Chapter 1
Introduction to Contaminated Site Management

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Abstract Over thousands of years, contaminants have been added to the world's upper soil layers and have led to contamination of the soil and the groundwater. However, it was not until the late 1970s that several notorious cases of contaminated sites led to a sudden awareness to the general public. Today, in most developed countries, the number of potentially contaminated sites has grown to six or seven digits. This chapter describes the basic principles of contaminated site management. It focuses on risks and Risk Assessment, that is, quantifying the risks from contaminated sites on the basis of chance (exposure) and effects. This process is widely accepted today as offering the best balance between a sound scientific basis and practical implementation for appraisal of contaminated sites. Moreover, this chapter describes Risk Management, this is the process that brings contaminated sites back into beneficial use. The four major protection targets are human health, the soil ecosystem, the groundwater and Food Safety. Specific attention will be given in this chapter to a wide variety of topics including public and political awareness, soils, local and diffuse contaminated sites, contaminants, background concentrations, emissions to soil, site characterisation, land use, Soil Quality Standards, Brownfields, cost-benefit analyses, Risk Perception and Risk Communication, sustainability, and the actors involved in contaminated site management. Finally, several approaches to contaminated site assessment and management will be described, including the Fitness-for-Use approach, and Risk-based Land Management. In doing so, specific attention will be given to practical aspects such as effective use of financial resources and integration of contaminated site management (e.g., with regard to spatial planning, socio-cultural issues, economics and other factors).

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1.1 Status of Contaminated Sites

1.1.1 History

1.1.1.1 Early Soil Contamination

Over thousands of years, since humans started mining for the iron-containing mineral hematite and later for malachite for copper production, potentially harmful chemical compounds (*contaminants*) have been added to the upper soil layers. And for centuries humans have dumped their waste materials into primitive waste dumps. However, large scale mining and, hence, large scale soil contamination, only came into existence in Europe, the USA and many other parts of the world in the 19th century. One phenomenon that sped up soil contamination was the Industrial Revolution which began in England and subsequently spread to several developed countries in Europe, the USA and Japan, from the turn of the 18th and 19th centuries. But the impact of the Industrial Revolution on contaminated sites was minor compared to the impact of the technological developments that took place mainly during the 20th century. These developments were characterized by a more than proportional increase in emissions of contaminants into the environment. Soil can often be considered as the ultimate sink for contaminants that enter the environment. As a consequence, emissions of contaminants to soil increased, for example, through the large-scale use of fertilizers, expansion of industrial production, the use of fossil fuels and, as an overall impact factor, a huge increase in population growth. It was not only the bulk rate of production of contaminants that significantly expanded. It was also the enormous increase in variety of types of chemical compounds that were produced for public or industrial use, or produced as a by-product, and eventually entered the environment and the soil.

The early examples of contaminated sites mainly resulted from delocalisation of contaminants, that is, metals in metal ores and crude oil from deeper soil layers to the upper soil layer. In the 20th century, however, an enormous variety of organic compounds, along with metal organic complexes, were produced out of existing less harmful compounds. Moreover, the soil was often intentionally used as a sink, for example, by primitive land filling or the release of contaminant holding fluids into the soil. Up to the 1970s, it was often believed that, very much like dumping waste water in a kitchen sink, superfluous materials simply disappeared towards some unknown destination where it was out of sight and, probably, would do no harm. A slightly more sophisticated approach towards the dumping of contaminant-holding materials was based on the belief that the soil-groundwater system was able 'to handle the burden', either by incorporating the contaminants in some kind of physical, chemical or biological cycling process, or simply by dilution. Although this latter approach included some arguments that we use in modern Risk Management procedures in regard to contaminated site management today, the power of the soil to 'clean' itself was far from being able to counterbalance the increasing contaminant load. Given the cost ratio between prevention measures and soil remediation, these approaches of dealing with contaminants must, in retrospect, be classified as immensely naive.

1.1.1.2 Public and Political Awareness

In the early 1970s, some soil protection-related policies came into existence in several countries. However, it was not until the late 1970s that several notorious cases of contaminated sites led to a sudden awareness among the general public and served as a loud alarm to decision makers. Those cases where contractors, generally without any bad intentions, had created situations in which humans came in close contact with notorious (carcinogenic!) contaminants in soil could especially count on intensive media attention.

In 1978 the *Love Canal disaster* became a national media event in the USA (Levine 1982). At the site of Love Canal, a neighbourhood near Niagara Falls in upper New York State, USA, a school and a number of residences had been built on a former landfill for chemical waste disposal, and thus sat directly on the dump site of thousands of tons of dangerous chemical wastes. The US Environmental Protection Agency (US EPA) discovered and reported on a disturbingly high rate of health afflictions for the residents, such as miscarriages and birth defects (Beck 1979). Although it was difficult to conclusively prove that the contaminants in the soil were the cause, recurring illnesses of the inhabitants and school employees were connected with the history of the site. In 1980, a state of emergency was declared and 700 families were evacuated.

In Europe in 1979, the site of *Lekkerkerk* in the Netherlands became an infamous national event. Again, a residential area had been built on a former waste dump which included chemical waste from the painting industry. Moreover, to prepare the site for the construction of a residential area, channels and ditches had been filled in with chemical waste-containing materials. The scandal started after a water pipe exploded because of the presence of aromatic contaminants in the soil. The specific known fact that benzene, a carcinogenic agent, was involved raised public concern. Nearly 300 families were evacuated, 1600 barrels of chemical waste were removed, and the soil under the residences was excavated.

Today, the Love Canal and Lekkerkerk cases still are often mentioned in introductions to reports on contaminated sites and in oral platform presentations at contaminated site management-related congresses.

1.1.2 The Present Situation

1.1.2.1 Extent of Soil Contamination

In the last two decades of the 20th century, the number of potentially contaminated sites grew in most developed countries to six or seven digits. During this period, most developed countries established monitoring systems for the purpose of assessing the extent of their contaminated sites. According to the European Environmental Agency (EEA) the number of contaminated sites requiring remediation in the EU member states was approximately 250,000 in 2007 (European Environmental Agency 2007). Today, it is expected that this number has grown significantly. According to the same source, potentially contaminating activities are estimated to have occurred at nearly 3 million sites (including the 250,000 sites already mentioned). In the European Union, 3.5 million sites are contaminated, affecting 231 million people and representing a market value of 57 billion Euros (Commission of the European Communities 2006). Soil contamination is one of the eight threats mentioned in the *EU Thematic Soil Strategy* (Commission of the European Communities 2006).

A contaminated soil map would roughly coincide with an anthropogenic map, since humans are generally recognized as the main polluters. Most of the contaminated sites are found in or close to cities.

In the present day, most countries have become aware of the huge practical, social and financial impact of contaminated sites.

1.1.2.2 Emissions to Soil

Emissions into the environment might occur through the air (atmospheric deposition) or directly (conscious or subconscious disposal). Examples of possible air emissions are:

- the deposition of Polycyclic Aromatic Hydrocarbons (PAHs) due to heating processes, often emitted tens or hundreds of kilometres away from the source (e.g., Ollivon et al. (2002), who measured substantial amounts of PAHs in atmospheric

fallout (precipitation, gas phase and particulate matter), especially in winter, at an urban site in Paris, France);
- metal deposition from lead smelters (e.g., Salemaa et al. (2004), who measured elevated concentrations of metals in different plants, especially in bryophytes, near a copper-nickel smelter in Harjavalta, Finland);
- traffic (e.g., Hjortenkrans et al. (2006), who measured elevated concentrations of copper and antinomy due to decelerating activities, and lead and cadmium due to the combustion of petrol, in top soils in the south of Sweden);
- incineration activities (e.g., Schuhmacher et al. (2000), who measured elevated concentrations of Poly Chlorinated Dibenzo-p-Dioxins (PCDD) and Poly Chlorinated Dibenzo Furans (PCDF) in soil and vegetation in the vicinity of an old municipal solid-waste incinerator in Barcelona, Spain).

Metals and PAHs are known to be the most abundant and widespread contaminants worldwide.

Several other major contaminant sources are known to have contributed to large-scale soil contamination, such as coal combustion and mining activities. Another notorious source of contaminants, such as all kinds of petroleum hydrocarbons, PAHs, BETX (benzene, ethylbenzene, toluene and xylenes), methyl-*tert*-butylether (MTBE), and metals, is the oil industry, through oil exploration and production, refining and petro-chemical activities. In Mexico, for example, the number of reported hydrocarbon spills for the year 2000 exceeded 185 thousand, equivalent to 6252 tons (Iturbe et al. 2005).

In agricultural areas, contaminants have been introduced by using soils for waste-water filtering, applying sludge onto the soil, or applying ash from waste materials, used for liming (e.g., Pasquini and Alexander (2004), who demonstrated an increase of mainly lead through ash addition to soils on the Jos Plateau in Nigeria). Also the application of mine waste contributed to soil and groundwater contamination (e.g., Cobb et al. (2000), who demonstrated the presence of relatively high metal uptake of lettuce and radishes in soils mixed with mine waste in Bingham Creek, USA).

There also are many types of direct emissions. These direct emissions could follow from unintentional leakage (leaching) from artificial layers applied for soil surface-raising purposes, filling materials in construction works (dikes, roads) or from old waste disposal landfills, gas stations or storage tanks. Another example of an activity that generally caused contamination of soil and groundwater are gas works. See Fig. 1.1 for a picture of the former gas works 'Delftse Wallen' in Zoetermeer, the Netherlands, as an example, around 1908. At this site, heavy metals and PAHs were found in the upper soil layer, due to soil surface-raising activities, and aromatic contaminants, petroleum hydrocarbons and PAHs in the groundwater, in the late 20th century. Currently, a soil remediation is ongoing, the latest cost estimate is 2 million euro.

Another type of direct emissions to soil is spilling of contaminants during production processes, transport and storage. This often relates to (petro)chemical industrial sites (e.g., Nadal et al. (2004) who measured elevated concentrations

Fig. 1.1 The former gas works 'Delftse Wallen' in Zoetermeer, the Netherlands, around 1908, as an example of an activity that often caused contamination of soil and groundwater (photo: Historical Society 'Oud Soetermeer'; reproduced with permission)

of arsenic, chromium and vanadium around Tarragona, Spain, an area with an important number of petrochemical industries).

One specific kind of contaminant sources are activities at military training facilities. It generally includes a wide scale of polluting activities that might lead to human health and ecological risks (e.g., Teaf (1995), who dealt with human health and ecological risks at former military sites in the former Soviet Union). Another specific type of soil contamination arises from shooting ranges where lead bullets are deposited into the soils. In the state of Oregon, USA, for example, 211 active firing ranges exist (Darling and Thomas 2003). Soils in clay pigeon shooting ranges can also be seriously contaminated by heavy metals such as lead, antimony, nickel, zinc, manganese and cupper (Migliorini et al. 2004).

In some cases human or technical failure causes soil contamination. An example of this is a series of spilled mine tailing accidents. Since 1970, there have been 35 major mine-tailing dam failures reported (Macklin et al. 2003). One example of these is the collapse of a tailing dam in the Chenzhou lead-zinc mine in China in 1985, which led to the spread of huge amounts of mining waste spills onto farmland, followed by an emergency remediation procedure (Liu et al. 2005).

According to the European Environmental Agency, the most important sources for soil contamination in Europe, as an example, are industrial production and commercial service (41%), municipal waste treatment and disposal (15%), the oil industry (14%), industrial waste treatment and disposal (7.3%), storage (5.4%), power plants (3.9%), transport spills on land (2.1%), mining (1.4%), military (0.9%) and others (8.2%) (European Environmental Agency 2007).

A substantial part of the existing contaminated sites in developed countries are a legacy from the past. Today, however, it is widely recognized that the consequences of soil contamination were badly underestimated in the past. As a consequence, *prevention* of emissions into soil is a mandated practice in practically all developed countries today, for ethical, practical and financial reasons. Therefore, all kinds of prevention measures are being taken. Production processes, transport and storage follow strict regulations in order to avoid or minimize spilling. Gas stations have been equipped with liquid impermeable foundations. And waste gases have been filtered before leaving chimneys to avoid or minimize the exhaust of contaminants. And agricultural practices, including the processes that result in emissions into soil, are also strongly regulated today. An interesting tendency is the trend towards sustainable agriculture where the inputs into the soil are in equilibrium with the outputs.

1.1.3 Public Awareness

Out of the three major environmental compartments, that is, soil, water and air, *soil* probably is the least known and the least appreciated. Water, in particular surface water, is a highly visible and widely appreciated part of the landscape. The general public despises contaminated waters, for ethical and practical reasons. Since we are directly surrounded by it, the air compartment is as much appreciated as feared for. We need the air every minute of the day to survive and are alert for any disturbance in air quality. Moreover, we are very much aware of bad air quality, since we cannot help but see and smell it. Soil, on the contrary, is hardly visible. Except from an agricultural viewpoint, humans generally do not have a positive association with soil. To the general public, soil is often thought of as a dark place in which creepy organisms reside and in which we bury our dead.

Individuals who generally have a positive association with soil are people who grow crops, and realise the meaning of the soil in terms of habitat and nutrient source. These individuals can either be farmers who grow crops for commercial reasons, or individuals who grow crops for their own consumption (see Fig. 1.2, in which small allotments are shown in Jiangsu province, China, where the local population uses the spare bare surfaces for their food supply). An interesting initiative to make soils more palpable to the general public relates to the relationship between soils and art (Wessolek 2006). This movement reveals the beauty of soil profiles and of artworks that use soil materials or soil visions.

In the last few years the unfavourable view of soil has changed a little in Europe, the USA, Australia and Canada. Many people are now aware of the huge pressure that humans put on the environment, including the soil. Moreover, many environmentally conscious people read articles in the newspaper about the amazing performances of soils in, for example, organic biological agriculture practices. And since global warming is at the top of the political priority list, soils are recognized as a powerful CO_2 sink.

