables after irrigation with city wastewater, but there have been no systematic investigations up to now. Nevertheless, excessive exposure to cadmium should be avoided in any case, because various illnesses are associated with cadmium input such as gastroenteritis, renal tubular dysfunction, hypertension, cardiovascular disease, pulmonary emphysema and cancer development (Quadir et al. 2000).

Vegetables grown in soils with enhanced metal concentration cause danger to human health, if the people cover their daily vegetable consumption mainly from their own garden. As the comparison between vegetables grown on sites with soil contamination and those bought at market-places in the African city of Dar es Salaam (Tanzania) with a population of 3,100,000 showed, this context should be taken into account (Table 7.13). Poorer people dependent on their own vegetables for preparing lunch and dinner are strikingly exposed to the problematical food compared to people who can buy food at market-places or in supermarkets. The metal loading of vegetable samples bought at Kariakoo market was mostly lower than the vegetables sampled in soils alongside Sinza River and Msimbazi River (e.g. cobalt, nickel, lead, zinc). The rivers passed through industrialized urban land,

**Table 7.13** Heavy metal concentration (mg $kg^{-1}$ DM) of vegetables from selected sources and from river sediment (mg $kg^{-1}$) in Dares Salaam, Tanzania (Data from Luilo 2000)

| Metal | Kariakoo market-place | Sinza river floodplain | Msimbazi river floodplain | Msimbazi river (sediment) |
|---|---|---|---|---|
| | Vegetables | | | |
| Cd | 0.2 | 0.2 | 0.3 | 1.9 |
| Cr | 2.1 | 1.7 | 2.8 | 63.4 |
| Co | 0.7 | 4.2 | 4.5 | 30.7 |
| Cu | 8.1 | 6.1 | 9.1 | 28.6 |
| Ni | 2.0 | 3.9 | 4.8 | 23.7 |
| Pb | 1.8 | 4.9 | 3.0 | 59.2 |
| Zn | 25.8 | 37.2 | 32.1 | 676.4 |

discharging hazardous substances, which may enhance the concentration of the garden soils in the floodplain as well as of the sediments of one river analyzed (see Section 3.3.3). In sediments the high levels of metals probably created adverse ecological effects. With respect to human beings vegetables grown around the city of Dar es Salaam might also pose a health risk to the consumers (Luilo 2000).

The soil-to-plant transfers of Polycyclic Aromatic Hydrocarbons (PAH) were taken into consideration in a greenhouse experiment with contaminated industrial soil in France. Lettuce, potatoes and carrots were chosen and different PAH contents compared. In general, in the aerial part the $PAH_{EPA}$ results varied from 0.2 to 2.7 mg $kg^{-1}$ DM. In the belowground part of the plants the PAH values were much higher – up to 22 mg $kg^{-1}$ DM. There are several reasons for recognizable PAH concentration in plant tissue, root uptake and following translocation, dust deposition onto the aerial part, chemical transformation and metabolism. For lettuce PAH concentration in plants and soils correlated significantly (aerial part: $r^2 = 0.57$, roots: $r^2 = 0.83$). Thus, root uptake and translocation from root to shoot were able to explain the PAH accumulation. It was possible to discover a correlation between peeled roots of carrots and soil content ($r^2 = 0.59$) in contrast to peeled potato tubers. It was supposed that in the case of carrots, in spite of washing and peeling, a strong adsorption of PAH was attributed to root epidermis, because carrots are both storage organ and root. With reference to potatoes it was not possible for this to be a reason and a translocation from leaves to tubers definitely did not occur (Fismes et al. 2000). In general, an accumulation of lipophilic organic pollutants like PAH cannot be excluded, particularly in relation to the plant organs growing below the soil surface. Concentrations exceeding 1 μg $kg^{-1}$ DM are reckoned to be problematical.

#### 7.3.3.3 Groundwater Pathway

The exposure to receptors is mainly linked to the use of contaminated groundwater that is used for public water supply. The direct pathway to humans is always present in the case of drinking water. Particular attention must be paid to liquid contaminants such as total petroleum hydrocarbons (TPH) and volatile chlorinated organic carbons such as trichloroethylene (TCE) and perchloroethylene (PCE).

Soils containing construction debris resulting from demolition of industrial buildings tend towards slow release of liquids bound in the solid matrix. Accordingly, a continuous leaching of the liquids takes place, endangering the aquifer (Fig. 7.4).

The migration into the groundwater depends on groundwater temperature and specific gravity of the compound. With increasing water temperature the viscosity decreases and the mobility grows. Contaminants with a specific gravity less than 1.0 g $cm^{-3}$ such as TPH (mineral oils) and benzene are up-welling, parameters falling below 1.0 g $cm^{-3}$ such as the chlorinated hydrocarbons are down-welling. The migration does not take place continuously – oils, for example, can form their own separated oil phase that swims with a velocity different from the groundwater (Fig. 7.5).

**Fig. 7.4** Soil profile in Gelsenkirchen, Germany consisting of industrial construction debris in which tar residues (*circles*) exist; the tar is moving downward using channels left by earthworms (*arrow*)

Furthermore, the migration is associated with the aquifer type present. Pore water aquifers like terrace sand and terrace gravel covering floodplain terrains exhibit a high texture-related hydraulic conductivity of more than $10^{-4}$ m $s^{-1}$. The water movement, however, can be limited because of the low slope gradient of the landscape. Crack water aquifers such as weathered sandstone and siltstone as well as karst water aquifers such as limestone, dolomite stone and gypsum stone show differing hydraulic conductivity depending on the weathering progress. The flow velocity, however, can be high due to the mountainous locations present.

Moreover, the exposure should not only involve the drinking water consumption, the inhaling of water vapour when showering and bathing as well as the dermal contact can play an additional role. But exposure scenarios are mostly restricted to only the drinking water uptake, which can be specified more or less exactly. The drinking water consumption, for instance, is calculated as being 1 $dm^3$ $day^{-1}$ (child) and 2 $dm^3$ $day^{-1}$ (adult) in the Dutch risk assessment model. In the same model other perhaps too luxurious prevailing conditions are estimated, e.g. residence time in bathrooms 0.5 h $day^{-1}$ and showering time 0.25 h $day^{-1}$ for children and adults. Moreover, if a contaminated site touches drinking water pipelines, it cannot be excluded that organic pollutants permeate through the pipes and contaminate the flowing water inside. Therefore, in order to carry out a complete risk assessment factors such as the duration of water stagnation in the pipes, radius of pipes and thickness of pipe walls should be included (Asante-Duah 1996; Brand et al. 2007).

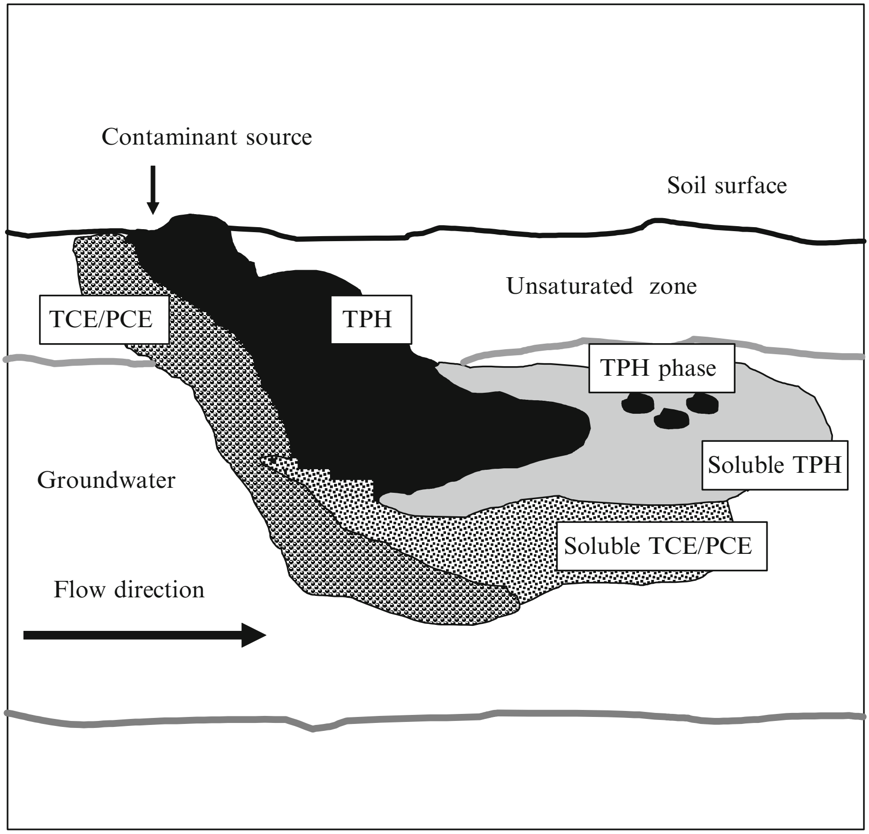



**Fig. 7.5** Soil-groundwater contaminant migration in the system soil – groundwater. TCE = trichloroethylene; TPH = total petroleum hydrocarbons; PCE = perchloroethylene

## 7.4 Assessment Based on Quality Standards

### *7.4.1 Definitions*

Thresholds are defined as the lowest dose of a contaminant at which a measurable effect is observed. The threshold limit means a chemical concentration above which adverse health effects occur or other environmental receptors like groundwater and surface water are endangered. In the first instance, risk assessment is linked to human health and secondly to environmental protection.

Quality standards are defined and published with reference to political and governmental processes. They are used in decision-making processes but they should not be viewed as absolute numbers. A set of fixed levels cannot embrace every risk management need of individual sites. In general, it is relatively easy to apply quality standards for decision-making purposes. But it may not happen that the fundamental understanding of the polluted system is reduced to quality standards only. Because the decisions about estimation and necessary remediation of a site should be risk-based and should take into consideration the complicated pollution systems, experts have to do more than the simple comparison of analyzed values and quality standards. In any case, experts should implement physical and chemical knowledge for assessment purposes. For instance, in the case of high cadmium concentration in the unsaturated zone, the pathway soil-groundwater must be assessed differently depending upon the pH value, texture, organic matter content and other soil properties the expert has additionally analyzed (Nathanail and Bardos 2004) (see Section 6.2).