Not only scientists and regulators, but also the general public are all aware of the presence and consequences of soil contamination today. This awareness is still partly based on negative events in the 1970s and 1980s. Although the general public has good reason to be worried when their direct living conditions are impacted by

Fig. 1.2 Small allotments in Jiangsu province, China, used by the local population for their food supply (photo: F. Swartjes)

contaminants, this negative approach is sometimes exaggerated. There were several reasons for negative sentiments associated with those events of the 1970s and 1980s. First, humans were confronted with an unknown threat. Today we understand contaminated soils much better, and it is widely recognized that clear and objective information about a specific case, and the risks involved, is essential. Second, the citizens concerned felt betrayed by the government. When they bought real estate, no information was provided about the health risks they might be confronted with. In most countries today the transfer of property is accompanied by detailed information, often supported by computer systems that show the actual soil quality. In reality, in both the Love Canal and the Lekkerkerk cases, the government was as much overwhelmed by the phenomenon of soil contamination as were the citizens.

Following the Love Canal scandal, a clay lining was used to prevent further leaching of contaminants and a dirt cover was provided to prevent contact between the contaminants and humans. Unfortunately, these protective layers were damaged during construction work as a result of underestimating the threats involved. It is unlikely, however, that such technical mistakes, made in the early days of Risk Management, are still being made today.

1.1.4 The Contaminated Site Management Framework

1.1.4.1 Schematization

Several different frameworks for contaminated site management on the basis of risks are available. In Fig. 1.3 a schematization of the contaminated site management framework that is followed in this book, roughly in accordance with the highly influential report on Risk Assessment by the US National Research Council (US National Research Council 1983), has been illustrated in a simple graph.

The first step in this contaminated site management framework is *problem definition* (aka: *issue framing*). The second step is *Risk Assessment*, which

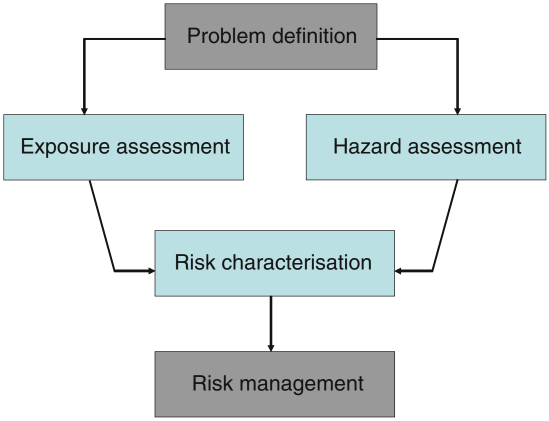

Fig. 1.3 The contaminated site management framework, as followed in this book (roughly in accordance with US National Research Council 1983). The *light-shaded boxes* relate to Risk Assessment

includes two different activities (see Fig. 1.3; the light-shaded boxes relate to Risk Assessment). These are the *Exposure Assessment* (aka: dose assessment) and the *Hazard Assessment* (aka: effect assessment), mainly used in Human Health Risk Assessment. The combination of the Exposure Assessment and the Hazard Assessment is called the *Risk Characterisation*, which results in an appraisal of the contaminated site. An alternative contaminated site management framework as used in the UK, including Risk Assessment, Risk Management but also economic and social issues, is given in Pollard et al. (2002a).

In several publications the phrase 'Risk Management' is used for the whole contaminated site management framework as shown in Fig. 1.3. In this book, however, *Risk Management* is the next logical step in the contaminated site management framework, following Risk Characterisation, in cases where this risk appraisal demonstrates the need for intervention, usually since the risk for a specific protection target is unacceptable. In this stage, solutions are sought for the purpose of bringing contaminated sites back into beneficial use, and are generally focused on risk reduction.

1.1.4.2 Problem Definition

The first step in a contaminated site management project is *Problem definition* or *issue framing*. In this step, the scope of the project needs to be clearly described. Moreover, the protection targets need to be defined. It is also very important to define the relevant time frame, since factors that impact risks will change over time.

Since regulators often have a profound impact on the initiation and performance of contaminated site management, it seems logical that they formulate the exact purpose of the contaminated site management project. Regulators also are responsible for defining the boundary conditions, for example, the required precaution/conservatism of the Risk Assessment for the given site. Therefore, intensive communication between regulators and scientists, and in fact between all stakeholders involved, must take place in the early stages of a contaminated site project.

1.1.4.3 Protection Targets

With regard to the definition of risk, which is a concept that denotes a potential negative impact on an asset, a relevant issue is determination of the nature and value of the asset. Since these assets are potentially negatively impacted, they need to be protected. Therefore, the term *protection targets* (aka: *receptors*) is often used.

With regard to contaminated sites, several protection targets have been recognised. The principle protection target with regard to contaminated sites, worldwide, is *human health*. More specifically, it is the physical health condition, not usually the mental health condition, of human beings that is considered. Alternatively, several risk-based quality assessment procedures use terms such as 'humans', 'human beings', or 'man' as protection targets, but they all refer to human health, that is, the state of physical health of human beings. There has, for decades, been an intensive debate on the extent of human health effects from contaminants in soil and groundwater. Although often overestimated, many studies have provided solid evidence that these effects are real. Beard and Australian Rural Health Research Collaboration (2005), for example, concluded that there is suggestive evidence for a role of exposure to DDT and DDE from soils with regard to pancreatic cancer, neuropsychological dysfunction, and reproductive outcomes. The relevant process with regard to the determination and often evaluation of risks is *Human Health Risk Assessment* (*HHRA*). Swartjes and Cornelis (Chapter 5 of this book) give a detailed overview of Human Health Risk Assessment. The subsequent chapters of this book (see Chapter 6 by Bierkens et al., Chapter 7 by Cave et al., Chapter 8 by Mclaughlin et al., Chapter 9 by Trapp and Legind, Chapter 10 by McAlary et al., Chapter 11 by Elert et al., this book) give details on the determination of Exposure Assessment, a crucial process in Human Health Risk Assessment. Langley (Chapter 12 of this book) describes the Hazard Assessment, another indispensable element in Human Health Risk Assessment.

A second protection target is the *soil ecosystem* (or Ecosystem health). Although not always appreciated to the extent it deserves, the soil ecosystem performs some immensely important tasks for humans (*Ecosystem Services*). Moreover, protecting the various species in soil contributes to the maintenance of Biodiversity. Only a few countries formally consider the soil ecosystem to be an protection target. Over the last few years, however, political and scientific interest in protection of the soil ecosystem has gained in importance, at least in Europe (Carlon and Swartjes 2007a). An enormous number of investigations have shown the adverse ecological effects of contaminants in soil. Nagy et al. (2004), for example, demonstrated the adverse effects of metals on nematodes in Hungarian soils. The relevant process with regard to the determination and often evaluation of risks is *Ecological Risk Assessment* (*ERA*). Swartjes et al. (Chapter 13 of this book) give an overview of Ecological Risk Assessment. The subsequent chapters of this book, these are Posthuma and Suter (Chapter 14 of this book) and Rutgers and Jensen (Chapter 15 of this book), describe Ecological Risk Assessment in more detail, from a generic and a site-specific perspective, respectively.

A third protection target is the *groundwater*. Juhler and Felding (2004), for example, demonstrated the presence of many organic contaminants in groundwater, including toluene, phenol, xylene, trichloromethane, benzene, dibuthylphthalate, 2,4-dichlorophenol, trichloromethane and pentachlorophenol, mainly originating from the upper soil layers, in 7671 groundwater samples collected from 1115 screens from the Danish National Groundwater Monitoring Program. Groundwater as a protection target has a special status, since groundwater is part of the soil as defined in the framework of this book. Moreover, the groundwater is both a protection target and a means of transport (*pathway*) for contaminant migration. The relevant process with regard to the determination and often evaluation of risks is called *Groundwater-related Risk Assessment* in this book. Swartjes and Grima (Chapter 17 of this book) give an overview of Groundwater-related Risk Assessment, considering groundwater both as a protection target and as a means of transport (pathway) of contaminants. The subsequent chapters (see Chapter 18 by Mallants et al., this book; and Chapter 19 by Rolle et al., this book) describe Groundwater-related Risk Assessment in more detail, that is, leaching of contaminants from soil into the groundwater and transport within the groundwater, respectively.

Finally, another important protection target is *Food Safety*. Generally speaking, this includes two different types of protection targets, namely, crops and animal products (meat, milk and eggs). An example is found in Yang et al. (2004), who evaluated the uptake of lead from soil into rice and meat, around a lead/zinc mine in Lechang, Guangdong Province, in China. The relevant process with regard to the determination and often evaluation of risks is called *Food Safety-related Risk Assessment* in this book. Specific elements with regard to Food Safety-related Risk Assessment, in particular to consumption of vegetables, are included in Mclaughlin et al. (Chapter 8 of this book), Trapp and Legind (Chapter 9 of this book) and Elert et al. (Chapter 11 of this book). The last chapter also includes a description of risk through consumption of animal products.

In addition to the protection targets, the level of protection also needs to be defined. The combination of protection target and protection level is often referred to as the ‘endpoint’.

The selection of appropriate protection targets and the level of protection in regulatory frameworks is primarily a policy decision. However, since the significance of protection targets and the levels of protection are often difficult to understand, policy decisions as to protection targets need to be supported by the scientific community.

1.1.4.4 Land Use

An important factor that affects both the risks and the degree to which those risks are evaluated, is the *land use* (often called: *function*) at a contaminated site. Generally speaking, the term land use applies to different categories that cover the main activity that is taking place on the site. Familiar land uses are *Residential land use*, *Industrial land use*, *Recreational land use, Children’s playgrounds, Infrastructural land use*, *Agricultural land use* and *Nature reserves*. Since sites with a similar land use can be used in quite different ways, the categorisation of land uses gives only a rough impression of the activities that are taking place at the site and the intensity of these activities. Therefore, land uses are sometimes subdivided, for example

Residential land use could be subdivided into 'residential with garden' (important with regard to exposure through vegetable consumption) and 'residential without garden' (no exposure through vegetable consumption). Moreover, several activities are covered by more than one land use. Housing, for example, is a prominent activity in Residential land use, but also occurs in Agricultural land use.

Risks for human health are strongly related to human behaviour. And human behaviour is highly impacted by the land use and activities taking place on the site. The degree to which risks are evaluated is mainly a policy decision. Generally speaking, the protection of human health risks warrants a greater weight in areas that are meant for human residence. It would be an option to give greater weight to human health protection in more densely populated areas, possibly with the weighting being proportional to the number of persons impacted, but this rarely occurs in existing contaminated site management frameworks.

In many countries with a high population density, land use changes in a relatively short term. The transformation from agricultural land to nature reserves and residential areas is especially common in many industrialised countries. In Europe, at least 2.8% of the land was subject to change in use between 1990 and 2000 (European Commission 2009) A change in land use has a profound effect on contact possibilities with the soil and on soil properties such as pH, organic matter dynamics (Römkens 1998) and, hence, mobility of contaminants and risks for humans, the ecosystem, the groundwater and Food Safety.

1.2 Soils and Sites

1.2.1 Soils

1.2.1.1 Definition

According to a broad definition, soil is the upper layer of the earth's crust or, in geological terms, the exterior weathered part of the earth's rocks. It has been formed out of rock material by physical, chemical and biological soil-forming processes over millions of years. Since climatical and geographical conditions varied over this long span of time, natural soils are typically characterised by a layered structure, that is, by the presence of *soil horizons*. According to a more popular definition, soil is the 'skin of the earth, representing the inheritance of human history'. This metaphor reflects the vital nature of soil, while at the same time referring to the presence of historical soil contamination.

Most natural soils have a darker coloured upper layer, the A horizon, with higher organic matter levels. In many regions of the world, there is a loose organic matter layer of humus of a few centimetres on top of this A horizon, called an O horizon.

Soil, structured or non-structured, consists of three different phases, namely, a solid (mineral and organic materials), a liquid (*pore water*), and a gas phase (*soil gas*). Moreover, it contains plant roots and an enormous number of different organisms of a wide variety. Soil includes two different entities, namely, a water-unsaturated upper soil layer (*upper soil*) and a water-saturated groundwater zone. These two entities are separated by a groundwater table. Typically, the gas phase is absent in the water-saturated groundwater zone. From the perspective of groundwater subtraction for the drinking water supply, a water-saturated groundwater zone with high water volume and replenishing capacities is referred to as the *aquifer*.

Some definitions of 'soil' only refer to the water-unsaturated upper layer which, depending on the depth of the water table, implies a layer of several centimetres in swamp areas and up to hundreds of meters in arid regions of the world. Other definitions, for example, from an agricultural perspective, link the term 'soil' to that part of the earth's crust that is actually used by humans. According to this definition soil usually includes the water-unsaturated upper soil layer and, often, the upper part of the water-saturated groundwater zone.

This book focuses on the impacts of contaminants. In this context, soil refers to that part of the earth's crust in which contaminants reside that might impact one of the protection targets. Impacts from non-private water supply (from deeper groundwater or surface waters) are excluded from this book, since it is assumed that Waterworks sufficiently control the water quality. As a consequence, the scope of this book roughly coincides with the 'agricultural definition' of soil, that is, the water-unsaturated upper soil layer and the first tens of meters of the aquifer.

For practical reasons, the terminology followed in this book is linked with the most common terms used in soil policies and management of contaminated sites, see the schematisation of soil as defined from a wider perspective, in Fig. 1.4. In

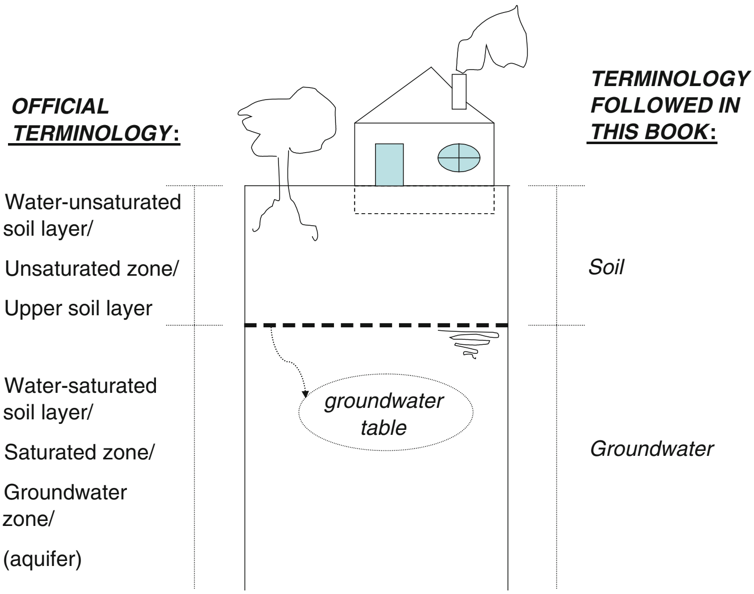

Fig. 1.4 Schematisation of soil

this much simpler terminology, the water-unsaturated layer above the groundwater table is often simply referred to as 'soil', while the water-saturated layer under the groundwater table is often called 'groundwater'. Note that in this book 'soil' is sometimes also used in the wider definition, for example, in the context of 'soil policy', or 'contaminated soils'. Clearly, both terms refer to both soil (in the more narrow definition of water-unsaturated upper layer) and groundwater.