In order to implement threshold limits a lot of countries have defined quality standards aimed at different levels. In general, quality standards are differentiated into:

- Trigger values at which further investigation procedures have to take place, since risk to human health and environmental receptors cannot be excluded
- Action values at which clean-up procedures or other treatments become necessary, since (acute) danger is present
- Precautionary values which are aimed at avoidance of soil contamination in future and hence the values are oriented to precautionary soil protection

## *7.4.2 Published Quality Standard Catalogues*

Some developed countries in Western Europe (e.g. Denmark, Germany, The Netherlands, United Kingdom), Asia (e.g. Japan, Korea) and Australia have determined quality standards. These are published in Soil Protection Acts and soil protection ordinances. In these countries the standards are, in principle, obligatory but their application varies. In some countries, for instance in (former) socialist Eastern European states such as Poland and Romania and Asian countries (e.g. China), the standards have only the status of orientation values without any compulsory nature and a low level of application up to now. However, it should be noted that in most countries, in particular in Africa and South America, quality standards have not been established at all.

In general, some important principles associated with the different national quality standards applied should be mentioned:

- Apart from the soil matrix further media are frequently involved in quality standard definition, e.g. soil vapour and groundwater.
- The amount of contaminant parameters ranges from country to country; most lists are mainly focused on heavy metals extracted by aqua regia and, to a less extent, on organic pollutants.
- With reference to the different pathways involved the standards are differentiated into land-use types covering highly sensitive to non-sensitive uses; in particular, the urban land-use types are taken into consideration in detail, because contaminated land is usually predominantly linked to these types.

- In relation to the values for the soil matrix chemical parameters determining the mobility and migration of the contaminants in question are included such as pH value, organic matter content and texture. Furthermore, different extraction means are recommended in order to reproduce the distinct migration pathways.
- With reference to the receptors, particularly the differences between human beings and environment, the quality standards are listed separately.

The structure and background of quality standards in several countries are presented in the following. The presentation focuses mainly on differences between urban land-use types and it will predominantly take contaminants into account which were discussed in the chapters above. Accordingly, the published lists are not shown completely.

The quality standards of Japan, qualified in the Japanese Soil Protection Act, involve both total concentrations and water-extractable metal concentrations (Table 7.14). Land-use types are not mentioned and the parameters in question refer generally to all types except sites with enhanced geogenic values, dumpsites and storage areas for toxic substances. Besides the parameters listed, some anionic compounds such as fluorides and borates, other volatile chlorinated hydrocarbons, organic phosphorus compounds as well as some pesticides are mentioned (MOE 2006).

The quality standards of the Danish Contaminated Soil Act are summarized in Table 7.15. The list is much longer and parameters such as additional volatile

**Table 7.14** Quality standards of the Soil Protection Act in Japan (Data from MOE 2006)

| Substance | Total concentration ($mg\ kg^{-1}$) | Water-extractable concentration ($mg\ L^{-1}$) |
|---|---|---|
| Metals | | |
| Arsenic | 150 | 0.01 |
| Cadmium | – | 0.01 |
| Chromium VI | 250 | 0.05 |
| Copper | 125 | – |
| Lead | 150 | 0.01 |
| Mercury | 15 | 0.0005<br>nd (organic Hg) |
| Selenium | 150 | 0.01 |
| Cyanides | | |
| Cyanide compounds | 50 | nd |
| Volatile chlorinated hydrocarbons | | |
| 1,2-Dichloroethane | – | ≤0.004 |
| 1,1-Dichloroethene | – | ≤0.020 |
| Dichloromethane | – | ≤0.020 |
| Tetrachloroethylene | – | ≤0.010 |
| 1,1,1-Trichloroethane | – | ≤1.0 |
| 1,1,2-Trichloroethane | – | ≤0.006 |
| Monoaromates | | |
| Benzene | – | ≤0.010 |
| Polychlorinated compounds | | |
| PCB | – | nd |

nd = not detectable

**Table 7.15** Quality standards of the Danish Soil Protection Act for soil, groundwater and soil vapour (Data from DEPA 2005)

| Substance | Soil quality criteria (mg kg$^{-1}$) | Cut-off value (mg kg$^{-1}$) | Groundwater quality criteria (µg L$^{-1}$) | Air quality criteria (mg m$^{-3}$) |
|---|---|---|---|---|
| Metals | | | | |
| Arsenic | 20 | 20 | 8 | – |
| Cadmium | 0.5 | 5 | 0.5 | – |
| Chromium (VI) | 20 | – | 0.1 | – |
| Chromium (III + VI) | 500 | 1,000 | 25 | – |
| Copper | 500 | 1,000 | 100 | – |
| Nickel | 30 | 30 | 10 | – |
| Lead | 40 | 400 | 1 | – |
| Thallium | 1 | – | – | – |
| Tin | 500 | – | – | – |
| Zinc | 500 | 1,000 | 100 | – |
| Cyanides | | | | |
| Cyanides (acid volatile) | – | 10 | – | 0.06 |
| Cyanides (inorganic) | 500 | – | 50 | – |
| Volatile chlorinated hydrocarbons | | | | |
| 1,2-Dichloroethane | 1 | – | 1 | $1 \times 10^{-4}$ |
| 1,1-Dichlororethane | 5 | – | 1 | 0.01 |
| Dichloromethane | 8 | – | 1 | 0.0006 |
| Tetrachloroethylene | 4 | – | – | 0.0003 |
| 1,1,1-Trichloroethane | 200 | – | 1 | 0.5 |
| Vinyl chloride | 0.4 | – | 0.2 | $4 \times 10^{-5}$ |
| Petroleum hydrocarbons | | | | |
| $C_5$-$C_{10}$ Hydrocarbons | 25 | – | 9 | – |
| $C_5$-$C_{35}$ Hydrocarbons | 100 | – | 9 | 0.1 |
| Monoaromates | | | | |
| Benzene | 1.5 | – | 1 | 0.00013 |
| Toluene | – | – | 5 | 0.4 |
| Xylene | – | – | 5 | 0.1 |
| Phenols | | | | |
| Phenols (total) | 70 | – | 0.5 | – |
| Chlorophenols (sum of mono, di-, tri- and tetraphenols) | 3 | – | 0.1 | $2 \times 10^{-5}$ |
| Polycyclic aromatic hydrocarbons | | | | |
| PAH (total) | 4[a] | 40[b] | 0.2 | – |
| Naphthalene | – | – | 1 | 0.04 |
| Benzo(a)pyrene | 0.3 | 3 | – | – |
| Dibenzo(ah)anthrazene | 0.3 | 3 | – | – |
| Pesticides | | | | |
| Dieldrine | – | – | 0.03 | – |
| Paraquat | 5 | – | – | 0.0002 |
| Parathion | 0.1 | – | 0.01 | – |
| Pentachlorophenol | 0.15 | – | 0.01 | $1 \times 10^{-6}$ |
| Lindan | 0.6 | – | 0.1 | – |
| Pesticides (total) | – | – | 0.05 | – |

(continued)

**Table 7.15** (continued)

| Substance | Soil quality criteria (mg kg$^{-1}$) | Cut-off value (mg kg$^{-1}$) | Groundwater quality criteria (μg L$^{-1}$) | Air quality criteria (mg m$^{-3}$) |
|---|---|---|---|---|
| Pesticides (individual) | – | – | 1.05 | – |
| Phtalates | | | | |
| Phtalates (with DEHP) | 250 | – | 1 | – |
| Di-(ethylhexyl) phthalate, DEHP | 25 | – | 1 | – |

[a]Soil: sum of benzo(a)pyrene, benzo(b + j + k)fluoranthene, dibenzo(ah)anthracene, fluoranthene and indeno(1, 2, 3-cd)pyrene
[b]Sum of benzo(a)pyrene, benzo(b + k)fluoranthene, fluoranthene, indeno(1, 2, 3-cd)pyrene and benzo(ghi)fluoranthene

organic hydrocarbons, monoaromates, phenols and pesticides as well as anionic parameters (e.g. borate, molybdate, fluoride) and tensides are added. In addition to the quality criteria, the guideline contains further important threshold values, the co-called Cut-off Values. The Cut-off Values and the quality criteria for soils refer to areas with highly sensitive land use (e.g. terraced houses, playgrounds). Cut-off Values are only established for contaminants that are immobile or slowly degradable. If the concentration of a specific contaminant is below this value, remediation is not necessary. However, if Cut-off Values are exceeded in areas with sensitive land use, remediation measures are obligatory. Apart from the soil matrix, quality standards for separately sampled groundwater and soil vapour are included. Soil vapour concentrations are limited to potentially volatile parameters only.

The well-structured and organized Danish Act regulates the prevention, elimination or reduction of soil contamination and the hindrance or prevention of the detrimental impact of soil contamination on groundwater, human health and the environment. The objectives of the law are groundwater protection and the prevention of human health problems due to the use of the contaminated areas. Groundwater protection has a very high priority in Denmark, because the water supply originates almost entirely (99%) from groundwater sources (DEPA 2001).

The Contaminated Soil Act is based on three main principles. The first principle is the provision against detrimental effects arising from soil contamination. This is mainly achieved by identifying, listing and mapping of contaminated sites. The second principle is the prevention of further pollution of the environment through the use and disposal of soil. This is realised by a soil management system. In the order to transport excavated soil from listed properties the local council must be informed. It is their duty to give instructions on how to dispose of soil excavations and how to announce for the documentation of the soil quality. The third principle is the Polluter Pays Principle. The environmental authority can give an enforcement notice to carry out sampling, analyses and measurements of substances in order to investigate the causes or effects of contamination. Furthermore, an enforcement notice to clean up pollution can be issued by the authorities.

In the Guideline on Remediation of Contaminated Sites, the handling of contaminated sites is divided into four phases, the initial survey phase, the investigation phase, the remediation phase and the operation and evaluation phase. After each phase it will be assessed whether further measures are necessary. The objective of the risk assessment is to verify the need for remediation. For the evaluation of risk assessments, quality criteria like the local geology and land-use type, among others, are used (DEPA 2005). In principle, the Danish approach can be found in different soil protection acts in Western European countries.

The Polish values based on a guideline (Table 7.16) are mainly focused on the pathway soil-groundwater and thus take the soil to a depth of 15 m into account. Accordingly, instead of the analysis of water-extractable concentrations, total

**Table 7.16** Quality standards of the Polish guideline (mg $kg^{-1}$) (Data from Dziennik 2002)

| | | Group B[b] | | | | | Group C[c] | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Depth below surface (m) | | | | | | | |
| | | | 0.3–15 | | >15 | | 0–2 | 2–15 | |
| | | | Hydraulic conductivity (m $s^{-1}$) | | | | | | |
| Substance | Group A[a] | 0–0.3 | > $10^{-7}$ | ≤ $10^{-7}$ | > $10^{-7}$ | ≤ $10^{-7}$ | | > $10^{-7}$ | ≤ $10^{-7}$ |
| Metals | | | | | | | | | |
| Arsenic | 20 | 20 | 20 | 25 | 25 | 55 | 60 | 25 | 100 |
| Barium | 200 | 200 | 250 | 320 | 300 | 650 | 1,000 | 300 | 3,000 |
| Cadmium | 1 | 4 | 5 | 6 | 4 | 10 | 15 | 6 | 20 |
| Chromium | 50 | 150 | 150 | 190 | 150 | 380 | 500 | 150 | 800 |
| Cobalt | 20 | 20 | 30 | 60 | 50 | 120 | 200 | 50 | 300 |
| Copper | 30 | 150 | 100 | 100 | 100 | 200 | 600 | 200 | 1,000 |
| Lead | 50 | 100 | 100 | 200 | 100 | 200 | 600 | 200 | 1,000 |
| Mercury | 0.5 | 2 | 3 | 5 | 4 | 10 | 30 | 4 | 50 |
| Molybdenum | 10 | 10 | 10 | 40 | 30 | 210 | 250 | 30 | 200 |
| Nickel | 35 | 100 | 50 | 100 | 70 | 210 | 300 | 70 | 500 |
| Tin | 20 | 20 | 30 | 50 | 40 | 300 | 350 | 40 | 300 |
| Zinc | 100 | 300 | 350 | 300 | 300 | 720 | 1,000 | 300 | 3,000 |
| Cyanides | | | | | | | | | |
| Cyanides (free) | 1 | 1 | 5 | 6 | 5 | 12 | 40 | 5 | 100 |
| Cyanides (complex) | 5 | 5 | 5 | 6 | 5 | 12 | 40 | 5 | 500 |
| Petroleum hydrocarbons | | | | | | | | | |
| C6-12 Hydrocarbons | 1 | 1 | 5 | 375 | 50 | 750 | 500 | 50 | 750 |
| C12-35 Hydrocarbons | 30 | 50 | 200 | 1,000 | 1,000 | 3,000 | 3,000 | 1,000 | 3,000 |
| Monoaromates | | | | | | | | | |
| Benzene | 0.05 | 0.1 | 0.2 | 25 | 3 | 50 | 100 | 3 | 150 |
| Ethyl benzene | 0.05 | 0.1 | 1 | 75 | 10 | 150 | 200 | 10 | 250 |
| Toluene | 0.05 | 0.1 | 1 | 75 | 5 | 150 | 200 | 5 | 230 |
| Xylene | 0.05 | 0.1 | 1 | 35 | 5 | 75 | 100 | 5 | 150 |

(continued)

**Table 7.16** (continued)

| | | Group B[b] | | | | | Group C[c] | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Depth below surface (m) | | | | | | | |
| | | | 0.3–15 | | >15 | | 0–2 | 2–15 | |
| | | | Hydraulic conductivity (m $s^{-1}$) | | | | | | |
| | | | > | ≤ | > | ≤ | | > | ≤ |
| Substance | Group A[a] | 0–0.3 | $10^{-7}$ | | $10^{-7}$ | | | $10^{-7}$ | |
| Polycyclic aromatic hydrocarbons | | | | | | | | | |
| Naphthalene | 0.1 | 0.1 | 5 | 20 | 10 | 40 | 50 | 10 | 40 |
| Benzo(a)pyrene | 0.02 | 0.03 | 5 | 10 | 5 | 40 | 50 | 5 | 40 |
| PAH (sum of 9) | 1 | 1 | 20 | 40 | 20 | 200 | 250 | 20 | 200 |
| Phenols | | | | | | | | | |
| Phenols (total) | 0.05 | 0.1 | 0.5 | 20 | 3 | 40 | 50 | 3 | 100 |
| Chlorophenol (individual) | 0.001 | 0.001 | 0.01 | 0.5 | 0.2 | 1 | 1 | 0.2 | 5 |
| Chlorophenol (sum) | 0.001 | 0.001 | 0.001 | 1 | 0.5 | 10 | 10 | 0.5 | 10 |
| Polychlorinated compounds | | | | | | | | | |
| PCB (sum) | 0.02 | 0.02 | 0.1 | 1 | 0.5 | 5 | 2 | 0.5 | 5 |
| Pesticides | | | | | | | | | |
| Atrazin | 0.00005 | 0.05 | 0.005 | 6 | 0.005 | 6 | 0.05 | 0.005 | 6 |
| DDT/DDE/DDD | 0.0025 | 0.025 | 0.025 | 4 | 0.025 | 4 | 0.25 | 0.025 | 4 |
| Pthalates | | | | | | | | | |
| Phtalates (sum) | 0.1 | 0.1 | 5 | 60 | 5 | 60 | 60 | 10 | 60 |

[a]Group A – Areas protected by Water Protection Act and Nature Protection Act; as long as the values are not exceeded further measures are not necessary
[b]Group B – Agriculturally used areas, forest, fallow land and urban areas except industrial, mining and traffic areas
[c]Group C – Industrial, mining and traffic areas

concentrations and the hydraulic conductivity are taken to assess the groundwater pathway. The hydraulic conductivity responsible for the contaminant migration downwards is included, as are the land-use types. The latter are restricted to a few modifications only. The three land uses include rural and urban sites. Group B contains agriculturally used areas and residential areas, avoiding a stringent differentiation between rural and urban land. Non-sensitive uses such as industrial and traffic sites allow the highest acceptable values. The list of pollutants contains rather a lot of parameters. For instance, apart from the parameters mentioned, further Polycyclic Aromatic Hydrocarbons, chlorinated monoaromates and pesticides are included (Dzinnek 2002).

The quality standards of the United Kingdom listed as Soil Guideline Values of the Contaminated Land Exposure Assessment are limited to a few parameters only but the differentiation of land use classes is more detailed. The values are only for general guidance and a full risk model should be used for specific contaminated sites (Table 7.17). It can be assumed that the standards are predominantly focused

**Table 7.17** Quality standards of the British guideline (mg kg$^{-1}$) (Data from DEFRA 2006)

| Substance | Residential areas with gardens | Residential areas without gardens | Allotments | Industrial and commercial areas |
|---|---|---|---|---|
| Metals | | | | |
| Arsenic | 20 | 20 | 20 | 500 |
| Cadmium | 1 (pH 6), 2 (pH 7), 8 (pH 8) | 30 | 1 (pH 6), 2 (pH 7), 8 (pH 8) | 1.400 |
| Chromium | 130 | 200 | 130 | 5,000 |
| Lead | 450 | 450 | 450 | 750 |
| Mercury | 8 | 15 | 8 | 480 |
| Nickel | 50 | 75 | 50 | 5,000 |
| Selenium | 35 | 260 | 35 | 8,000 |
| Phenols | | | | |
| Phenols (total) | 78 (1% OM) | 21,900 (1% OM) | 80 (1% OM) | 21,900 (1% OM) |
| | 150 (2.5% OM) | 34,400 (2.5% OM) | 155 (2.5% OM) | 43,000 (2.5% OM) |
| | 280 (5% OM) | 37,300 (5% OM) | 280 (5% OM) | 78,100 (5% OM) |

OM = organic matter

on urban areas where the degree of soil contamination might be more problematical. Apparently, the pathway soil-plant-human in addition to the direct soil contact seems to be very significant, since garden areas divided into sites with ornamental plants and with food crops as well as allotments are separately classified. Consequently, a division of distinct land-use types depending on the sensitivity of the uses improves the opportunity to assess urban soils. The British values consider some soil properties influencing the mobility of contaminants such as the pH value for the mobile element cadmium (see Section 7.3.3) and the organic matter content for phenols (DEFRA 2006).

The standards published in China (Table 7.18) are rarely applied and limited to a few parameters only. On the other hand, some mobility determining factors like the pH value and soil moisture are taken into account. For arsenic and chromium values between moist areas (paddy fields) and dry areas (e.g. cereal cropland) are distinguished, taking the different mobility of As and different toxicity of Cr into consideration. In paddy field soils arsenic becomes more mobile in reductive conditions, whilst chromium will be present as toxic Cr VI in dry soil conditions. With reference to the heavy metals it is pointed out that the quality standards are lower in the presence of alkaline soil reaction. The land-use types are related to typical agricultural and silvicultural ones, while urban land uses are not stated (SEPA 2006).