An important difference between the upper soil and the aquifer is that groundwater is an important consumer product. This implies that clean groundwater has intrinsic value. From this perspective, the aquifer could be considered as a protection target. Soil, on the other hand, is not commonly used as a consumer product, but principally serves as an indispensable source for many useful products and activities.

It is important to realize that the separation between the upper soil layer and the aquifer, although from a scientific viewpoint convenient, is transient and partly artificial. With regard to the presence of contaminants, and the risks related to them, this separation is rather confusing. The reason for this is that contaminants migrate and do not necessarily belong to one of the two entities. Contaminants move throughout the soil-groundwater system, predominantly downwards, but sometimes also upwards and laterally, without acknowledging any borders between the soil and groundwater zones.

However, there are important differences between the water-unsaturated upper soil and the water-saturated aquifer. The upper soil, for example, enables the rapid transport of volatile contaminants via the gas phase. And the presence of a substantial amount of organic matter in the upper layer has a strong influence on the behaviour of contaminants and on the problems associated with these contaminants. Generally speaking, transport of water and contaminants is much faster in the groundwater than in the upper soil.

Another important difference between the upper soil and the deeper layers is the biological activity. Although organisms are found at every depth in the soil profile (see Chapter 13 by Swartjes et al., this book), the number of organisms is higher in the water-unsaturated upper soil, due to the presence of a gas phase. Within this upper soil layer, the number of organisms is even higher in the top of the soil, that is, the organic matter-rich layer that varies from a few centimetres up to several decimetres.

In many inhabited areas in the world the natural soil profiles are often disturbed. Many human activities, from the past and the present, are responsible for this feature, for example, (mechanical) digging activities in cities, tillage in agricultural areas or the addition to soils of foreign materials such as debris, stones, tar, and waste materials. Since most contaminated sites are within urban areas and disturbed soils are complex due to their heterogeneity, urban soil science is seen as a challenging, current frontier of soil science today (Norra 2006). In more extreme cases, whole new layers of soil material, mostly sand, sometimes clay, have been added onto the (natural) soil in many urban areas for infrastructural or filling purposes. Man-made soils, often with a high contribution of extraneous materials, are called *Technosols*. Generally speaking, soil structure is lacking in these artificial soil layers. In the lower parts of the Netherlands, for example, from the 16th century until

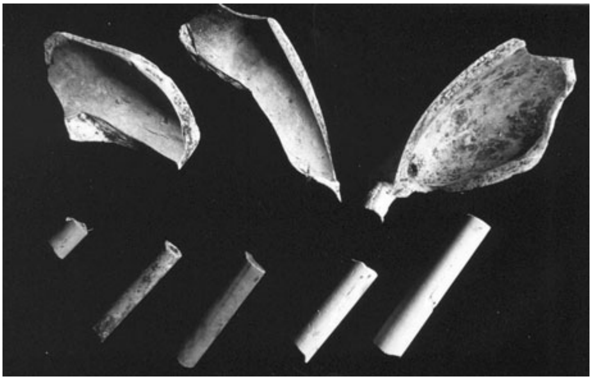

Fig. 1.5 Potsherds and pipe remains, found in the upper soil layers of the Central Western Peat area of the Netherlands, as evidence of layers of municipal waste of often several decimetres that have been brought onto the land from the 16th century until the 1940s (photo: M. Rutgers; reproduced with permission)

the 1940s, layers of municipal waste of often several decimetres have been brought onto the land in the Central Western Peat area, including potsherds and pipe remains (see Fig. 1.5), over areas as large as hundreds of square kilometres. Meuser and Blume (2001) describe the problematic classification of man-made soils, with regard to the anthropogenically influenced soils around the city of Osnabrück, Germany. Some of these soils contain coal and ore mining materials and waste materials. Meuser and Van de Graaff (Chapter 2 of this book) give a detailed description of the characteristics of and processes related to natural soils, urban soils and Technosols.

In fact, in all countries in the world all kind of waste materials have been brought onto the land over many centuries for the purpose of getting rid of these materials, whether or not in combination with soil improvement. After decades or centuries of evolution, these soils might have developed their own structure, for example, with an organic matter-rich upper layer. In any soil, whether disturbed or not, unconsolidated rock material on top of consolidated rock is found at greater depths. Typically, human activities that directly caused soil contamination also have resulted in disturbance of the original soil profile.

Unlike most bodies of water and air volumes, the soil is often privately owned. It is widely recognised that soil is a valuable and, at least on the time scale of decades, a non-renewable material. It serves several functions that are crucial for human survival such as crop production and as a supporter of buildings and humans themselves. Moreover, soil is the habitat and nutritional source for organisms.

1.2.2 Contaminated Sites

Contaminated sites are locations at which the soil and/or the groundwater are chemically polluted. In this context, a broad, three-dimensional definition is given to the concept ‘contaminated site’, including the soil (upper soil and aquifer) *underneath* the surface and the human occupation *on* the surface.

The focus of this book is on the threats for the four major protection targets; these are human health, the ecosystem, the groundwater and Food Safety, as related to contaminants present in the soil or the groundwater. From this perspective, the extent of the earth’s crust that is relevant in the context of this book is limited to that part that impacts human health and the ecosystem, and the groundwater that is within human reach. More concretely, this primarily relates to the upper, unsaturated soil layer and the first tens meters of the groundwater layer. To a lesser extent, those groundwater layers will be considered from which groundwater is extracted (up to several hundreds of meters). Since this book relates to compounds that have adverse effects, these compounds are called *contaminants* throughout the book.

Two types of contaminated sites exist with regard to the extent and shape of the location that is contaminated, namely, *diffuse* and *local* contaminated sites. This extent and shape of a contaminated site is often dependent on the type of source that is responsible for the contamination. Generally speaking, atmospheric deposition and, to a lesser extent, large scale agricultural activities lead to diffuse contamination. Diffuse contaminated sites caused by atmospheric deposition are characterised by large contaminated areas. Often the contaminant concentration decreases along regular circles from the source, for example, in the case of lead smelters (e.g., Filzek et al. (2004), who measured the metal concentrations in soil along a transect from a smelter at Avonmouth, UK), where the concentration contours are possibly stretched according to the wind direction. Diffuse contaminated sites caused by agricultural practices generally are characterised by a relatively homogeneous contamination pattern. One specific version of diffuse contaminated sites is known as *ribbon contaminations*, for example, along roads or railroad tracks.

Locally contaminated sites vary in size from a small back yard of a few square metres to an industrial site of several tens of thousands of square metres. These locally contaminated sites generally are characterised by a heterogeneous contamination pattern, often with one or more cores (hotspots) of contamination, related to the source of the contamination. In many cases, the larger locally contaminated sites can be considered as a collection of smaller locally contaminated sites. There is no absolute definition of diffuse or local contaminated sites. Some sites have characteristics of diffuse and local contaminated sites combined, for example, in large diffusively contaminated sites with contaminant hotspots.

This book deals with contaminated sites, either diffusely contaminated or locally contaminated. However, since most chapters of this book deal with Risk Assessment tools that can be used for any type of contaminated site, this distinction is not always relevant.

In principle, the book does not implicitly deal with agriculturally managed sites. The reason for this is that agricultural activities often lead to a continuous supply of contaminants to the soil as part of agricultural business. This means that managing the contaminant status of agricultural sites, and the related risks, is a matter of balancing the inputs and outputs of contaminants. Pesticide application, for example, is focused on administering the applications needed for the goal to be reached (for example, prevention of crop diseases), while the load for soil and groundwater must be acceptable. The consequence is that the soil inputs from agriculture have been regulated in specific legislation in practically all countries in the world.

1.3 Contaminants

1.3.1 Terminology

No chemical substance leads to toxic effects by definition. Whether substances will cause toxic effects depends on the combination of exposure, the nature of the substance and the characteristics of the receptor (a human being or an organism). The overall exposure depends on the dose which humans or organisms are exposed to, the period over which this exposure takes place, the frequency of the exposure and the form (*speciation*) in which the chemical substance is available.

Several terms are used for the very generic term ‘chemical substance’ in contaminated site management. Often the term ‘compound’ is used. However, this term is considered too generic in the context of this book. Moreover, it literally does not cover all chemical substances in soil, since ‘compound’ refers to chemical substances that are composed of two or more elements, which means that pure metals are excluded. Alternatively, the term ‘compound of concern’ (or ‘chemical of concern’) (COC) is sometimes used. Of all the terms used, the word ‘pollutant’ evokes the most negative association, since definitions include adjectives such as ‘harmful’, ‘unsuitable’ or even ‘toxic’. The term ‘contaminant’ is used throughout this book, although it also has a negative ring, but this term does the best justice to the ‘potential’ aspect of causing adverse effects.

1.3.2 Daily Life

Potentially toxic chemical substances in the soil (*contaminants*) are part of our daily life. In modern times, humans and organisms are continuously exposed to a wide spectrum of contaminants. Humans are surrounded by all kinds of materials that contain a variety of potentially harmful chemicals, on a daily basis. Cloth, furniture, decorative objects or children’s toys, all contain chemical substances that are potentially toxic. Humans even eat and drink materials and inhale air that contains contaminants that are designated on several lists of Soil Quality Standards. Analogously, soil organisms are surrounded by all kinds of contaminants. They also feed on contaminant-holding materials.

Actually, humans have been in contact with contaminants since early human existence, due to the presence of metals in the soil, for example, or through PAHs from the burning of wood and roasting of meat. However, as long as humans lived in equilibrium with nature, exposure was limited and the threat to human health from contaminant exposure was generally negligible.

1.3.3 Categorisation

Since hundreds of thousands of contaminants are present in the environment, it is useful to categorise them. Several criteria can be used for this purpose, such as 'related production processes' (for example, heavy metals from zinc smelters, and cyanide from gas works), 'type of application' (for example, pesticides) or 'chemical or physical characterisation'. A systematic categorisation is given in Fig. 1.6.

A popular policy-related categorisation is given here that is partially based on chemical or physical characterisation and, hence, chemical properties. This results in the following six categories:

- metals and metalloids;
- Polycyclic Aromatic Hydrocarbons (PAHs);
- monocyclic aromatic contaminants;
- persistent organic pollutants (POPs);
- volatile organic contaminants (VOCs);
- other organochlorides.

Note that some of these categories overlap. In addition, three other useful categories can be added, based on 'frequency of occurrence of contaminants in soils'; these are:

- other inorganic contaminants (other than metals);
- petrolium hydrocarbons;
- asbestos.

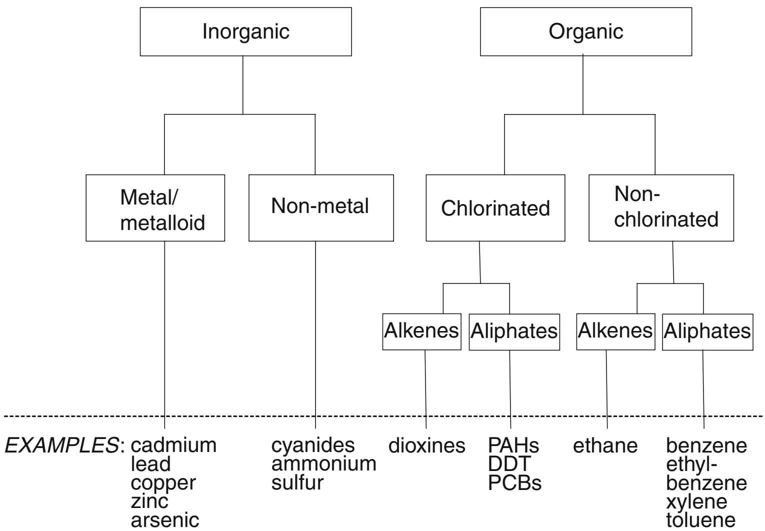

Fig. 1.6 A systematic categorisation of contaminants

1.3.3.1 Metals and Metalloids

'Metals and metalloids' is an important group of soil contaminants, since they are very often found in soils practically everywhere in the world. Metals (from the Greek word *metallon*) usually are characterised by a hard, malleable and shiny appearance, mostly solid at room temperature, with a high density and a high melting point and a good conductance of electricity and heat. Since metals readily lose electrons, they easily form positive ions (cations) in soils and, hence, have metallic bonds between metal atoms and ionic bonds with non-metals. Examples of the latter often found in soils are salts of metals and anions such as chloride (Cl^-) and carbonate ($CO_3{}^{2-}$). Different from most other contaminants, metals are elements included in the periodic table. Often the term 'heavy metals' is used for a sub-group of metals that are very often found at contaminated sites, although according to Duffus (2002) the term is controversial and archaic. Very important representatives with regard to soil contamination are cadmium (Cd), copper (Cu), lead (Pb), zinc (Zn), mercury (Hg), and selenium (Se).

Except for metals, this category of contaminants also includes the metalloids (or semi-metals), representatives of which, such as arsenic (As) and boron (B), are also often found in soils. These metalloids can be considered as transitional elements between metals and non-metals.

It is important to realise that metals and metalloids do not have a specific exposure affinity. The exposure depends on many factors, among them the form (speciation) in which the contaminant is available in the soil. This speciation follows from a combination of specific contaminant characteristics, soil properties and other chemical elements present. Many different speciations are found in soils for each metal. In this respect, a Soil Quality Standard for a specific metal could be considered as a lump standard for a whole group of chemicals which have the presence of the metal in common.

A general phenomenon in many residential areas is the presence of lead in soil, mainly due to the former use of tetraethyl lead in gasoline as an anti-knock agent (e.g., Wong and Xiang (2004), who measured elevated lead concentrations due to traffic activities in Hong Kong, China). This generally is a long-lasting major problem, since lead can result in retardation of the brain development of young children and is relatively immobile and will stay in the soil for decades or even centuries.