Quality standards in Romania based on a guideline (Table 7.19) contain only two urban land-use types. The standards define soil contamination in urban areas of different sensitivity. Rural land, however, is not of concern. For comparison purposes, background values are listed. The values are based on two kinds of quality standards for soils, namely the Trigger and the Action Values. They are defined

**Table 7.18** Quality standards of the Chinese guideline (mg kg$^{-1}$) (Data from SEPA 2006)

| | Grade I | Grade II | | | Grade III |
|---|---|---|---|---|---|
| pH($H_2O$) value | – | <6.5 | 6.7–7.5 | >7.5 | >6.5 |
| Metals | | | | | |
| As (paddy field) | 15 | 30 | 25 | 20 | 30 |
| As (dry land) | 15 | 40 | 30 | 25 | 40 |
| Cd | 0.2 | 0.3 | 0.3 | 0.6 | 1.0 |
| Cr (paddy field) | 90 | 250 | 300 | 350 | 400 |
| Cr (dry land) | 90 | 150 | 200 | 250 | 300 |
| Cu (cropland) | 35 | 50 | 100 | 100 | 400 |
| Cu (orchard) | – | 150 | 200 | 200 | 400 |
| Ni | 40 | 40 | 50 | 60 | 200 |
| Pb | 35 | 250 | 300 | 350 | 500 |
| Zn | 100 | 200 | 250 | 300 | 500 |
| Insecticides | | | | | |
| Insecticides (total) | 0.05 | 0.5 | 0.5 | 0.5 | 1.0 |

Grade I – Values close to nature
Grade II – Cropland, orchard, pasture (uses acceptable, if values fall below standards)
Grade III – Woodland and cropland near mining areas (uses acceptable, if values fall below standards)

**Table 7.19** Quality standards of the Romanian guideline (mg kg$^{-1}$) (Data from MOR 2007)

| | | Sensitive uses (residential areas, playgrounds) | | Industrial and commercial areas | |
|---|---|---|---|---|---|
| Substance | Background values | Trigger value | Action value | Trigger value | Action value |
| Metals | | | | | |
| Arsenic | 5 | 15 | 25 | 25 | 50 |
| Antimony | 5 | 12.5 | 20 | 20 | 40 |
| Barium | 200 | 400 | 625 | 1,000 | 2,000 |
| Beryllium | 1 | 2 | 5 | 7.5 | 15 |
| Cadmium | 1 | 3 | 5 | 5 | 10 |
| Chromium (total) | 30 | 100 | 300 | 300 | 600 |
| Cobalt | 15 | 30 | 50 | 100 | 250 |
| Copper | 20 | 100 | 200 | 250 | 500 |
| Lead | 20 | 50 | 100 | 250 | 1,000 |
| Mercury | 0.1 | 1 | 2 | 4 | 10 |
| Nickel | 20 | 75 | 150 | 200 | 500 |
| Selenium | 1 | 3 | 5 | 10 | 20 |
| Silver | 2 | 10 | 20 | 20 | 40 |
| Tin | 20 | 35 | 50 | 100 | 300 |
| Thallium | 0.1 | 0.5 | 2 | 2 | 5 |
| Vanadium | 50 | 100 | 200 | 200 | 400 |
| Zinc | 100 | 300 | 600 | 700 | 1,500 |
| Cyanides | | | | | |
| Cyanide (free) | <1 | 5 | 10 | 10 | 20 |
| Cyanide (complex) | <5 | 100 | 250 | 200 | 500 |

(continued)

**Table 7.19** (continued)

| Substance | Background values | Sensitive uses (residential areas, playgrounds) Trigger value | Action value | Industrial and commercial areas Trigger value | Action value |
|---|---|---|---|---|---|
| Petroleum hydrocarbons | | | | | |
| Total petroleum hydrocarbons | <100 | 200 | 500 | 1,000 | 2,000 |
| Monoaromates | | | | | |
| Benzene | <0.01 | 0.25 | 0.5 | 0.5 | 2 |
| Ethyl benzene | <0.05 | 5 | 10 | 10 | 50 |
| Toluene | <0.05 | 15 | 30 | 30 | 100 |
| Xylene | <0.05 | 7.5 | 15 | 15 | 25 |
| Phenols | | | | | |
| Phenol (total) | <0.02 | 5 | 10 | 10 | 40 |
| Polycyclic aromatic hydrocarbons | | | | | |
| Naphthalene | <0.02 | 2 | 5 | 5 | 50 |
| Benzo(a)pyrene | <0.02 | 2 | 5 | 5 | 10 |
| PAH (sum of 11) | <0.1 | 7.5 | 15 | 25 | 150 |
| Polychlorinated compounds | | | | | |
| PCB (sum of 7) | <0.01 | 0.25 | 1 | 1 | 5 |
| PCDD (total) | <0.0001 | 0.0001 | 0.001 | 0.0001 | 0.001 |
| PCDF (total) | <0.0001 | 0.0001 | 0.001 | 0.0001 | 0.001 |
| Pesticides | | | | | |
| DDT (total) | <0.15 | 0.5 | 1 | 1.5 | 4 |
| Chlorinated pesticides (total) | <0.2 | 1 | 2 | 2 | 5 |
| Triazines (total) | <0.1 | 1 | 2 | 2 | 5 |

in the same way as mentioned in Section 7.4.1. Whilst Trigger Values are aimed at examining whether further investigation procedures are necessary, the Action Values are intended to react as soon as possible. In harmony with the Polish list a lot of parameters are included. Apart from the parameters presented further mono-aromates and anionic compounds such as sulphate and borate are named. Neither the Polish list nor the Romanian lists take mobility-determining soil properties into consideration (MOR 2007).

In the quality standards of the Soil Protection Act of Korea Precaution Values are defined, because the Soil Protection Act is strongly oriented to the idea of precautionary soil protection (Table 7.20). The values are aimed at the prevention of soil contaminations in future. Land-use types are differentiated to a lesser extent and the parameter list is limited compared to most European quality standard lists. Sensitive land uses independent of the location city or rural land exhibit lower values than the non-sensitive land uses located mainly in urban areas (ECOREA 2007).

In Australia state organisations have developed methods and criteria for assessing impacts on ecology, groundwater and human health. Thus, the ecological investigation

**Table 7.20** Quality standards of the Soil Protection Act of Korea (mg $kg^{-1}$) (Data from ECOREA 2007)

| | Precaution value | | Action value | |
|---|---|---|---|---|
| Substance | Area A | Area B | Area A | Area B |
| Metals | | | | |
| Arsenic | 6 | 20 | 15 | 50 |
| Cadmium | 1.5 | 12 | 4 | 30 |
| Chromium VI | 4 | 12 | 10 | 30 |
| Copper | 50 | 200 | 125 | 500 |
| Lead | 100 | 400 | 300 | 1,000 |
| Mercury | 4 | 16 | 10 | 40 |
| Nickel | 40 | 160 | 100 | 400 |
| Zinc | 300 | 800 | 700 | 2,000 |
| Cyanides | | | | |
| Cyanides (total) | 2 | 120 | 5 | 300 |
| Phenols | | | | |
| Phenol (total) | 4 | 20 | 10 | 50 |
| Petroleum hydrocarbons | | | | |
| Total petroleum hydrocarbons (total) | 500 | 2,000 | – | 5,000 |
| Monoaromates | | | | |
| Benzene, Ethyl benzene, Toluene, Xylene (sum) | – | 80 | – | 200 |
| Volatile chlorinated hydrocarbons | | | | |
| Trichloroethylene | 8 | 40 | 20 | 100 |
| Tetrachloroethylene | 4 | 24 | 10 | 60 |

Area A – Cropland, paddy field, bamboo plantation, orchard, pasture, forest, nursery, water bodies, areas with water pipes, park (vegetated) sports field, recreational area, religious and cultural site

Area B – Industrial and commercial area, roadland, railway area, fallow land

level (EIL), groundwater investigation level (GIL) and human investigation level (HIL) are used as quality standards beyond which special risk assessments must be carried out. Table 7.21 displays some criteria for inorganic contaminants. Unfortunately, because of the enormous variety of geological materials in the Australian continent State Environment Protection Authorities ignore national standards and develop their own. This situation can also be discovered in other large countries with heterogeneous geology and numerous more or less powerful federal states such as Canada, China and USA.

It should also be noted that the entire focus is always on the known contaminant metals or metalloids and takes no account of other pollutants (see Section 4.3.2). Hence, private consultants, who carry out contamination assessments and site remediation, determine only those elements in laboratory analyses. Furthermore, soil properties that play a major role in immobilising the contaminant elements are not included. Likewise, the role of substances like chloride, which can render metals such as cadmium much more soluble and mobile by complexation

**Table 7.21** Quality standards of the Australian guideline (mg kg$^{-1}$) (Data from National Environment Protection Council (NEPC) of Australia, cited in Meuser and van de Graaf 2010)

| Substance | Ecological investigation level (EIL) | Human health investigation level (HIL) for standard residential areas |
|---|---|---|
| Metals | | |
| Antimony | 20 | – |
| Arsenic | 20 | 100 |
| Beryllium | – | 20 |
| Cadmium | 3 | 20 |
| Chromium (total) | 50 | – |
| Cobalt | – | 100 |
| Copper | 100 | 1,000 |
| Lead | 600 | 300 |
| Mercury (total) | 1 | 15 |
| Nickel | 60 | 600 |
| Tin | 50 | – |
| Zinc | 200 | 7,000 |

(Cd-chloro-complexation), or soil pH, which also affects specification and mobility, is ignored in these criteria. All soils are, as it were, regarded as being standard soils.

Therefore, intelligent assessment is deprived of relevant information. For contaminated urban soils, which can have complicated contaminant distributions, it is neither logical nor practical to maintain individual ranges of typical background concentrations for trace elements to be used in environmental protection policy. Perhaps this is the reason why in Australia single quality standards are used to determine where further investigations and assessment is required. But then it also implies that the regulators must be flexible in applying the guidelines and must be open to scientific argument in individual cases (Meuser and van de Graaff 2010).

In the Soil Protection Act of The Netherlands different quality standards have also been defined. They are called Target Values, which are focused on the potential risk to humans, and Intervention Values, which are focused on the potential risk to humans and ecosystems. Moreover, an Intermediate Value is introduced, this being the average between Target and Intervention Value. If the analysed pollutant concentration is lower than the Target Value, no measures are necessary. Minor restrictions have to be carried out in the case of results between Target and Intermediate Value, since the site is assessed to be slightly contaminated. If the concentration exceeds the Intermediate Value and falls below the Intervention Value, the Public Authority has to react by ordering further investigations or restrictions. Remediation has to be applied, if the concentrations exceed the Intervention Value, and hence the soil will be described as highly contaminated. The soil-related limits based on a Dutch standard soil are corrected by organic matter content and clay content in order to assess the availability in more detail.

All pathways, which are separate for children and adults, are included. In relation to the value definitions distinctions between non-threshold, carcinogen contaminants and threshold, non-carcinogen contaminants are made. For the latter ones the quality standards are based on the Acceptable Daily Intake (ADI). For the carcinogen contaminants the values are defined as the dose constituting an additional risk of a lethal tumour in 10,000 lifelong exposed persons.

In contrast to most of the soil protection regulations, in The Netherlands ecotoxicological risk limits are defined as a limit, if 50% of the ecosystem is affected. Because the extent of adverse effects varies between the different species the lowest value of terrestrial species was taken as the ecotoxicological risk limit. The defined values were based on a number of species exposed to pesticides as a function of surface area. In most cases, for assessment purposes the lowest value of the human toxicological and ecotoxicological limits, designated integrated value, will be taken. The Dutch quality standards are summarized in Table 7.22 (Swartjes 2003).

Precaution, Trigger and Action Values are defined in the Soil Protection Act of Germany (Tables 7.23 and 7.24). While the first consider mobility-determining soil

**Table 7.22** Quality standards (Target value, intervention value) of the Dutch Soil Protection Act for Soil (standardized soil with 10% organic matter and 25% clay) and deep groundwater >10 m (Data from Swartjes 2003)

| | Intervention values (mg kg$^{-1}$) | | | Target value (mg kg$^{-1}$) | Intervention value (µg L$^{-1}$) | Target value (µg L$^{-1}$) |
|---|---|---|---|---|---|---|
| | Soil | | | | Groundwater | |
| Substance | Risk to ecosystem | Risk to human life | Integrated value | | | Deep groundwater >10 m |
| Metals | | | | | | |
| Arsenic | 40 | 678 | 55 | 29 | 60 | 7.2 |
| Barium | 625 | 4,260 | 625 | 160 | 625 | 200 |
| Cadmium | 12 | 34.9 | 12 | 0.8 | 6 | 0.05 |
| Chromium III | 230 | 2,250 | 380 | 100 | 30 | 2.5 |
| Cobalt | 240 | 452 | 240 | 9 | 100 | 0.7 |
| Copper | 190 | 15,700 | 190 | 36 | 75 | 1.3 |
| Mercury (inorganic) | 10 | 197 | 10 | 0.3 | 0.3 (total) | 0.01 (total) |
| Mercury (organic) | – | – | – | – | | |
| Lead | 480 | 300 | 530 | 85 | 75 | 1.7 |
| Nickel | 210 | 6,580 | 210 | 35 | 75 | 2.1 |
| Zinc | 720 | 56,500 | 720 | 140 | 800 | 24 |
| Cyanides | | | | | | |
| Cyanides (free) | – | 16.8 | 20 | 1 | 1,500 | 5 |
| Cyanides (complex) | – | 4.36 | 50 (pH > 5) | 5 | 1,500 | 10 |
| Petroleum hydrocarbons | | | | | | |
| Total petroleum hydrocarbons | – | – | 5,000 | 50 | 600 | 50 |

(continued)

**Table 7.22** (continued)

| Substance | Intervention values ($mg\ kg^{-1}$) Soil: Risk to eco-system | Risk to human life | Integrated value | Target value ($mg\ kg^{-1}$) | Intervention value ($\mu g\ L^{-1}$) Groundwater | Target value ($\mu g\ L^{-1}$) Deep groundwater >10 m |
|---|---|---|---|---|---|---|
| Monoaromates | | | | | | |
| Benzene | 25 | 1.09 | 1 | 0.01 | 30 | 0.2 |
| Ethyl benzene | – | 50 | 50 | 0.03 | 150 | 4 |
| Toluene | 130 | 339 | 130 | 0.01 | 1,000 | 7 |
| Xylenes (sum) | – | 25.6 | 25 | 0.1 | 70 | 0.2 |
| Styrene | – | 249 | 100 | 0.3 | 300 | 6 |
| Cresoles (sum) | 50 | 117 | 5 | 0.05 | 200 | 0.2 |
| Phenols | | | | | | |
| Phenol (total) | 40 | 74.1 | 40 | 0.05 | 2,000 | 0.2 |
| Polycyclic aromatic hydrocarbons | | | | | | |
| Naphthalene | – | – | – | – | 70 | 0.01 |
| Benzo(a)pyrene | – | – | – | – | 0.05 | 0.0005 |
| PAH (total) (sum of 10) | – | 40 | 40 | – | – | – |
| Volatile chlorinated hydrocarbons | | | | | | |
| 1,2-dichloroethane | 60 | 3.86 | 4 | 0.02 | 400 | 7 |
| Dichloromethane | 60 | 18.9 | 10 | 0.4 | 1,000 | 0.01 |
| Vinyl chloride | 60 | 0.077 | 0.1 | 0.01 | 5 | 0.01 |
| Polychlorinated compounds | | | | | | |
| PCB (sum of 7) | 1 | – | 1 | 0.02 | 0.01 | 0.01 |
| PCDD/F (sum TE) | 0.046 | 0.001 | 0.001 | – | 0.000001 | – |
| Pesticides | | | | | | |
| DDT/DDD/DDE (sum) | 4 | – | 4 | – | 0.01 | 0.004 $ng\ L^{-1}$ |
| Aldrin | 0.35 | 13.8 | – | 0.06 | – | 0.009 $ng\ L^{-1}$ |
| Atrazine (triazines) | 6 | 21 | 6 | 0.0002 | 150 | 0.029 |
| Pentachlorophenol | 5 | 79.8 | – | 0.002 | 3 | 0.04 |
| Phtalates | | | | | | |
| Phtalates (sum) | 60 | – | 60 | 0.1 | 5 | 0.5 |

properties, trigger and the few Action Values are divided into different land-use type. They are defined as playgrounds, residential areas, parks and recreational facilities as well as industrial and commercial properties and consequently mainly focused on the urban land uses. The Trigger Values differentiate between the pathways soil-human, soil-plant and soil-groundwater, respectively. With reference to the pathway soil-plant uses such as agriculture, vegetable garden and grassland are points for consideration. It should be noted that distinct extraction means are provided for analysis purposes. The $NH_4NO_3$ method is recommended for exact

**Table 7.23** Quality standards (Trigger values for the pathways soil-human and soil-groundwater) of the soil protection ordinance of Germany (Data from BBODSCHV 1999)

| | Pathway soil-human (mg kg$^{-1}$) | | | | |
|---|---|---|---|---|---|
| Substance | Playground | Residential area | Park and recreational facility | Industrial and commercial property | Pathway soil-groundwater (µg L$^{-1}$) |
| Metals | | | | | |
| Arsenic | 25 | 50 | 125 | 140 | 10 |
| Cadmium | 10[a] | 20[a] | 50 | 60 | 5 |
| Chromium | 200 | 400 | 1,000 | 1,000 | 50 (total) |
| Lead | 200 | 400 | 1,000 | 2,000 | 25 |
| Mercury | 10 | 20 | 50 | 80 | 1 |
| Nickel | 70 | 140 | 350 | 900 | 50 |
| Cyanides | | | | | |
| Cyanides (total) | 50 | 50 | 50 | 100 | 50 |
| Polycyclic aromatic hydrocarbons | | | | | |
| Naphthalene | – | – | – | – | 2 |
| Benzo(a)pyrene | 2 | 4 | 10 | 12 | – |
| PAH (sum of 15) | – | – | – | – | 0.2 |
| Petroleum hydrocarbons | | | | | |
| Total petroleum hydrocarbons | – | – | – | – | 200 |
| Monoaromates | | | | | |
| Benzene | – | – | – | – | 1 |
| BTEX (sum) | – | – | – | – | 20 |
| Polychlorinated compounds | | | | | |
| PCB (sum of 6[b]) | 0.4 | 0.8 | 2 | 40 | 0.05 |
| Pesticides | | | | | |
| DDT | 40 | 80 | 200 | – | 0.1 |
| HCH-mix or β-HCH | 5 | 10 | 25 | 400 | – |
| Pentachlorophenol | 50 | 100 | 250 | 250 | – |

[a]In gardens where children play and food is simultaneously grown, 2.0 is applied
[b]Sum of 6, multiplied by a factor of 5

evaluation of the plant transfer. The handling of the assessment of the soil-plant pathway appears to be relatively complicated. Apart from food quality assessment, the growth impairment is taken into account for some metals well-known for their phytotoxicity. The soil-groundwater pathway does not differentiate between land-use types and it contains more parameters, as presented in Table 7.23.