Another common problem is the presence of cadmium at agricultural sites or vegetable gardens (e.g., Wong et al. (2002) who measured enriched cadmium concentrations in crops, paddy and natural soils in the Pearl River Delta, one of the most developed regions in China; heavy metal enrichment was most significant in the crop soils, which might be attributed to the use of agrochemicals). Cadmium, which is often found in soils, due to atmospheric deposition from smelters or the application of fertilizers, is easily taken up by vegetables and can induce kidney dysfunction and several types of cancer at relatively low exposures.

1.3.3.2 Other Inorganic Contaminants (Other than Metals and Metalloids)

A specific case of other inorganic contaminants are the co-called *nutrients*. With regard to plant nutrition and, hence, soil contamination, the most relevant nutrients are the macro-nutrients nitrogen, phosphorus, potassium, and sulphur. These nutrients are needed in relatively large quantities in agricultural management, and are usually applied as nitrate, phosphate, potassium salts, and sulphate. Since this book does not primarily focus on agricultural practices, no further attention will be given to (the consequences of) nutrients.

A representative of the category of other inorganic contaminants often found in soils is cyanide, since cyanides are frequently found at former gas work sites, often in (inner) cities. Cyanide is a contaminant that contains a cyano group (C≡N) as a functional group, often found as the anion CN^-. Many organic contaminants feature cyanide as a functional group. Of the many kinds of cyanide contaminants, some are gases, while others are solids or liquids. Those that can release the cyanide ion CN^- are highly toxic.

Although it has been shown that rhizobacteria are cyanogenic (that is, able to synthesize cyanides), and hence negatively impact the seedling root growth of various plants (Kremer and Souissi 2001), most cyanide in soil has an anthropogenic origin. However, since cyanide is mainly present as iron cyanide complexes at gas work sites, the risk of effects on humans from exposure to cyanides often seems to be of minor relevance (Kjeldsen 1999).

1.3.3.3 Polycyclic Aromatic Hydrocarbons

Polycyclic Aromatic Hydrocarbons (PAHs) form also an extremely important group in regard to soil contamination, since they are among the most widespread contaminants found in soils, worldwide. They are characterised by a fusion of aromatic rings and do not contain many heteroatoms (atoms other than carbon or hydrogen). PAHs are primarily formed by incomplete combustion of carbon-containing fuels such as wood, coal, diesel, fat, tobacco, or incense (Fetzer 2000), and are concentrated in oil, tar and coal. Common PAHs in soil are naphthalene, phenanthrene, anthracene, fluoranthene, benzo[a]anthracene, benzo[k]fluoranthene indeno[1,2,3-cd]pyrene, benzo[g,h,i]perylene and benzo[a]pyrene. PAHs in soils might show local or diffuse (due to atmospheric deposition) contamination patterns. Different types of combustion yield different combinations of PAHs, both in terms of relative amounts of individual PAHs and with regard to the isomers that are produced.

Some PAH representatives are known or suspected to be carcinogenic, mutagenic, or teratogenic.

1.3.3.4 Monocyclic Aromatic Hydrocarbons

Some monocyclic aromatic hydrocarbons are frequently found in soil and groundwater. The representatives most often found are usually categorized as BTEX (benzene, toluene, ethylbenzene and xylenes). They were, or are, used on a large scale in cleaning applications such as degreasing.

Sort-term effects due to exposure of monocyclic aromatic hydrocarbons usually relate to skin and sensory irritation, as well as effects on the respiratory system and the central nervous system. Prolonged exposure to these contaminants also affects these organs as well as the kidney, liver and blood systems (Oregon Department of Human Services 1994). According to the US Environmental Protection Agency, there is sufficient evidence from both human epidemiological and animal studies to denote benzene as a human carcinogen. Workers exposed to high levels of benzene in occupational settings have been found to have an increased number of cases of leukaemia.

1.3.3.5 Persistent Organic Pollutants

Persistent Organic Pollutants (POPs) are organic contaminants that are resistant to chemical and biological degradation processes and to photolytic processes. For this reason they are capable of persisting in the environment and bioaccumulate in human and animal tissue. POPs are often halogenated, usually with chlorine. The United Nations Environment Programme Governing Council (GC) includes the following contaminants as POPs: aldrin, chlordane, DDT, dieldrin, endrin, heptachlor, hexachlorobenzene, mirex, polychlorinated biphenyls (PCBs), polychlorinated dibenzo-p-dioxins, polychlorinated dibenzofurans, toxaphene, certain brominated flame-retardants, some organ metallic contaminants such as tributyl tin (TBT), as well as some Polycyclic Aromatic Hydrocarbons (PAHs). Many POPs are pesticides which are banned in many countries, but still are found in soils and will reside in the soil for many more decades. In addition, POPs can originate from the production of solvents, polyvinyl chloride, and pharmaceuticals. Generally speaking, POPs have a high molecular mass and show a low water solubility, high lipid solubility and limited volatility.

According to the Stockholm Convention, POPs can lead to serious health effects, including certain cancers, birth defects, dysfunctional immune and reproductive systems, greater susceptibility to disease and diminished intelligence (Stockholm Convention 2009).

1.3.3.6 Volatile Organic Contaminants

Volatile organic contaminants (VOCs) are generally characterised by high enough vapour pressures under normal conditions to significantly vaporize. There is not one univocal exact definition for these contaminants. Under European law, the definition of VOCs is based on evaporation into the atmosphere, rather than reactivity. In the European Union Directive 2004/42/CE, for example, VOCs are defined as an 'organic compound having an initial boiling point less than or equal to 250°C, measured at a standard atmospheric pressure of 101.3 kPa' (European Union 2008). The US Environmental Protection Agency defines VOCs as 'any compound of carbon, excluding carbon monoxide, carbon dioxide, carbonic acid, metallic carbides or carbonates, and ammonium carbonate, which participates in atmospheric photochemical reactions' (US EPA 2008). Infamous sources of VOCs are dry cleaning facilities. Other sources are paint, fabric softeners, petroleum fuels (e.g., gasoline), and crude oil. Moreover, several indoor sources are recognized, for example, photocopiers, carpet backings, and furniture. Widespread VOCs in soils include trichloroethylene, tetrachloroethylene, 1,1,1- trichloroethane, vinyl chloride, and, to a lesser extent, glycol ethers, hexane, formaldehyde, methyl bromide, methyl chloride and methyl ethyl ketone.

VOCs are readily soluble in fat. They may result in many different effects on human health, mainly after inhalation, ranging from dizziness, via narcotic effects, to neurotoxicological effects. Some agents of this group are carcinogenic, mutagenic or tetragenic.

1.3.3.7 Other Organochlorides

Organochlorides contain at least one chlorine atom. These chemicals are typically non-aqueous and are usually denser than water due to the presence of heavy chlorine atoms. The simplest forms of organochlorides are chlorinated hydrocarbons. These consist of simple hydrocarbons in which one or more hydrogen atoms have been replaced with chlorine. Many chlorinated hydrocarbons (e.g., dichloromethane, dichloroethene, trichloroethane, chloroform, and dioxins) are used as solvents. These solvents tend to be relatively non-polar and are therefore immiscible with water and effective in cleaning applications such as degreasing and dry cleaning. Other organochlorides are used as effective insecticides, such as DDT, heptachlor, endosulfan, chlordane, and pentachlorophenol. Polychlorinated biphenyls (PCBs) were once commonly used in electrical insulators and heat transfer agents. Their use has generally been phased out due to health concerns. Actually, BTEX (benzene, toluene, ethylbenzene and xylenes), here classified as monocyclic aromatic hydrocarbon, also could be included in this category.

Organochlorines generally affect the stomach, blood, liver, kidneys, and the nervous system.

1.3.3.8 Petroleum Hydrocarbons

Petroleum hydrocarbons (aka: petrol- or gasoline-related hydrocarbons; often called Total Petroleum Hydrocarbons or TPH) is a group of frequently found contaminants, which actually are complex mixtures of a whole spectrum of contaminants. These separate contaminants, which can add up to several hundred chemical compounds, mainly are hydrocarbons, both aliphatic and aromatic, and a whole spectrum of additives such as benzene, toluene, xylenes, naphthalene, and fluorene. No specific petroleum hydrocarbon mixture equals another existing petroleum hydrocarbon mixture. TPH compounds can affect the central nervous system, the blood, immune system, lungs, skin, and eyes or cause headaches, dizziness or a nerve disorder called 'peripheral neuropathy,' consisting of numbness in the feet and legs (ATSDR 2009). Several TPH compounds are (probably or possibly) carcinogenic.

For a long time, risk assessors seemed quite helpless with regard to the risk-based assessment of TPH. In some countries (e.g., the USA, the UK, Australia, and the Netherlands) expert judgement-based Soil Quality Standards have been implemented. An elegant approach for dealing with these complex mixtures was provided by Franken et al. (1999). They described a procedure for dealing with the human health risks of petroleum hydrocarbons, based on five groups of aliphatic hydrocarbons and five groups of aromatic hydrocarbons. Analogous to the US Total Petroleum Hydrocarbon Criteria Working Group, these hydrocarbon groups are characterized by a specific equivalent carbon number index range, representing equivalent boiling points.

An overview of the detection and remediation of soil and groundwater contaminated with petroleum products is given in Nadim et al. (2000).

1.3.3.9 Asbestos

The contaminant that provided the ultimate challenge for risk assessors, maybe even more than oil and petrol-like mixtures, is asbestos. Asbestos is also frequently found in soils. First of all, asbestos distinguishes itself from almost all other contaminants by the fact that asbestos is a mineral. It also is completely different from other contaminants in its behaviour: asbestos does not adsorb to soil particles and does not migrate through soils via the pore water, or soil gas. It is also not taken up by plants. The only pathway by which asbestos can give rise to adverse effects is by inhalation. These effects, although very serious (mesothelioma, that is, cancer of the pulmonary membrane and peritoneum, asbestosis, and increased risks for lung cancer), will reveal themselves over the longer term, that is, decades after exposure.

A concrete way of dealing with asbestos is described in Swartjes and Tromp (2008). They derived a Soil Quality Standard (Intervention Value) from measured data and described a tiered approach (as preferably used for other (composited) contaminants) to assess the site-specific risks of asbestos in soils. In the first step, measured asbestos concentrations in soil are compared with the Intervention Value of 100 mg/kg$_{\text{dw}}$ asbestos equivalents (0.01% by weight). 'Asbestos equivalents' is the sum of the concentration of chrysotile asbestos (also serpentine asbestos or white asbestos) and 10 times the concentration of amphibole asbestos (other asbestos types), for both friable and bound asbestos. When this value is exceeded, a tiered approach is used for the determination of site-specific human health risks. A site-specific human risk is assumed, unless it can be proved otherwise ('risk, unless. . ..'). The three tiers are as follows:

- Tier 1, Simple test: investigating the possibilities/likelihood of exposure;
- Tier 2, Determination of the *respirable fraction* in soil: investigating the possible site-specific exposure to humans, independent of the actual site use or site-specific elements, based on the determination of the respirable concentration of asbestos fibres in soil, in conformity with the Dutch standard NEN 5707.
- Tier 3, Measurement and testing of the concentration of asbestos fibres in outdoor and indoor air under standardised conditions.

1.3.4 Occurrence in Soils and Groundwater

Contaminants coming from various natural and anthropogenic sources may be found in soils. Contaminants enter the soil via emissions onto the soil surface, usually unintentionally (e.g., through atmospheric deposition, spills, etc.), sometimes intentionally (e.g., the use of metal-containing fertilizers, illegal dumping). In most cases, the contaminants migrate downwards. The velocity of migration varies greatly, depending on the type of contaminant, soil type, soil properties and climatic conditions. Ultimately, contaminants leach into the groundwater, where migration continues, both in vertical and in horizontal directions. As a consequence, contaminants are found in the entire depth range of a soil. However, since most soils have an upper layer with high organic matter content, which provides a high potential for adsorption of both inorganic and organic contaminants, generally the total soil concentration is higher 'in the first few decimetres' of the soil.

Since contaminant characteristics differ widely, the contamination profile vis-à-vis depth also differs. The shape of this profile is determined by the sorption, desorption and degradation potential of contaminants and of the physical characteristics (water flow transport) and physico-chemical characteristics (sorption and desorption) of the soil. Generally speaking, immobile contaminants have a higher ratio between the concentration in the solid phase of the soil and the concentration in the pore water or the groundwater than do mobile contaminants. However, all contaminants have higher concentrations in the solid phase of the soil and, hence, higher total soil concentrations in the upper soil, rich in organic matter and in clay horizons (mainly metals).

For practical reasons the concentration of contaminants in soil is usually expressed by weight of contaminants per unit weight of dry soil (kilogram), while the concentration of contaminants in groundwater commonly is expressed by weight of contaminants per unit of volume (l). Since this proved to result in the most convenient figures, the contaminant weight is usually expressed in milligram (mg) for soil and in microgram (μg) for groundwater. In summary, the concentrations are expressed as $\mathrm{mg_{contaminant}/kg_{soil,\ dry\ weight}}$ for soil and in $\mathrm{\mu g_{contaminant}/l_{groundwater}}$ for groundwater, most often shortened to $\mathrm{mg/kg_{dry\ weight}}$ (or mg/kg), and μg/l, respectively.

1.3.5 Mixtures of Contaminants

In the great majority of contaminated sites, more than one contaminant is found in soil or groundwater. There are two reasons for this. First, most materials from which contaminants originate contain more than one contaminant. Metal ores, for example, often contain several metals which may be simultaneously released from metallurgic industrial processes. Similarly, in most activities or processes, where contaminants are released into the environment, several contaminants are involved. One example of this is a dry cleaning facility, where several chlorinated hydrocarbons are simultaneously used. From both these examples it can be concluded that the same combinations of contaminants are often found in soils and groundwater. The risk assessor needs to use this information in order to investigate the site for the whole contaminant mixture. The exact composition of contaminants in soils and groundwater, however, may differ. By incomplete combustion of different organic materials, for example, different mixtures of PAHs are produced, depending on the type of organic material and combustion characteristics such as temperature.

Second, specific sites, for example sites just outside the city limits of several of the larger cities around the turn of the nineteenth and twentieth centuries, lent themselves to several soil contaminating activities. At these sites, an often incoherent cocktail of contaminants is present. For these sites, it is more difficult to determine which contaminants to search for.