Additionally, Precaution Values are listed, because the aspect of precautionary soil protection is of importance as well. These values integrate the soil texture, the pH value and the organic matter as availability-determining characteristics. Their aim is to prevent soil contamination in future. In accordance, soil contamination of untouched rural areas should only be polluted to a minor extent. The Public Authority, however, is allowed to neglect the Precaution Values in the case of naturally increased and large-area settlement-related regions where an enhanced

**Table 7.24** Quality standards (Trigger and action values for the pathway soil-plant) of the soil protection ordinance of Germany (Data from BBODSCHV 1999)

| | Food quality | | | Growth impairment |
|---|---|---|---|---|
| | Land-use type | | | |
| | Agriculture, vegetable garden | | Grassland | Agriculture |
| Substance | Trigger value | Action value | Action value | Trigger value |
| Metals | | | | |
| Arsenic | 200[a] | – | 50 | 0.4[b] |
| Cadmium | – | 0.04[b,c] | 20 | – |
| Copper | – | – | 1,300[d] | 1[b] |
| Lead | 0.1[b] | – | 1,200 | – |
| Mercury | 5 | – | 2 | – |
| Nickel | – | – | 1,900 | 1.5[b] |
| Thallium | 0.1[b] | – | 15 | |
| Zinc | – | – | – | 2[b] |
| Polycyclic aromatic hydrocarbons | | | | |
| Benzo(a)pyrene | 1 | – | – | – |
| Polychlorinated compounds | | | | |
| PCB (sum of 6) | – | – | 0.2 | – |

[a]In reductive soil conditions 50 mg $kg^{-1}$

[b]Extraction by $NH_4NO_3$, the other analyses are performed by aqua regia

[c]In the case of Cd-accumulating plants and wheat, otherwise the value is 0.1 mg $kg^{-1}$

[d]In the case of sheep grazing 200 mg $kg^{-1}$

background value can be expected in most cases. Therefore, particularly in urban areas the Precaution Values are not seriously applied (BBODSCHV 1999).

The definitions of the quality standards cannot absolutely relate to each other, but comparisons between the standards published in the different countries are striking because of their variability. For instance, the parameter cadmium varies enormously, as shown in Fig. 7.6. All presented values are related to topsoils of urban residential areas with gardens, indicating neutral pH values, if mentioned. Whilst in Australia a cadmium value of 20.0 mg $kg^{-1}$ is assumed to be acceptable, in China the soil is already assumed to be potentially contaminated, if the value exceeds 0.3 mg $kg^{-1}$. The variability does not refer to potentially mobile elements like cadmium, because the parameter lead transferable to plants to a small extent also ranges from 50 to 450 mg $kg^{-1}$ (Fig. 7.7). Critically, the question must be asked, whether the soil-plant pathway may really differ from country to country. The differences are also visible, even if the climatic conditions are principally comparable, as the values defined for Japan (150 mg $kg^{-1}$) and Korea (300 mg $kg^{-1}$) indicate. As observed for the parameter lead, in spite of the locally comparable climatic and geologic conditions the defined thresholds are different in Denmark (400 mg $kg^{-1}$) and The Netherlands (308 mg $kg^{-1}$).

The same problems can be discovered if one looks at the quality standards defined for the direct contact pathway. For example, in The Netherlands the Intervention Value for cadmium with reference to risk to human life is prescribed

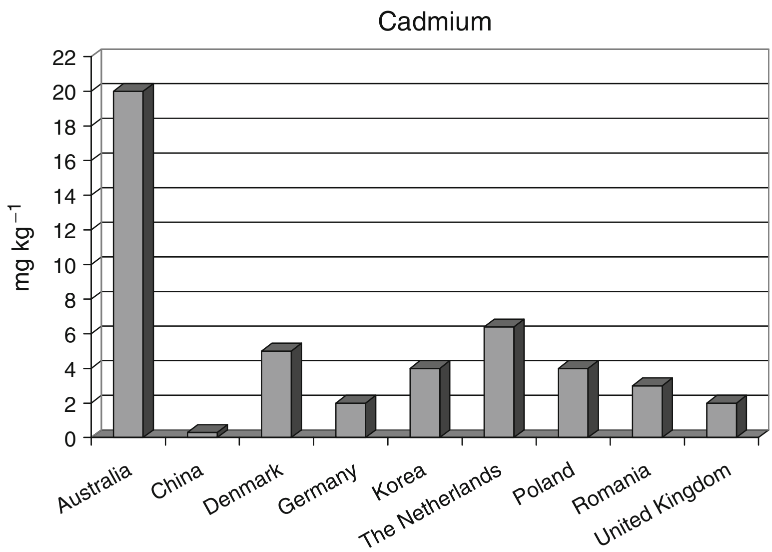



**Fig. 7.6** Comparison of cadmium quality standards for topsoils in residential areas with vegetable plots and in presence of neutral pH values in different countries. In Denmark the Cut-off Values and in The Netherlands the value between the Intermediate and the Intervention Value are chosen

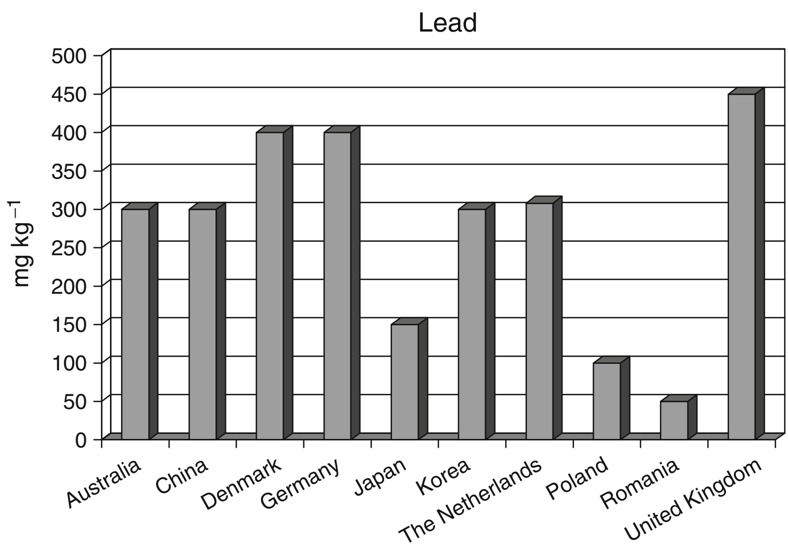



**Fig. 7.7** Comparison of lead quality standards for topsoils in residential areas with vegetable plots and in presence of neutral pH values in different countries. In Denmark the Cut-off Value and in The Netherlands the value between the Intermediate and the Intervention Value are chosen

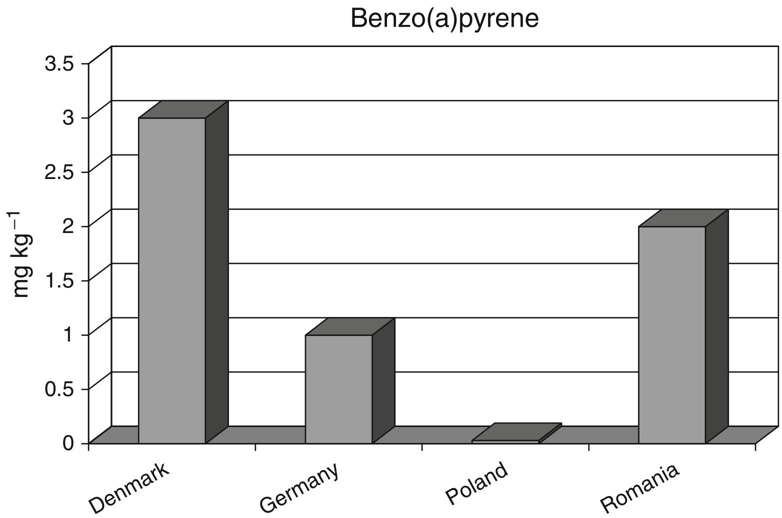



**Fig. 7.8** Comparison of benzo(a)pyrene quality standards for topsoils in residential areas with vegetable plots in different countries. In Denmark the Cut-off Value is chosen

by law to be a value of 34.9 mg kg$^{-1}$ and the integrated Intervention Value taking simultaneously the risk to ecosystems into account is prescribed to be 12 mg kg$^{-1}$. In Germany the cadmium quality standard is 10 mg kg$^{-1}$ for the highly sensitive land-use type playground, 20 mg kg$^{-1}$ for residential areas with vegetable gardens and 2.0 mg kg$^{-1}$ for sites where children stay and food is simultaneously grown, respectively. In other words, in residential areas where usually both the pathway soil-human and the pathway soil-plant are of relevance, acceptable cadmium concentrations vary between two neighbouring countries.

The quality standard comparison for organic pollutants like benzo(a)pyrene (Fig. 7.8) shows the general problem of missing values in a number of countries. Only four European countries can be illustrated and the variability mentioned above recurs. While in Poland the acceptable concentration touches the analysis detection limit, in Germany 1.0 mg kg$^{-1}$ and in Denmark 3.0 mg kg$^{-1}$ are classified as being permissible.

Generally speaking, the nationally prescribed quality standards may result from toxicological studies and scientifically correct research projects but they are man-made values which are obviously influenced by politically determined opinions and decisions.

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# Chapter 8
# Outlook

Many problems associated with contaminated urban soils are linked to the process of urbanization. The ongoing rural-to-urban migration of the population is causing a rapid expansion of cities, in particular in the developing countries. The result of this development is an uncontrolled urban sprawl and a number of additional problems such as inadequate waste disposal or drinking water supply. In developed countries shrinking cities on the one hand and booming regions on the other hand are causing increasingly tendencies, burdening the soil in different ways. The real reasons for most tendencies mentioned are social and economic ones and require political solutions. The impacts resulting from urban expansion, particularly the immense loss of agricultural land and the increasing conflicts with wildlife conservation, are related without doubt to the environmental compartment soil. Besides the question of sufficient land for food production which people are probably going to ask in future, the transformation of cropland and pasture into built-up areas irreversibly damages the soil. The destruction of former untouched areas, including wildlife habitats, also damages and interrupts natural soil functions. And ultimately, unplanned land use for landfill or mining heap purposes, for example, means a detrimental impact on the physical and chemical soil conditions. Accordingly, the process of land consumption and urbanization will involve both urban and rural land. Taking the agglomeration areas into account, this problem should not be reduced to the rural areas directly surrounding the cities. In other words, the urban soil business soil scientists are dealing with might occupy the thinking of soil scientists to a greater extent in the near future.

In the first instance, the contaminated site problem should receive more attention. In general, urban land shows a higher soil pollution level compared with the rural area, as the urban-rural-gradient indicates. Moreover, the depth gradient demonstrates the predominant reason for topsoil contamination, i.e. the deposition of airborne particulate matter. Especially in the proximity of industrial plots lacking appropriate filter techniques, the topsoil concentrations are drastically enhanced and correlations between emitting factories and adjacent topsoils are not difficult to prove. As a number of investigations have revealed, mining sites, in particular, which are often located in or around urban environments, cause an extremely high level of contamination. In urban land such as residential areas it is hard to define a clear division between industrial and traffic sources responsible for dust deposition.

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Both industrial emission and traffic sources, which impact the soil to a distance of up to 100 m if barriers such as sound-insulating walls and roadside green are absent, overlap with each other, accumulating considerably the topsoil contamination potential.

There is no clear correlation between the current land-use type and the soil contamination. However, with increasing age of the built-up area the contamination potential accelerates. For this reason, urban areas used for a long time appear to be important sites which threaten human health. The most important areas of concern, however, are industrial sites where emission occurs in a relatively uncontrolled manner, as observed to a great extent in less developed and developing countries. Therefore, the contaminated site problem can only be solved, if the source, namely air pollution, is strongly reduced.

It should be borne in mind that soil contamination acts more or less as the memory of human industrial history. As seen in relation to the lead accumulation of roadside soils, in spite of the availability of unleaded gasoline for several decades, this element has continuously accumulated in soils. Accordingly, the industrial dust deposition of today will cause soil problems in the future. Particularly with regard to the polluter traffic it is possible to eliminate some sources of soil contamination immediately, such as the use of bottom ash for road construction and slag for railway embankments as well as the application of de-icing salts in the northern hemisphere. The solution, however, must be combined with soil protective waste management (e.g. bottom ash, slag) or it must be connected with a search for alternative materials.

Considering the contamination problem in alluvial floodplains the significance of the soil as a sink or, metaphorically speaking, as a victim becomes even clearer. Discharges and surface water runoff from industrial sites accumulate in sediments and longitudinal transportation leads to dispersion in the horizontal and vertical direction. In some heavily industrialized areas the increase in contamination took place during the industrial period of former times but the dispersion additionally influenced by re-suspension occurs continuously over considerable distances for long periods. In order to reduce the contamination level the only appropriate way is the reduction of all kinds of discharges.

In urban environments areas can reveal enhanced contaminant concentration associated with an agricultural background. For instance, pesticides were found in gardens and allotments in spite of never having been used on site. Apart from the drift from adjacent agriculturally used areas, the reason can be the import of former agricultural topsoil which is deposited on urban land. Furthermore, application of sewage sludge, industrial wastewater and municipal garbage for fertilizing purposes or due to application in former times when the area was still agriculturally used, frequently caused topsoil contamination in urban landscapes. Thus, for a better understanding of urban soil contamination developments in the past should be taken into consideration.

There is no doubt that the most important causes of soil contamination in urban areas are related to abandoned industrial sites and man-made deposits. Many brownfields which look back on a long history of industrial use are highly contaminated. Missing inventory, risk assessment and particularly clean-up strategies led to long-term untouched territories with high contamination potential, but in some developing countries these sites are increasingly of importance because of the urban expansion mentioned above. Consequently, investigation, assessment, rehabilitation and clean-up are necessary steps for the realisation of future town planning which eliminates danger to human health and the environment.

In particular, deposits like fills and heaps usually consisting of polluted technogenic substrates should be taken into consideration in more detail. The exact study of different sorts of technogenic materials such as construction debris, slag, ash, mining and household waste and sludges might attract more interest, since it is possible to re-use some materials, substituting valuable raw materials. Otherwise, problematical materials thrown away or deposited without any cover or barrier system such as household garbage in less developed countries without favourable waste management as well as heavy metal mining slag in industrialized regions have to apply an adequate containment in order to avoid contaminant migration into environmental compartments like the groundwater. In relation to the technogenic materials one considerable problem might be that soil scientists have insufficient knowledge to identify and subsequently to assess the distinct materials deposited. It does not appear to be satisfactory only to name deposited soils fills and deposits without any differentiation of the materials, because it is well-known that the materials indicate different contaminant concentrations.

Up to now, registration and standardized investigation of contaminated land only occur satisfactorily in developed countries, whereas in less developed and developing countries these necessary tools are hardly developed or completely missing. In order to enable town planning without threatening the environment and particularly human beings the tools have to be initiated, taking predominantly the specific contamination of technogenic deposits into account.

In urban environments a very high percentage of anthropogenic soils are obviously present in the soilscape. Apart from mainly horticultural Anthrosols and Technosols, covering large areas in the cities, completely artificial soils such as rooftop gardens and planters, which are not connected to the original soil medium, can be found. The Technosols cannot be reduced to fills and deposits, man-made soils covering for instance sports field and footpaths are also constructed. The contamination level depends predominantly upon the material used and consequently varies enormously. Thus, more attention has to be paid to the construction material used in urban areas and, again, this question is linked to the management of waste products. In most countries registration, collection, treatment, recycling and decontamination of all sorts of waste materials such as household waste, ashes, mining waste, industrial slag and particularly construction debris are less developed, leading to an accelerated soil contamination problem.

Horticulturally used Anthrosols present in gardens and allotments are usually not deposited soils, but there are nevertheless many examples of soil contamination with respect to the handling of the soils. For instance, gardeners apply bottom ashes, sewage sludge and garbage in order to improve soil properties such as the nutrient capacity and the pH value. However, many materials contain hazardous substances, causing soil contamination after long-term application. Therefore, in green areas of the cities where originally near-nature soils were present, topsoil contamination can be expected irrespective of the particulate matter deposition and other reasons mentioned above. In developed countries the contamination took place mainly in the past when there was not much knowledge or awareness of the problematical substances. In less developed and developing countries the process continues because inhabitants recognize the necessity to fertilize the soil with problematical substances also in order to survive at all.

In built-up areas where long-term construction activities, soil excavation, backfill and levelling occurred the soil indicates heterogeneity related to physical and chemical properties and, in particular, the presence of anthropogenic artefacts. The soil handling and belowground structures lead to soil alteration and soil contamination that is not only reduced to the topsoil. In city centres it is a common situation that the anthropogenic soils show enhanced pollutant concentration to the depth of the parent material. Due to civil demolition procedures and results from warfare many urban soils contain construction debris which can considerably take part in soil contamination. Deep soil contamination is particularly connected with some types of anthropogenic deposits such as landfills and mining heaps. Due to the detrimental waste handling in less developed and developing countries uncountable wild dumpsites exist, causing enhanced and dangerous concentrations in soil, groundwater and soil vapour. Again, the initial position is related to the unfavourable waste management and less to the medium soil itself.

As already mentioned above, mining soils may exceed the contamination level of urban soils in a number of locations. Whilst coal mining soils mostly cause danger to the environment with respect to soil acidification, ore mining soils indicate extremely high values for heavy metals and processing agents. Plant toxicity and steep slope gradients are responsible for bare soils and subsequently wind drift into adjacent residential areas. Thus, many mining sites are so-called hot spots in the landscape where extreme soil contamination has been found. In future, these sites will require clean-up strategies to stop continuous migration of pollutants.

In general, it is supposed that innovative soil clean-up measures might become more significant. For instance, dredged harbour sediment fields requiring simultaneously drying and decontamination appear to be suitable for phytoremediation and it might be feasible to treat industrial sludge fields such as refinery sludge fields by bioremediation. While there will be an increasing tendency to apply the clean-up technology in developed countries, the main problem seems to be the cost-intensive application of innovative methods in countries with a more limited budget. Nevertheless, it should not be concealed that up to now it has not been possible to offer modern decontamination technology for a number of highly contaminated sites.

Apart from the contaminated site problem in urban environments many additional disadvantages impacting the soil have to be discussed. But they influence the contamination and especially the migration of contaminants considerably, and consequently they should be linked to the contaminated site problem. Soil sealing, which is well-known for its negative impact on, for example, the water cycle, can cover highly contaminated soils and protect environmental receptors from contaminant exposure. Although rightly assessed to be disadvantageous, soil sealing appears to be ambivalent. In contrast, both erosion and deflation cause accelerated contaminant dispersion, so that these physical factors might worsen the contaminated site problem in any case. Compaction frequently detected in urban areas influences water infiltration and percolation. Whether leachate is likely to be reduced or, on the contrary, mobilization is enhanced in reductive conditions are typical questions underlining the impact of physical soil conditions on contamination. Anthropogenically deposited soils might alter physical conditions, taking for instance the intra-pore system of technogenic substrates into consideration, which significantly improves the available water capacity. These effects as well as other physical processes such as groundwater table lowering and subsidence in mining areas are of importance for different points of consideration such as the water supply for the vegetation on the one hand and the solubility of contaminants on the other hand. Consequently, physical properties and, particularly their alterations, should be taken into account with respect to the investigation of soil contamination.

The context becomes still more importance in relation to the chemical properties. Every soil scientist knows that the mobility of heavy metals depends on chemical characteristics such as pH value, texture and content of humus and oxides. Moreover, it is known that the chemical behaviour of organic pollutants is strongly influenced by chemical and biological properties such as humus content and biological activity. But in urban soils a lot of relationships are different from natural soils. Therefore, the distinct analytical methods associated with the expected mobile fractions of contaminants (e.g. ethylenediaminetetraacetic acid, HCl, Na acetate, etc.) cannot always confirm the results normally found in natural soils. For this reason, it makes sense to have a look at special constituents present in deposited soils such as technogenic carbon and magnetic iron oxides. Obviously, contaminant behaviour is influenced by these compounds and generally by the presence of technogenic substrates. For instance, coal mining heaps stemming from the Carbonaceous Age cause increasingly accelerated metal mobility in the course of time due to acidification, while in deposited soils consisting of alkaline construction rubble the inverse process occurs, resulting from immobile carbonatic binding forms of some elements. However, a lot of relationships are not clear if one takes the different binding forms of heavy metals or the different biodegradation sequences of organic pollutants into account. Accordingly, further research appears to be necessary to assess satisfactorily the mobility of contaminants in urban soils.