1.3.6 Scope of This Book

A specific class of potentially harmful contaminants found in soil consists of *radioactive contaminants* (e.g., Callahan et al. (2004), who evaluated the human health risks due to the presence of depleted uranium at a military training site in the USA). Since radioactive substances are of a different nature and require a different kind of Risk Assessment, these contaminants do not fall within the scope of this book. For the same reason *endocrine disruptors* (aka: 'hormonally active agents'; see Lintelmann et al. (2003), who provided an overview of the biochemical and biological background of endocrine disrupters in the environment) are not considered in the scope of this book. Furthermore, no attention will be paid in this book to the *microbial contaminants*, mainly relevant in groundwater, that originate from both human and animal faeces via sewer leaks, septic tanks and manure disposal, although these are of great concern for human health (e.g., Celico et al. (2004), who found several microbial contaminants, related to pasture and/or manure spreading, in different carbonate aquifers of southern Italy).

Recently, there has been much attention paid to the impact of *nanoparticles* in the environment. Since these nanoparticles are central to many natural processes in soil and groundwater and in human physiology, they are a potential threat to the soil ecosystem and human health. Given the limited scope of their use, it is currently unlikely that they pose a substantial risk to the soil ecosystem and human health (Colvin 2003). However, since the widespread use of nanomaterials will result in higher concentrations in soils, the future impact is unknown. For the same reasons that radioactive contaminants and endocrine disruptors require a different kind of Risk Assessment, nanoparticles are not considered in the scope of this book.

1.4 Site Characterisation

Site characterisation is an essential step in identifying contaminated sites and, in the steps that follow, contaminated site management. Therefore, each project involving contaminated sites needs to begin with a preliminary study of the site under investigation. This study includes information on the layout of the site (the presence of buildings, sealed surfaces, bare surfaces, vegetation, (micro)relief, the presence of soil-foreign materials, elements that relate to former activities) and a detailed evaluation of the history of the site (which activities might have been responsible for which contaminants, at which spots on the site). Some sites evoke a clear suspicion of being contaminated, while other sites have a rather innocent appearance with regard to soil contamination. See Fig. 1.7, as an example, in which a suspicious site, based on the presence of drums (upper photo), and a site that seems above suspicion (lower photo), both in the Silvermines area in Tipperary county, Ireland, contaminated with several metals due to former mining activities.

A *site visit*, including a so-called *organoleptic investigation* ('looking and smelling'), is an essential activity at this stage of the project. The evaluation of the history of the site might include a visit to the municipal archives and the historical records found in the library. Moreover, interviews with former workers or inhabitants might be helpful. A map of the site might also help in the interpretation, and digital photos will support the memory of the risk assessor. This preliminary study should result in a hypothesis about the type of contaminants and the spots where these contaminants can be present.

Obviously, samples need to be taken and analysed in order to determine the concentrations in soil and groundwater. If no information about the possible contaminants is known, the samples can be analysed for a group of 'frequently found contaminants'. This group differs for soil and groundwater, since the more immobile contaminants are often found in soils, while the more mobile contaminants usually reside in the groundwater. Several countries have defined standard groups of frequently found contaminants, often formalised in protocols. In the Dutch NEN 5740 protocol, for example, a standard series of contaminants has been defined that must be determined when there is no information about the possible contaminants present (NEN 2009). The selected contaminants differ for:

Fig. 1.7 A suspicious site (**a**) and a site that seems above suspicion (**b**), in the Silvermines area in Tipperary county, Ireland, contaminated with several metals due to former mining activities (photo: F. Swartjes)

- soil: barium (Ba), cadmium (Cd), cobalt (Co), cupper (Cu), mercury (Hg), lead (Pb), nickel (Ni), selenium (Se), vanadium (V), zinc (Zn), chloride (Cl), mineral oil, sum of EOX (Extractable Organic Halogens) and the sum of PAHs (10 specified representatives);
- and groundwater: cadmium (Cd), cobalt (Co), cupper (Cu), mercury (Hg), lead (Pb), nickel (Ni), selenium (Se), vanadium (V), zinc (Zn), chloride (Cl) , mineral oil, naphthalene, some specified volatile aromatic hydrocarbons (including BTEX), and some specified volatile halogenated hydrocarbons.

Since sampling and laboratory analysis are relatively expensive, generally there is lack of data. Multivariate and geostatistical tools can support the characterisation of a site, e.g., Carlon et al. (2000) who extracted additional PAH concentrations by Kriging interpolation of spatial data and Principle Component Analyses (PCA), at an industrial site close to Parma, Italy.

Generally, two important decisions need to be taken, which require a combination of science and pragmatism. First, the number of samples needs to be determined. Often, the number of samples that results from a pure statistical analysis is too costly. Therefore, statistics need to be combined with pragmatism. In Lamé (Chapter 3 of this book) the procedure for sampling has been described, primarily from a practical perspective. In Brus (Chapter 4 of this book), this procedure is approached from a statistical perspective.

Second, a decision needs to be taken about the construction of *composite soil samples*; these are lumped samples through mixing of separate samples. Obviously, a chemical analysis of composite samples is factors cheaper than a sampling of the separate samples. In case the composite samples do not provide enough information for a well-founded risk appraisal, appropriate separate samples could be analysed at a later stage. The decisions pertaining to the number of individual samples and composite samples depend on the degree of heterogeneity of the contaminant (and of some important soil characteristics such as pH, organic matter content) in the soil and groundwater.

Also the (statistical) interpretation of the measured concentrations is important. Altfelder et al. (2002), for example, showed that part of the area that may be declared safe based on merely kriged estimates can actually exceed the German limit values by a probability of up to 50%. Millis et al. (2004) showed for lettuce (variety Crispino) that variation in plant-scale heterogeneity of cadmium in soil affects bioavailability and hence the concentration factors plant-soil by a factor of two.

1.5 Risk Assessment

1.5.1 Principles

A measured concentration in soil or groundwater is a rather vague criterion with regard to determine possible associated problems. The simple purpose of Risk Assessment is to transfer this measured concentration into a more manageable appraisal of the status of the contaminated site in terms of risks for one of the protection targets (human health, the ecosystem, groundwater or Food Safety). Let's consider, as an example, the appraisal of a PCB soil concentration of 1 mg/kg_{dw} soil. Without Risk Assessment, it is extremely difficult to give an objective and useful opinion about this measured concentration. To some, it might be a non-problem, since 1 mg/kg_{dw} soil implies one in a million, and that seems very low when compared with some (undefined) standard of high and low. But another person might approach the case differently, that is, by noting that 1 mg PCBs equals about $1.8.10^{18}$ molecules.[1] Such a high amount of a contaminant that is able to impact the immune, hormone, nervous, and enzyme systems is associated with serious health problems, again by comparing it with some (undefined) standard of high and low. But obviously both positions are not very useful, since the numbers do not tell anything about the magnitude of the problem. And it is exactly that, an estimate of the magnitude of the problem, which is the purpose of Risk Assessment.

1.5.2 The Concept of Risk

Risk is a concept that denotes a *potential negative impact to an asset*. There must be a source for this potential negative impact, and this is generally called a hazard. With regard to contaminated sites, the hazards are the adverse effects on human health from contaminants in the soil or groundwater.

Many authors describe the magnitude of a risk in terms of probability (or change, or frequency) and effect (harm). Since a doubling of the probability of a negative impact on an asset often is judged similar to the doubling of the effect, risk is often described as the multiplication of probability and effect. The determination and often the evaluation of risks are called *Risk Assessment* and helps in making transparent, rational, and defendable decisions.

With regard to the seriousness of an effect, it is very important if, and if so, to what extent, one can influence the probability of a negative impact on an asset. In this respect, it is useful to distinguish between a risk that humans deliberately take, for example, the risk of getting lung cancer from smoking (a *voluntary risk*), and a risk that is beyond human control, for example, the risk of a natural catastrophe (an *imposed risk*). Humans can control voluntary risks, for example, by reducing the number of cigarettes they smoke. Imposed risks, on the contrary, are not or are difficult to manage. At best, if one is prepared to take extreme measures that often impact one's personal circumstances, some risks can be reduced, for example, by moving to a place on the globe where the chance of natural catastrophes is relatively low.

Risk, both voluntary and imposed risks, relates to a concept we deal with on a daily basis. Some examples of familiar voluntary risks, with human health as the asset that can be negatively impacted, relate to the consumption of alcohol-containing drinks, going out in traffic, and engaging in sport activities where injuries

[1] Assuming an average average molecular weight for PCBs of 327 g/mol.

to joints are possible. Clearly, we are prepared to take some of these risks, since these activities provide us with evident advantages. Some examples of imposed risks are living in polder regions beneath sea level, protected by dunes and dikes, the threat of natural catastrophes in hurricane-prone areas of the world and of terrorist attacks.

Risks from contaminated sites typically are in the category of imposed risks: humans can only avoid, or at best reduce the risks, by adapting their lifestyle or, in the most extreme case, by moving to a place where the soil has not been impacted by soil contamination. From the perspective of Risk Assessment, it is very important to realise that risk is not necessarily a bad thing. Being at risk is part of our life; many risks will never impact human health, and there are many risks we are not even aware of.

When contaminants are present in soil there is a risk by definition, since there is ‘a probability’ (the chance, although miniscule at very low concentrations, that human beings or soil organisms are exposed to these contaminants) and ‘an effect’ (impact on human health or ecosystem health, when contaminants indeed intrude on humans or soil organisms). The whole idea behind Risk Assessment is not to find out whether there are risks, but to investigate whether or not the risks are *acceptable*. Nevertheless, risk assessors often use the phrase ‘no risks’ when in fact ‘no unacceptable risks’, or more concretely ‘acceptable risks’ is meant. Often, regulators or stakeholders seduce the risk assessor into using the ‘no risks’ qualification, since the ‘no unacceptable risk’ qualification (double denial), or ‘acceptable risks’, is more difficult to interpret and often provokes further discussion.

Risk Assessment can also be used to support the optimal allocation of financial resources in contaminated sites projects.

1.5.3 Procedure

Risk Assessment (aka: Risk Analyses) is a process which serves the purpose of examining risks and, when possible, quantifying risks. It is an old concept. Risk Assessment can almost be considered as a science unto itself. It is used in widely differing disciplines such as environmental engineering, the design of building constructions, financial impact assessments or in the military. Analogous to risks, we are dealing with Risk Assessment daily, mostly without realising it. When a person crosses the road, for example, that person makes a judgement on the chance of being hit by a passing vehicle and the following consequences.

The Risk Assessment framework is illustrated by the light-shaded boxes in the contaminated site management framework, in Fig. 1.3. The first step in the Risk Assessment framework includes two different activities, namely, the Exposure Assessment (aka: dose assessment) and the Hazard Assessment (aka: effect assessment), mainly used in Human Health Risk Assessment. Conventionally, the determination of exposure is performed for human beings or larger animals, and not so much for smaller soil-dwelling organisms. Ideally, the amount of a contaminant that reaches the blood stream (in case of systemic effects, i.e., related to effects in the whole body after systemic circulation and, hence, absorption and distribution in the body) or target organs (in case of local effects, i.e., related to effects to specific organs at the place of contact or intake) is determined. This amount is expressed as mass contaminant, per body weight mass, per unit of time (mg/kg$_{\text{body weight}}$/day), that is, as the *internal* exposure. Target organs are the organs that could be adversely affected by specific contaminants. However, in most cases the *external* exposure is calculated, that is, the amount of a contaminant that reaches the human body or the organism.

With regard to the determination of human exposure, multimedia calculations are usually combined with the calculation of human exposure in so-called exposure models. A more detailed introduction to Human Health Risk Assessment is given in Swartjes and Cornelis (Chapter 5 of this book). A quantitative determination of human exposure is described in detail in Chapters 7, 8, 9, 10, and 11 of this book.

An important first step in Exposure Assessment is to determine the *representative soil concentration*. The representative soil concentration, and hence, the procedure for soil and groundwater sampling, is dependent on the purpose of the Risk Assessment, the site characteristics and the exposure pathways that are the most relevant. Smart choices need to be made for location and depth of the samples, construction of the composite samples, and contaminants that must be analysed (see Section 1.4). A site visit and historical survey can be important activities in support of the chosen sampling strategy.

Hazard Assessment includes two steps, namely, *Hazard Identification* and *Hazard Characterisation* (IPCS 2004). Hazard Identification focuses on the possible effects of specific contaminants and the time frame for which these effects occur. Subsequently, the Hazard Characterisation results in a dose-response assessment, relating exposure to effects, and is the basis for the determination of *Critical Exposure* (aka: Reference Dose).

The combination of Exposure Assessment and Hazard Assessment, the second step of the Risk Assessment framework, is called *Risk Characterisation*. When translated in objective terms, the Risk Characterisation results in a *Risk Index*. This is the ratio between *actual exposure* and Critical Exposure with regard to Human Health Risk Assessment, or the ratio between the actual concentration and acceptable concentration in the soil with regard to Ecological Risk Assessment, respectively. In Ecological Risk Assessments often the PEC/NEC (Predicted Effect Concentration/No Effect Concentration) ratio is used for one organism, several organisms or the whole ecosystem, with the same goal in mind. Dawson et al. (2007), as another example, established a Biological Soil Quality Index to help visualize significant differences in hydrocarbon-polluted soils.

The actual performance of Risk Assessment is generally supported by appropriate *Risk Assessment tools*. In Swartjes et al. (2009) a Risk Assessment tool is defined as any instrument that can contribute to the determination of risks at a contaminated site. A Risk Assessment tool can be an equation, a description, a database, a model, an instrument, a protocol, or a table. A combination of selected Risk Assessment tools is called a *Toolbox*. Such a Toolbox does not include policy points of view. The Toolbox for the determination of human health risks, for example, may include algorithms for the calculation of exposure through different pathways, a measurement protocol for the determination of the indoor air concentration, a table with Critical Exposure values, and many more Risk Assessment tools. A Decision Support System (DSS) does include selected Risk Assessment tools, but also policy points of view. A DDS could for example be used to calculate national Soil Quality Standards on the basis of several selected Risk Assessment tools and national policy points of view.