In particular, pedogenesis visible in anthropogenic soils as well as in natural soils might continuously impact the migration of contaminants. One must be aware of anthropogenic soils not being designated as stagnant systems. Soils currently indicating high aqua regia extractable concentration but a low metal mobility can become problematical and consequently they can cause threats to human health and the environment in future.

The assessment of urban soils is either based on the precautionary soil protection idea or it results from the application of quality standards for contaminants. For the first tools of a functional soil assessment are developed with the aim of protecting soils from sealing, excavation, construction, etc. The methods are often combined with the protection of other environmental compartments such as surface water and vegetation. Accordingly, landscape architects and ecologists should work hand in hand with soil scientists to prevent ongoing land consumption and degradation. Furthermore, cooperation is also important with reference to soil clean-up and soil rehabilitation. For instance, a brownfield which has been untouched for a long time and which exhibits rare plant and animal species requires a plant inventory as well as soil investigation. However, attention should be paid to possible arguments between ecologists and soil scientists, because highly contaminated soils with rare vegetation (e.g. ore mining heaps) may require both soil rehabilitation (combined with soil excavation or covering and consequently removal of the vegetation) and plant protection that excludes soil movement.

Quality standards listed in several acts and regulations in different countries and to some extent differently defined are mainly aimed at the protection of human health. The values resulting from exposure scenarios and oriented to distinct pathways such as soil-human and soil-plant are applied differently. In most countries authorities neglect quality standards, in some countries they are defined but not examined and in a few, particularly Western European countries, they are considered, in principle, in town planning processes. In general, the application of quality standards for urban soil assessment purposes appears to be necessary in order to exclude threats to humans and the environment. Critically, it is hard to understand that children from country A are allowed to take in more lead, for instance, than children born in country B. Differences in climate and geology should surely be taken into account in the definition of standard values but a general standardization would improve the acceptance of toxicologically determined values.

In summary, dealing with contaminated urban soils is not a new research field, since intensive worldwide studies of the urban environment have already yielded a lot of knowledge. Urban soils are decisively different from their natural counterparts in relation to many physical and chemical properties and a lot of information about natural soils cannot simply be transferred to urban soils. It cannot be ignored that dealing with contaminated urban soils will become increasingly important. The world knows that the soil problems due to the process of urbanization and due to the results of contamination must be solved in developed, developing and less developed countries in order to ensure satisfactory living conditions for human beings in the future.

# Appendix

| | | |
|---|---|---|
| 1. | - s. organic, inflammable or smouldering at low temperature | 2. |
| | - s. different | 3. |
| 2. | - s. smelling of wood or of rotting material, usually splintery | Wood |
| | - s. dark brown, wooden, hardened | Lignite coal |
| 3. | - s. sharp-edged and transparent or milky | Glass |
| | - s. with glassy surface and with intra-pores or dense | Ceramics |
| | - s. different | 4. |
| 4. | - s. elastic, breakable or flexible and inflammable or smouldering at low temperature, usually coloured | Plastic |
| | - s. different | 5. |
| 5. | - main colour white | 6. |
| | - main colour different | 7. |
| 6. | - s. fibrous and exothermal reaction after HCl addition | Blast furnace wool |
| | - s. >2 mm, rough surface, slowly soluble in water and with white or yellowish white colour | Plaster |
| | - s. <2 mm, single grains filled in dry conditions, quickly soluble in water, salty, with white to pink colour | Salt mining waste |
| 7. | - main colour yellow or yellowish brown | 8. |
| | - main colour different | 11. |
| 8. | - s. silty, sometimes with a lumpy structure | 9. |
| | - s. sandy or sharp-edged gravel | 10. |
| 9. | - s. yellow, after de-watering dense, with smooth edges | Plaster |
| | - s. yellowish brown, grey single particles possible, lumpy in wet conditions | Ash (domestic fuel) |
| 10. | - texture sand to sharp-edged gravel, single grains glassy and rough-edged, carbonate content moderate to high, $H_2S$ odour after HCl addition | Blast furnace sand |

(continued)

(continued)

| | | |
|---|---|---|
| | - texture sand, high percentage of single grains with a glassy and metallic sheen, carbonate content missing or very low | Sand from foundry |
| 11. | - main colour grey, no recognizable Fe oxides | 12. |
| | - main colour different | 24. |
| 12. | - s. silty to sandy (fine sand) | 13. |
| | - s. gravel, sharp-edged gravel and/or stones | 14. |
| 13. | - carbonate content high to extremely high, colour light grey | Fly ash (garbage incinerator) |
| | - carbonate content missing to moderate, colour (dark) grey | Fly ash (hard coal-fired) |
| | - carbonate content low to moderate, colour (dark) brownish | Fly ash (lignite coal-grey fired) |
| 14. | - s. mainly with slate/shale structure | 15. |
| | - s. with no slate/shale structure | 16. |
| 15. | - s. light grey, texture always <200 mm | Coal mining waste (after washing) |
| | - s. dark grey, weathered clay or silt stone | Coal mining waste (heap waste) |
| 16. | - s. conglomeratic, cement-like matrix | 17. |
| | - s. conglomeratic, slag structure | 18. |
| | - s. not conglomeratic | 19. |
| 17. | - single grains <7 mm, rough surface, colour usually light grey | Mortar |
| | - single grains >7 mm, dense surface, colour grey to greyish blue, sometimes with steel fragments | Concrete, steel concrete |
| 18. | - colour grey, coloured inclusions | Steelworks slag |
| | - colour grey to greyish brown | Chromium slag |
| 19. | - s. porous and with specific structure (spherical, puffed up) | 20. |
| | - s. without spherical/puffed up structure | 23. |
| 20. | - s. with metal and glass percentage | Bottom ash (garbage incinerator) |
| | - s. without metal and glass percentage, colour grey to greyish brown | 21. |
| 21. | - carbonate content missing to very low, texture <35 mm | Bottom ash (hard coal-fired) |
| | - carbonate content missing to very low, texture >35 mm | Dross |
| | - carbonate content at least moderate | 22. |
| 22. | - colour brownish grey, texture indifferent | Bottom ash (lignite coal-fired) |
| | - s. partly with reddish and yellowish inclusions, rough surface, texture usually >35 mm | Zinc slag |
| 23. | - s. very porous (honeycomb-like), pores >4 mm, s. very light | Blast furnace pumice |

(continued)

(continued)

| | | |
|---|---|---|
| | - s. very porous, pores usually <4 mm, colour grey | Blast furnace slag |
| | - s. hardly porous, very heavy, colour grey to greyish blue | Steelworks slag |
| 24. | - s. with rusty surface, percentages with metallic sheen possible | Metal |
| | - main colour grey, Fe oxides visible | 25. |
| | - main colour different | 26. |
| 25. | - s. not porous, partly with shale structure, rusty inclusions possible | Coal gangue (heap waste) |
| | - s. porous, partly with oxidized surface, metal fragments inside | Blast furnace slag (containing Fe) |
| | - s. conglomeratic, porous single grains, rusty spots at the surface, texture <45 mm | Slag (steelworks dust furnace) |
| 26. | - main colour red | 27. |
| | - main colour different | 28. |
| 27. | - s. with shale structure, grey inclusions possible, sometimes granulated | Coal mining waste (heated waste) |
| | - s. without shale structure, few pores, colour red to reddish yellow, carbonate content missing | Brick |
| | - s. without shale structure and pores, colour dark red, sometimes granulated | Copper slag |
| 28. | - main colour greyish blue, conglomeratic, dense | Concrete (derived from fly ash) |
| | - main colour dark blue to black, s. very heavy, few pores | Lead slag |
| | - main colour dark brown, greyish black or black | 29. |
| 29. | - s. with naphthalene odour | Tar asphalt |
| | - s. without naphthalene odour | 30. |
| 30. | - main colour very dark brown | 31. |
| | - main colour dark grey to black | 32. |
| 31. | - s. fibrous, dense | Lignite coal |
| | - s. sandy, partly glassy and single grains with metallic sheen | Sand from foundry |
| 32. | - s. conglomeratic, surface often shining, colour black, no naphthalene odor | Bitumen asphalt |
| | - s. different | 33. |
| 33. | - s. with shining surface or glassy, predominantly black colour | 34. |
| | - s. different | 35. |
| 34. | - texture <8 mm, granulated, no shale structure | Bottom ash (smelting furnace) |
| | - texture >8 mm, glassy and shining surface, pores possible, no shale structure | Slag (foundry) |
| 35. | - s. with shale structure | 36. |
| | - s. conglomeratic and/or porous | 37. |

(continued)

(continued)

| | | |
|---|---|---|
| 36. | - colour dark grey to black, high weathering influence | Coal mining waste (with coal remnants) |
| | - s. usually causing dirty fingers, hardly shining, texture dust, sharp-edged gravel or stone, colour black | Coal products |
| 37. | - s. conglomeratic, with porous single grains, often rusty spots at the surface, texture <45 mm | Slag (steelworks dust furnace) |
| | - s. surface porous (small pores) | 38. |
| 38. | - s. with fine pores, light, silvery shining surface possible, colour dark grey to black | Coke |
| | - s. with fine pores, light, in combination with bottom ash (coal-fired power station) | Ash–coke mixture |

s. = substrate

# Index

**O**



**P**