It is often said that Risk Assessment is an objective process and that scientists need to operate independently of the interests of any stakeholder. To a certain extent this is truth, since scientific independence is the key to an objective risk qualification. This independent position, however, certainly does not justify a strict 'no communication policy'. The reason for this is that Risk Assessment includes several policy decisions, for example, as to the degree of conservatism and the required level of protection of human health, the soil ecosystem, the groundwater or agricultural products.

The independent status of scientist will not be affected by the adaptation of specific political boundary conditions, as long as it is made transparent what these boundary conditions are. Risk assessors can do an excellent and objective job when they, for example, commit themselves to the political boundary condition that a Risk Assessment for an industrial site should focus on 'average adult workers' and not relate to children or other sensitive groups. Again, it is important to make these boundary conditions and, hence, the validity range of the conclusions from the Risk Assessment, transparent. Therefore, this political boundary condition needs to be clearly described in the Risk Assessment report. This enables regulators to guarantee the safety of these sensitive groups, for example, by fencing of the site with anti-trespassing controls in order to protect children as in the case just mentioned.

In the USA, there is a standardised procedure for performing Human Health and Ecological Risk Assessment called *Risk-Based Corrective Action (RBCA)*. A summary of this three-tiered Risk Assessment procedure is given in ASTM (2009).

It is generally acknowledged that the total concentration is not the optimal measurement with regard to risk to the soil ecosystem, the groundwater and, to a lesser extent, for human health. Especially exposure and leaching are strongly related to an 'effective' fraction of the contaminant in soil, this is, the *bioavailable fraction* with regard to ecological risks and a specific part of human health risks (like the risk through vegetable consumption) and the *available fraction* with regard to leaching from the upper soil into the groundwater. In Mallants et al. (Chapter 18 of this book) a detailed description of the leaching process is given. The (bio)available relevant fraction depends on the type of organism and, last but not least, the relevant timeframe.

An enormous number of papers have been written on calculating and measuring bioavailability, in particular with regard to metals. An example is given in Alvarenga et al. (2008), who determined two bioavailable metal fractions, that is, a 'mobile fraction' and a 'mobilisable fraction' using a sequential extraction, with the purpose to assess the risks in an acid metal-contaminated soil from the Aljustrel mining area in Southwest Portugal, in the Iberian Pyrite Belt. An example with regard to plant uptake is given in Kalis et al. (2007), who described a procedure for assessing metal uptake by *Lolium perenne.* To this purpose they used a four-step approach, starting with the total metal content in soil, including the calculation of the concentration in the pore water, the metal concentration adsorbed to the root surface, the metal contents in the roots and the metal contents in the shoots.

In Hodson et al. (Chapter 16 of this book) a detailed description of bioavailability is given.

1.5.4 Reliability

1.5.4.1 Uncertainties and Variability

It is generally acknowledged that Risk Assessment, although it is said to be an objective process, is also an unreliable process (e.g., Ferguson et al. 1998). There are several reasons for this. First, Risk Assessment includes many parameters and equations that have large uncertainties and variability. Uncertainty is the variation in these Risk Assessment tools due to lack of knowledge or to lack of scientific consensus. Variability is the variation due to spatial and temporal variations. The large variability is explained by the heterogeneous nature of soil and the large differences in human characteristics and behaviour among individuals. The large uncertainties, often found in exact sciences, are mostly related to the transfer of contaminants from soil into contact media, and biokinetic fate and transport processes in the human body (Human Health Risk Assessment and Food Safety), the immensely complex functioning of the soil ecosystem with mutual interactions between many different organisms and the soil properties (Ecological Risk Assessment), and transport processes in soil and aquifers (Groundwater-related Risk Assessment). Every variability found in the Risk Assessment tools is interwoven with uncertainty, while uncertainty does not necessarily go together with variability. Examples of equations that are characterised by both large uncertainties and variability are transport processes of contaminants through the aquifer, pore water and soil gas, and the equations that describe bioavailability in soil. A parameter that mainly has large variability is, for example, the fraction of total vegetable consumption that humans grow in their own garden. For a specific case this fraction can be accurately estimated, but for the use of a generic value for the derivation of Soil Quality Standards there is a huge variation between sites, and most definitely for bigger geographical entities. Generally speaking, patterns that describe human behaviour, and the behaviour and composition of soil ecosystems show a wide variation in time and space.

Second, Risk Assessment includes a whole chain of calculations and measurements, which means that small uncertainties in an earlier step (e.g., in the sampling strategy) might add up to large uncertainties in the final step (e.g., the risk characterisation). Third, several elements in Risk Assessment require a subjective judgment, which means that quite a number of uncertainties are involved with the sometimes arbitrary choices of the risk assessor.

It must be realised that measurements, although the general belief is that these are much more accurate than calculations, are also often characterised by limited reliability. Nevertheless, in some specific cases, the reliability could indeed be improved by including measurements in the calculations, namely, measurements of 'supportive input parameters' or the concentrations in contact media. By measuring the organic matter content of soil (a 'supportive input parameter'), for example, clearly the reliability of a site-specific Risk Assessment can be improved, as compared to an assessment based on an average organic matter content of a specific region (elimination of variability).

A specific type of uncertainty relates to the lack of clear definitions of political boundary conditions or the wrong interpretation of these political boundary conditions by scientists. When, for example, the degree of precaution (e.g., whether the average human being or the great majority of human beings must be protected) has not been clearly defined, or is incorrectly interpreted, the input parameter identification of exposure parameters could take on an arbitrary character.

1.5.4.2 Dealing with Uncertainties and Variability

As was mentioned in Section 1.5.4.1, each input parameter for Human health, Ecological, and Groundwater-related Risk Assessment is characterised by uncertainty and variability. Nevertheless for many Risk Assessment applications it is useful to represent the input parameter by one single value. Options that are mostly used, depending on the type of Risk Assessment, the purpose of the Risk Assessment, and the possible political boundary conditions, are based on a single value of the *central tendency* or some kind of worst-case estimate. Most often, a single value for the central tendency is the mean (or average) value or the median. Generally speaking, for normally distributed data the arithmetic mean is appropriate, when for non-normal data the medium value usually is the best representative of the central tendency. The worst-case estimate is mostly based on a specific percentile (usually 80th, 90th, or 95th percentile), or on the highest value found in a series of data. Although the choice for a specific percentile is also subjective, the use of a percentile is preferred over the use of an arbitrary high value. In many Risk Assessments, no specific choice for the level of precaution is made; instead, rather arbitrary values are selected on the basis of available data in the literature.

Most outputs from site-specific Risk Assessment, such as calculated human exposure or the number of ecological species affected, must be regarded as indications of truth values. Nevertheless, Risk Assessment is an extremely useful tool, as long as it is smartly used.

First, outputs from Risk Assessments can always be safely used for comparison of risks (*comparative Risk Assessment,* aka: *relative Risk Assessment*), for example, for priority setting. Higher exposure, for example, generally means a higher risk; or to put it even better, a higher *Risk Index* generally means a higher risk. Second, Risk Assessments based on worst-case assumptions can be used in a first step of a Risk Assessment procedure. Generally speaking, this implies that when there is no unacceptable risk, even under these worst-case conditions, it is relatively safe to state that unacceptable risks to human health, the ecosystem, the groundwater or agricultural products are very unlikely. The risk assessor, however, needs to be alert to the fact that the worst-case conditions indeed apply to the specific site. Imagine, for example, that a worst-case Risk Assessment is performed, for the purpose of investigating whether people in a specific residential setting might experience unacceptable risks, by using upper limit estimates for the crucial exposure parameters. If the result from this Risk Assessment under these worst-case conditions shows a Risk Index that does not exceed a value of 1, an unacceptable risk is very unlikely for normal conditions at the site. However, when inhabitants at the residential site, now or in the future, grow a much larger percentage of their vegetables on the site (large gardens), unacceptable risks cannot be excluded.

Another possible pitfall is that the boundary condition 'based on worst-case assumptions' is a subjective criterion, which is difficult to motivate and communicate. The level of conservatism is rarely concretised in Risk Assessments or, at best, at the level of subjective terminology such as 'based on worst-case assumptions'. Moreover, risk assessors sometimes might feel the urge to protect themselves from false negatives (the assumption that there is no unacceptable risk, when in reality there is one) which might lead to an unnecessary over-conservatism.

Scientists and regulators usually are looking for a balance between 'to be sure to be on the safe side', and realism and pragmatism. For this purpose the term 'realistic worst case' is often used, although this still is a subjective criterion. The use of a specific percentile, for example, the 90th percentile of each input parameter representing worst-case conditions, is a more objective criterion. However, the selection of this percentile is also a very subjective process. Choices for specific percentiles (usually 80th, 90th, or 95th percentiles) are often mentioned in Risk Assessments, but are seldom explained.

For more ambitious applications, the risk assessor needs to be aware of the sensitivities and uncertainties that are involved in the Risk Assessment tools. An experienced risk assessor needs to use insight when it comes to the most sensitive input parameters. A *sensitivity analysis* and *uncertainty analyses* can help to systematically identify the most sensitive input parameters, model equations, etc. The risk assessor also needs to be aware of the limitations of the outputs from Risk Assessment and check these against the purpose of the Risk Assessment. When the uncertainties are too great, the performance of additional assessments will be necessary, or the power of the results of the Risk Assessment will have to be adapted to more modest conclusions, that is, by communicating the restrictions and uncertainties.

A relatively simple, though quite time-consuming way of dealing with the lack of reliability, is to follow a *probabilistic* instead of a *deterministic approach*. A deterministic approach, based on point estimates in input parameters and resulting in a single value, does not give any information about the variation in that value. Moreover, since information about the lack of variation is lacking, stakeholders might get a misleading idea about the accuracy involved. In a probabilistic approach, input parameter point estimates are replaced by probability density functions, for at least the most sensitive input parameters. The most popular probability functions are normal, lognormal, cumulative and uniform distributions. Several software packages are available, for example, Crystal Ball, to determine the probability density functions from a series of data.

The procedure for the determination of probability density functions depends on the purpose of the Risk Assessment. For 'generic Risk Assessment purposes', for example, the derivation of Soil Quality Standards, the most effective way is to incorporate both uncertainty and variability in the probability density functions. For site-specific applications, however, most of the variability could be eliminated by measurements, so that the probability density functions mainly cover uncertainty. The most popular way of performing a probabilistic Risk Assessment is based on Monte Carlo techniques (e.g., Seuntjes (2004), who assessed the risk of the leaching of cadmium from soil, originating from the former presence of non-ferrous industries, into the groundwater in Lommel, Belgium). Burmaster and Anderson (1994) described 14 principles of good practice to assist people in performing and reviewing probabilistic or Monte Carlo Human Health and Ecological Risk Assessment.

The result of a probabilistic Risk Assessment is a probability density function of an important measure for risks, for example, human exposure or percentage of soil organism affected, or of a Risk Index. The huge advantage of such a probabilistic procedure is that the impact of uncertainties and variability is made transparent in the resulting risk appraisal. However, a choice needs to be made for the level of acceptability, in terms of a specific percentile of the probability density function as output of the Risk Assessment. Although this offers a more sophisticated way of dealing with acceptable risks, there are no objective criteria to underpin this choice.

Since Risk Assessment is a relatively unreliable process, it is of the utmost importance to describe each and every step taken, from the field survey on up to Risk Management solutions. This should be done in such a detailed way that the Risk Assessment is reproducible for third-party risk assessors. The report must explain which political boundary conditions are incorporated in the Risk Assessment. Furthermore, it should refer to all the Risk Assessment tools (including all input parameter values) that were used, along with associated references.

Because of the characteristically limited reliability involved with Risk Assessment, it is recommended to organise, at least for crucial reports, peer reviews and/or second opinions. Peer reviewers cannot eliminate the uncertainties, but they can judge whether risk assessors have made these uncertainties transparent and also, very importantly, whether the uncertainties rectify the conclusions. Several countries include peer review or second opinion procedures in their acts and laws. Alternatively, these procedures are often included in national guidance documents.

1.5.4.3 Validation

The lack of reliability of Risk Assessment results is supported by numerous validation, comparison and round-robin studies. It must be realised, however, that validated models hardly (if at all) exist (Leijnse and Hassanizadeh 1994). In fact, only model applications can be validated. The reason for this is that for each specific model application, different equations and input parameters are the most relevant. Therefore, in each specific model application a different part of the model is tested.

When a smart combination of model applications is validated, however, it at least ensures a level of confidence in the whole model. But since it is a subjective decision as to what kind and how many validated model applications are needed to cover the whole range of possible model applications, and the criterion for 'validated' for separate validations is quite vague, the term 'validated model' is better off being avoided.

A similar process that investigates the performance of models and procedures is verification. Verification focuses on the testing whether a predefined hypothesis is true.

1.6 Risk Management

1.6.1 Scope

Sometimes, a broad definition of *Risk Management* is followed, that is, Risk Management is the whole risk-based procedure for contaminated site management. According to this broad definition, Risk Assessment is considered as an important component of Risk Management. In this book, however, a more narrow definition of Risk Management is used (see Fig. 1.3) that focuses on the development of the strategies for dealing with the risks, only. From this perspective, the term Risk Management is more directly related to the dictionary definition of management, which includes active words such as 'handling' and 'controlling', generally with the purpose of bringing contaminated sites back into beneficial use.

Risk Management is appropriate when the conclusion from a Risk Assessment is that a particular risk is unacceptable. It includes avoiding the risks, mitigating or removing risks and, last but not least, communication about the risks with the parties involved. The keyword in Risk Management is *risk reduction*. There are many ways to achieve risk reduction. Basically, Risk Management relates to removal or controlling of the source, that is, *source control treatment*, or to blocking the pathway from source to receptor. The challenge is to find the optimum balance between the most effective and most cost-efficient way of doing this by weighing the short-term advantages against the costs of aftercare.

Remediation (aka: restoration, or clean up), that is in its most strict definition elimination of the source and the resultant soil contamination, is the most direct way of risk reduction. However, remediation often is too drastic an activity, whose results are not in alliance with the social and technical impact at the site and the costs. Alternatively, source control or the application of barriers, that is, a process which eliminates or blocks the source, might be sufficient. In some cases, compliance with policies requires more stringent measures than are absolutely necessary from a risk perspective.

Communicating with all stakeholders is necessary to find the optimal end goal of Risk Management and to define the procedure for how to achieve this. Often, an intensive negotiating process is needed in which decision-makers play an important role.

1.6.2 The Source

The term ‘source’ could do with some further attention. At any contaminated site, a primary source that is responsible for ongoing contamination of the upper soil layer, e.g., a leaking pipeline, an oil spill, waste materials stored on the surface of the soil, must be fully eliminated, when possible. In case of ongoing atmospheric deposition, elimination of the source often is a long-term political process addressing the responsible parties for immissions of contaminants, and it is not always possible. Other sources might be part of agricultural practices, such as the application of fertilizers and pesticides. In that case, an acceptable soil quality would constitute a harsh boundary condition in agricultural soil management.

Contamination of the upper soil layer, in addition to being a potential cause of risk to human beings, the soil ecosystem and to Food Safety, is a source for groundwater contamination. This might lead to the necessity of removal or control of contaminants in the upper soil layer for the purpose of protecting the groundwater.

1.6.3 Procedures

In the late 1970s, Risk Management was often the same thing as complete removal of the contaminants and, hence, of the risks involved. Harsh remediation measures, such as *Dig-and-Dump* (remediation of the upper soil) and *Pump-and-Treat* (remediation of the groundwater) were the most popular mechanisms to achieve this goal. Alternatively, insulation of the contaminants, and hence of the risks involved, was used as a less strict but cheaper solution. Since the early 1990s, the general focus of Risk Management has evolved into the elimination of *unacceptable risks*, which does not necessarily mean complete removal of the contaminants. Today, the remediation objective is often set at a concentration where the risks for human health, the soil ecosystem, the groundwater and/or Food Safety relate to an *acceptable risk level*.

Moreover, the weighing of the end goal of remediation against necessary costs has evolved into the common way of performing Risk Management.

The most simple and generally least expensive solution for contaminated site problems relates to changing the land use, or adapting the layout of the site within the same land use, in terms of blocking the major exposure pathways. An example of change of land use is using cadmium contaminated sites at the border of a municipality for city expansion, which does not allow substantial vegetable production, instead of using it for vegetable gardens or for agricultural purposes. In this way human exposure through vegetable consumption is reduced or eliminated. An example of changing the layout of a site within the same land use is given by a lead contaminated site with a heterogeneous contamination pattern. The human health risks can be substantially reduced when the buildings are situated on the locations with the highest lead contents and the bare surfaces (garden and borders) on the locations with the lowest lead concentrations. In this way exposure of children to lead through soil ingestion is avoided or reduced. A popular option, mainly efficient for immobile contaminants, is covering contaminated hotspots with pavement, grass or any other vegetation, also reducing the possibilities for hand-mouth contact and, hence, exposure through soil ingestion. Another example is found in Arienzo et al. (2004), who revegitated a soil at a former ferrous metallurgical plant in Naples, Italy, for the purpose (among others) of preventing dispersion of metal-contaminated particles by water or wind erosion. Fencing off highly contaminated parts of a contaminated site, as, for example, described in Louekari et al. (2004), for the purpose of avoiding practically any lead exposure near a former lead smelter in Finland, would be a good example of rather drastic measures in regard to adapting the site use.

The disadvantage of changing land use or the layout of the site with the same land use is that concessions often have to be made in regard to the ideal way the site is used.

Moreover, risks for other protection targets should also be investigated. Therefore, this solution often offers limited possibilities.

1.6.4 Remediation Technologies

1.6.4.1 Scope

Remediation is a hard-to-protocollise activity. It is often not feasible to follow a cookbook-type recipe for the design of a remediation plan. The reasons for this are that for every combination of contaminant, site, soil properties and land use, a different remediation technology may be appropriate. Moreover, the execution of one specific remediation technology can be carried out in many different ways. Therefore, the development of the remediation plan typically must be done on a site-by-site basis. Remediation experts often lobby against rigid remediation plans. Instead, they would prefer a remediation approach in which the proceedings develop during the remediation activities.

The basic distinction in remediation technologies is in situ (at the site) and ex situ (off the site) technologies. In situ technologies, mainly applicable to organic contaminants, have the advantage that no transport of soil material is needed. The huge advantage of ex situ technologies is that the physical-chemical treatment of soil is generally more efficient in a factory than on site. In Fig. 1.8, an illustration of an excavation in Bilthoven, the Netherlands is given, as an example of an ex situ remediation.

Bardos et al. (Chapter 20 of this book) give a detailed description of innovative, sustainable remediation technologies.

1.6.4.2 In Situ Remediation Technologies

The US Environmental Protection Agency includes 12 different in situ remediation technologies in their Annual Status report on contaminated sites treatment

Fig. 1.8 An illustration of an excavation in Bilthoven, the Netherlands, as an example of an ex situ remediation (photo: K. Versluijs; reproduced with permission)

technologies (US EPA 2007). These remediation technologies are Bioremediation, Chemical treatment, Electrical separation, Flushing, Multi-phase extraction, Mechanical soil aeration, Neutralization, Phytoremediation, Soil vapour extraction, Solidification/Stabilization, Thermal treatment and Vitrification.

Phytoremediation, using hyper-accumulators to extract contaminants from soil, focuses on metal elimination from the soil (e.g., Vassilev et al. (2004), who gave an overview of the use of plants for the remediation of metal-contaminated soils, including site decontamination (phytoextraction), stabilization techniques (phytostabilisation), and the use of soil amendments to enhance (in case of phytoextraction) or reduce (in case of phytostabilisation) mobilization of metals). Generally, this is a slow remediation technology. An extensive root proliferation increases metal uptake. At too-high metal concentrations, phytotoxicological effects might hamper an efficient uptake. A relatively efficient plant for Phytoremediation is Brassicaceae, which has a high metal uptake affinity and a relatively high tolerance to metals. Brassicaceae was used, for example, by Kidd and Monterroso (2005) for the purpose of extracting metals from mine-soil material in Spain. Robinson et al. (2000) demonstrated the possibilities of willow (Tangoio) and poplar (Beaupré) clones for phytoremediation of cadmium-contaminated sites.

Electrical separation is based on an electric field in the soil between inserted electrodes, which forces the migration of pore water or groundwater, including metals and organic contaminants. It is a relatively new technology and, therefore, still in the experimental stage, for which further development is necessary. It might, however, be an alternative for the remediation of clayey soils when 'Pump-and-Treat' methodologies are not efficient. Amrate et al. (2005), for example, demonstrated a successful migration of lead in a highly contaminated soil near a battery plant in Algiers, Algeria, where EDTA was added to enhance lead transport.

A spectacular extensive remediation technology that has gained enormous popularity since the mid 1990s is based on biodegradation of organic contaminants and dilution, and is often called *Natural Attenuation*. Indigenous or cultured organisms can be used for biodegradation. In spite of the sometimes high starting costs, the overall budget for this Risk Management procedure is generally low. In addition, it allows for a minimal disturbance of the natural conditions in the soil or groundwater, and there are limited engineering activities needed at the site. Moreover, it even offers opportunities for difficult sites with clayey soils and difficult contaminants such as chlorinated hydrocarbons, even under anaerobe conditions. Natural Attenuation is often combined with ex situ remediation techniques, such as removal of the source. The adage is: use the natural self-cleaning capacities of the soil as much as possible, stimulate natural conditions when necessary and use ex situ remediation technologies only when strictly needed. The success of Natural Attenuation depends primarily on the type of organic contaminant and the performance of the soil ecosystem. The latter depends on the organisms present. Zytner et al. (2006), for example, demonstrated the important contribution of fungal metabolism for the degradation of branched hydrocarbons. For this reason the chemical and physical characteristics of the soil and the artificial oxygen and nutrient supply are dominant factors.

In situ Bioremediation of organic contaminants is especially difficult in low permeability soils. Athmer (2004) described a procedure for integrating electro kinetics with in situ treatment for the remediation of TCE (trichloroethylene) contaminated clay soils in Paducah, Kentucky, USA, to address this problem. It generated a uniform migration of trichloroethylene through the soil to treatment zones.

Peter et al. (Chapter 22 of this book) give a detailed description of Natural Attenuation and of its practical possibilities.

As specific applications, 'bio-screens' are used, that is, zones with an active, often stimulated, degradation at strategic positions in the soil system, or Funnel-and-Gates techniques, in which contaminants are led to zones with an active degradation.

Several materials have been proven to be effective in *Solidification/Stabilization* (aka: *immobilization*, or fixation) of heavy metals in soils. A proven method to fixate metals in soils is mixing the soil with lime (liming) or cement. Yukselen and Alpaslan (2001), for example, successfully immobilized copper, and iron in soils in an old mining and smelting area located along the Mediterranean coast in northern Cyprus. They showed that an additive/soil ratio of 1/15 (on mass basis) resulted in the optimal immobilization, for both lime and cement. This ratio very much depends, of course, on the soil type and soil properties, mainly pH. Tlustoš et al.

(2006) demonstrated that by the addition of lime and limestone to a contaminated Cambisol with 7 mg/kg cadmium, 2,174 mg/kg lead and 270 mg/kg zinc, the mobile (0.01 mol/l $CaCl_2$ extractable) fractions dropped by 50, 20 and 80% for cadmium, lead and zinc, respectively. The pH was increased from 5.7 to 7.3. Consequently, the metal concentrations in straw and grains of wheat were significantly reduced.

Száková et al. (2007) found substantial differences in reduction of the mobile (0.01 mol/L aqueous $CaCl_2$) fractions of metals when applying lime, limestone, and zeolite to contaminated soils. However, although the mobile fraction of cadmium and zinc indeed decreased, the mobile fraction of lead was hardly affected and the mobile fraction of arsenic even increased in some of the treated spots. The availability of arsenic was more affected by different characteristics of experimental soils than by individual soil amendments. Moon et al. (2004) contributed the fixation of arsenic to inclusion of arsenic in pozzolani cement reaction products and the formation of calcium-arsenic precipitates.

Ameliorating soil materials can be of natural origin, such as clay or bauxite residue. Alternatively, several by-products of production processes are used for this purpose. Red mud, a by-product of the aluminium industry, for example, has been identified as an effective amendment for in situ fixation of heavy metals in soil because of the high content of Fe and Al oxides (Zhang et al. 2002). Friesl et al. (2003) demonstrated the efficiency of the amendment of red mud (10 g/kg) in four soils, in the vicinity of a former Pb-Zn smelter in Austria, highly polluted with (among others) Zn (2,713 mg/kg) and Cd (19.7 mg/kg). This resulted in the reduction of metal extractability of 70% for Cd and 89% for Zn.

Other cements used as fixation material are sulfoaluminate cement, powdered activated carbon, quick lime (Guha et al. 2006), and ferrous sulphate (Warren et al. 2003). These authors showed that accelerated carbonated treatment substantially reduces the availability and, hence, the risks, of mercury in soil.

One disadvantage of the amendment of immobilizing soil materials may be the presence of other contaminants, which implies that the immobilization of specific metals is accompanied by the introduction of other contaminants. Red mud, for example, includes arsenic, chromium and vanadium (Friesl et al. 2003). Therefore, the optimal application of immobilizing materials to soils requires the optimum balance between an effective binding of metals and minimizing the negative effects of other contaminants. Friesl et al. (2004), for example showed that, at a red mud addition of more than 5% of total soil weight, the disadvantages of introducing other contaminants exceeds the advantage of fixation of metals.

Grotenhuis and Rijnaarts (Chapter 21 of this book) give a detailed description of in situ remediation technologies.

1.6.4.3 Ex Situ Remediation Technologies

The US Environmental Protection Agency includes 14 different ex situ remediation technologies in their Annual Status report on contaminated sites treatment technologies (US EPA 2007); these are Bioremediation, Chemical treatment, Incineration, Mechanical soil aeration, Neutralisation, Open burn/Open detonation, Physical separation, Phytoremediation, Soil vapour extraction, Soil washing, Solification/Stabilization, Solvent extraction, Thermal desorption, and Vitrification.

1.6.4.4 Barriers

Barriers are used to isolate contaminants at contaminated sites from the surroundings and, hence, to protect any protection targets in the surroundings of the contaminated site. Compacted soil materials are recognized liners. Clay, or clayey soil material, is the most obvious natural barrier material. They both have a low hydraulic permeability and a chemical buffering capacity through adsorption. Kabir and Taha (2004), for example, demonstrated an effective barrier function of compacted sedimentary granite residual soil material for the isolation of contaminants in landfills. They showed that this material has a hydraulic conductivity lower than the suggested limit (1×10^{-7} cm/s) of the various waste regulatory agencies in the USA. In addition, it has adequate strength for stability, and exhibits small shrinkage potential upon drying. Qian et al. (2002) specified the requirements of soil materials as effective barriers, in terms of contribution of silt and clay, plasticity, and limitations to the contribution of gravel-size materials and chunks of rock.

Several waste materials such as fly ash could be used a barrier material. Sivapullaiah and Lakshmikantha (2004), for example, demonstrated that the addition of bentonite to fly ash improves the chemical buffering function and the geotechnical properties of the barrier. Kaolonite and bentonite (a commercially available high swelling clay) are artificial alternatives for barrier materials.

1.6.5 Ecological Recovery

A problem with many remediations, especially ex situ remediation measures, is that the soil ecosystem, vegetation and above-ground fauna generally are negatively impacted, at least on the short term. In case of Dig-and-Dump technologies, the habitats, the organisms and the seed pool are removed from the site. Ex situ thermical treatment results in total elimination of soil organisms and organic matter. Especially when a remediation is triggered because of unacceptable ecological risks, the question is relevant if the ecological benefits on the longer term counteract the negative impact on the short term. The possibilities and timeframe for ecological recovery strongly depend on the type of soil that is applied in the final stage of the remediation. Especially clayey soil material high in organic matter speeds up the recovery process. The application of comparable soil as the wider environment, however, improves the development of a regionally appropriate ecosystem. The effects on above-ground fauna can be reduced through a stepwise remediation procedure, in which in different stages only a part of the site is remediated, so that recolonisation of organisms in the 'new soil material' can take place from the parts that have not yet been remediated.

1.6.6 Remediation Objectives

An end goal of a Risk Management procedure must be defined, and generally expressed as soil or groundwater concentration. Although the term *Risk Management objective* would have been a more appropriate term, this intended soil or groundwater concentration generally is called *remediation objective* (aka: *remediation goal*).

In the early days of contaminated site management, in the late 1970s, the remediation objectives were commonly set at the zero level, often not supported by any explicit considerations. Today, the goal of Risk Management generally relates to an *acceptable risk level* for the relevant protection targets. The selection of protection targets and the definition of an acceptable risk level are beyond the scope of science and are the responsibility of decision-makers. Contrary to the appraisal of existing soil contamination, which relates to imposed risks, Risk Management is supposed to 'create' a desirable situation, versus certain efforts and costs. Therefore, there are good reasons to select more protection targets and more stringent protection levels for the objectives of Risk Management than for curative decisions on existing soil quality.

Examples of *acceptable risk levels* are the Negligible Risk (NR) for human health as a target for the soil upper layer, or the Negligible Risk (NR) for the aquatic ecosystem as a target for the groundwater. Another option for a remediation target that is not based on risks is the (natural) background concentration (as a target for the upper soil or the groundwater), or commercial production criteria as a target for agricultural products.

The process of deriving remediation objectives includes the following steps:

- Selection of protection targets.
- Definition of 'policy requirements' for each protection target (e.g., 'it must be possible to grow the complete vegetable package of a family in a vegetable garden'; or 'the soil ecosystem must be fully protected in a nature reserve').
- Translating the 'policy requirements' into Risk Assessment terms (e.g., in analogy with the examples above, 'exposure through the complete vegetable consumption from the own garden equals the Reference Dose for exposure'; or: '95% of the soil ecosystem must be protected (in that case, an affected fraction of 5% is assumed as "full protection")'.
- Derivation of risk limits in soil or groundwater for every protection target, and for all selected contaminants.
- Selection of the appropriate risk limit in soil or groundwater as a remediation objective (usually the lowest of all risk limits in soil or groundwater) for all selected contaminants.

Except for soil concentrations, alternative types of remediation objectives could be defined. Von Lindern et al. (2003a), for example, used the lead concentration in house dust as a goal for the remediation of the Bunker Hill Superfund site in northern Idaho, USA, since exposure through dust ingestion has been recognized as a principal exposure pathway. Von Lindern et al. (2003b) focused on lead blood levels as a remediation objective of the Bunker Hill Superfund site.

In some cases there are good reasons for focusing on lower soil concentration levels as remediation objectives than is strictly needed for human health protection. Several remediation technologies, such as Dig-and-Dump for example, do not always allow for 'gradual' risk levels after remediation, but may result in a clean soil.

1.7 A Closer Look into Risk Assessment

1.7.1 Types of Risk Assessment

1.7.1.1 Purpose

Generally speaking, contaminated site Risk Assessment offers two possibilities. First, Risk Assessment can be used to investigate a specific site. This type of Risk Assessment is called *site-specific Risk Assessment* or *actual Risk Assessment*. In this case, information about the specific site is available. Second, Risk Assessment can be used to derive Soil Quality Standards. This type of Risk Assessment is often called *potential Risk Assessment* or *generic Risk Assessment*. Often, generic Risk Assessment is the first step in Risk Assessment frameworks, followed by site-specific Risk Assessment when the generic Risk Assessment does not result in a clear decision as to risks.

1.7.1.2 Site-Specific Risk Assessment

Risk Assessment related to a specific site is often called *actual Risk Assessment*. From this perspective, 'actual' is used in the sense of 'existing in fact' and not necessarily in the sense of 'existing at this moment'. It is possible, for example, that a Risk Assessment might be performed for the purpose of investigating whether it is 'safe' to reside at a specific residential site, which might be contaminated. In that case, the relevant time frame for Risk Assessment can vary from several years up to several decades, depending on the time frame over which the specific contaminants reveal effects. Therefore, it does not make sense, for example, to focus on the actual layout of the garden, with or without vegetables grown for one's own consumption, since this layout may change over a period of years or decades. In fact, an assumption needs to be made for a representative contribution of vegetables from one's own garden, independent of the situation at the time that the Risk Assessment is performed. Specifically, in situations in which a Risk Assessment is performed for the purpose of investigating the risks for a future land use, it does not make sense to base the Risk Assessment on features that relate to the present land use. For this reason, the term 'site-specific Risk Assessment' is used in this book, rather than 'actual Risk Assessment'.

Nevertheless, for site-specific Risk Assessment, relevant information might be available that facilitates the assessments of risks for human health, the soil ecosystem, the groundwater, or Food Safety. Although not all information present might be relevant, a huge asset is that measurements can be performed, which significantly improve the quality of the Risk Assessment, and reduce uncertainties (see Section 1.7.3).

1.7.1.3 Potential Risk Assessment

The term 'potential' is not to be missed in any list of vague terminology. Paradoxically, it also is a very useful term in contaminated site management. Literally, in the sense of 'possible, when certain conditions apply', there is a potential unacceptable risk at any site where contaminants have been measured by definition, independent of the concentration. The 'certain conditions that apply' are, for example, an intensive contact of human beings with the soil with regard to human health risks, a relatively high bioavailability of the contaminants in soil with regard to Ecological Risk Assessment and Food Safety, or a low pH and the presence of metals with regard to Groundwater-related Risk Assessment.

A potential unacceptable risk also might refer to the fact that a site-specific Risk Assessment has resulted in the conclusion 'unacceptable risk, with a low level of reliability'. This could be the case, for example, when only a first step in a wider Risk Assessment framework has been performed, with conclusions based on the chemical analyses reports of a limited number of samples. However, since risk-based soil quality assessment is characterised by substantial uncertainties in general, the adjective 'potential' in the meaning of 'conditional' could practically always be added. The only benefits of this use of the term 'potential' would be to stress the lack of reliability to the stakeholders. This function increases the level of confusion rather than supporting the Risk Assessment and the Risk Communication. Therefore, 'potential' should not be used for the purpose of alerting those involved to the limited reliability of a Risk Assessment.

A more appealing use of the phrase *potential Risk Assessment*, in this context also referred to as *generic* Risk Assessment, is related to the derivation and use of Soil Quality Standards. Since Soil Quality Standards are not focused on a specific site, but rather relate to a whole series of unknown contaminated sites, these Soil Quality Standards must be derived from generic scenarios.

1.7.2 Soil Quality Standards

Soil Quality Standards (aka: (soil) Guideline Values, (soil) Screening Values, or Target Levels) are generic values enabling a distinction into two classes for which the measured concentrations in soil are either higher or lower than the Soil Quality Standard. They can be considered as the core of contaminated site management. In the early days of contaminated site management, a list with Soil Quality Standards was about the only appraisal framework available, and often used for the separation between acceptable and unacceptable cases of soil contamination. Since the late 1980s, *risk-based* Soil Quality Standards have been derived in several developed countries.

Several types of Soil Quality Standard exist, for different purposes. Carlon and Swartjes (2007b) distinguished three classes of Soil Quality Standards. The first class of Soil Quality Standards, with the most stringent values, represents the upper limit for long-term sustainable soil quality, appropriate for prevention purposes or as remediation objectives. The second class, with the highest values, triggers actions such as either a more detailed Risk Assessment or Risk Management actions (e.g., remediation) when exceeded, and are used for curative purposes, that is, for supporting the risk appraisal for existing contaminated sites. The third class is an intermediate class, and supports further research actions such as the performance of more detailed soil sampling.

As was mentioned in Section 1.7.1.3, Soil Quality Standards are applied to a whole series of contaminated sites. Therefore, a generic exposure scenario needs to be defined for a hypothetical site. Generally speaking, such a generic scenario either relates to standard assumptions, as for frequently found contaminated sites, or to conservative assumptions. The latter must certainly be the case when 'false negatives' (the incorrect assumption that there is no unacceptable risk) get a higher political negative weight than 'false positives' (the incorrect assumption that there is an unacceptable risk). Also in a case where Soil Quality Standards are used as a trigger for possible site-specific Risk Assessments, generic scenarios as basis for the Soil Quality Standards need to be based on conservative assumptions.

A variation on generic Soil Quality Standards relates to 'land use-specific Soil Quality Standards'. As the term says, it refers to several Soil Quality Standards for different land uses, for each specific contaminant. One advantage of human health-based land use-specific Soil Quality Standards is that more realistic exposure scenarios for the respective land uses can be used. An advantage of ecologically based land use-specific Soil Quality Standards is that a more appropriate level of ecological protection can be chosen for the respective land uses. The disadvantage of land use-specific Soil Quality Standards is that the derivation process is much more intensive, since a series of Soil Quality Standards must be derived for each contaminant. The use of land use-specific values in practice is less convenient, since a choice needs to be made for each site as to which land use is appropriate. Moreover, the application of land use-specific Soil Quality Standards may give a misleading idea of accuracy.

Examples of human health-based Soil Quality Standards are given in Hristov et al. (2005) for *Human Health Soil Screening Levels (CHHSSL)* in California, USA, and in DEFRA and EA (2002) for *Soil Guideline Values* for metals in the UK. Examples of ecologically based Soil Quality Standards are given in Canadian Council of Ministers of the Environment (1999) for Canada, and in National Environmental Protection Council (2003) for Australia. An example of combined (human health and ecological based) Soil Quality Standards is given in Ministry of VROM (2008) for the Netherlands.

Worldwide the Soil Quality Standards used differ to a large extent. This is partly due to differences in the purpose of the Soil Quality Standards, but the technical frameworks also show many differences. Provoost et al. (2006) compared Soil Quality Standards for eight metals and metalloids, from Canada, Flanders (Belgium), France, Germany, the UK, the Netherlands, Norway, Sweden, Switzerland and the USA. For most contaminants they found differences between the highest and lowest value of more than a factor of 1000. They concluded that some of these differences could be explained by political differences, such as the choice of protection targets and risk levels. Some of the other differences between Soil Quality Standards, however, are explained by technical/scientific differences between the procedures used in the different countries. Swartjes and Carlon (2007) came to similar conclusions for Soil Quality Standards used in 16 European countries. They found even higher differences for organic contaminants. Swartjes (2007) concluded that differences between seven European exposure models, important instruments in the derivation of human health-based Soil Quality Standards, can result in widely different risk appraisals for the same exposure scenarios, especially for contaminants that are mobile and even more for contaminants that are volatile. Therefore, the need for a higher consistency of Risk Assessment tools is acknowledged in Europe (Swartjes et al. 2009).

1.7.3 Measurements

Measurements in contact media can significantly improve the quality of a site-specific Risk Assessment. The kind of measurements that are possible and the benefits of these measurements vary. In this section, a general view concerning measurements in contact media will be given. In the introductionary chapters on Human Health Risk Assessment (see Chapter 6 by Swartjes and Cornelis, this book), Ecological Risk Assessment (see Chapter 13 by Swartjes et al., this book) and Groundwater-related Risk Assessment (see Chapter 17 by Swartjes and Grima, this book) the most important measurements are described in more detail.

First, *direct input parameters* could be measured, such as the concentration in human blood or body tissue (Human health Risk Assessment), or the number of earthworms in soil (Ecological Risk Assessment). Second, *basic input parameters* could be measured, such as the concentrations in soil compartments, or in contact media. Third, s*upportive input parameters* could be measured, such as soil properties or input parameters that relate to long-time human behaviour, if relevant for the site over the relevant time span.

Measuring of supportive, basic or direct input parameters have both advantages and disadvantages. Generally speaking, the reliability of the assessment of the *actual* Risk Assessment improves when more supportive measurements are available, even more so when basic input parameters are measured and the most often when direct measurements are performed. The disadvantage, however, is that measurements are expensive and generally more expensive in the order of supportive, basic and direct input parameters, and, therefore, not always suited for routine Risk Assessment.

Moreover, the general belief that measuring gives a more reliable value than a calculation is often true, but not always. In some cases, measuring a representative value is extremely difficult for technical reasons and due to spatial and/or temporal variation. Moreover, measurements are strongly related to the *present* situation and may not represent the long-term conditions for which it is stated, for example, that 'the site is safe to live on'. The risk assessor needs to find a smart balance between the determination of a more reliable measured input parameter that might be less representative for the long-term risk, or a less reliable calculated input parameter that does represent long-term site conditions. Obviously, the risk assessor also has to account for the additional costs of measurements when formulating the involvement of measurements.

An example of such a dilemma is the Human Health Risk Assessment for a well-maintained (manuring, liming) vegetable garden. The risk assessor has to decide if the Risk Assessment benefits from measured concentrations in the vegetables present at the site, which are relatively accurate for the present situation, but may give underestimations for future situations in which another owner neglects soil liming. Alternatively, the risk assessor could perform relatively unreliable calculations of the concentrations of a representative combination of vegetables of choice, on the basis of total soil concentrations and soil properties belonging to an appropriate, that is, average or neglected liming conditions.

Generally speaking, measurements of direct input parameters are most valuable for Risk Assessments that relate to, let's say, the first few years (maybe one to three years), while measurements of basic input parameters are useful for the subsequent few years, see Fig. 1.9. As illustrated in this figure, supportive input parameters often represent time spans up to decades.

This picture only gives a general insight into the time spans for which measurements are useful, mainly for the purpose of illustrating the importance of the time frame when deciding on measurements. Direct measurements for Human Health Risk Assessment, such as measurements of the cadmium levels in blood, are often

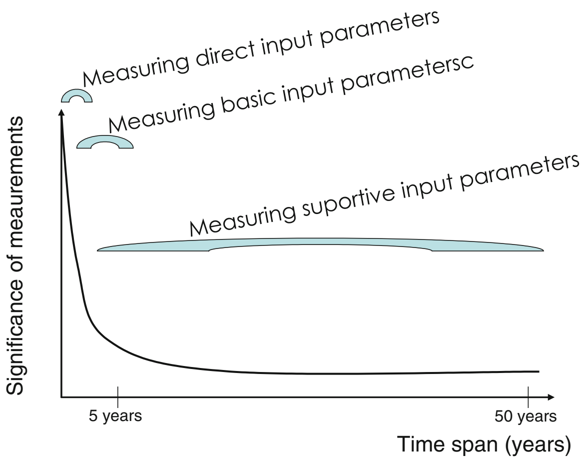

Fig. 1.9 Significance of measurements as a function of the time span for which the Risk Assessment applies (*straight line*). Indication of the time span, for which different type of measurements are useful (*dark shaded arches*)

useful for short-term risk appraisal. Measurements of cadmium in urine, however, represent a measure of cumulative lifelong exposure. Since this represents the site exposure history, is it defensible to assume this measure also is representative for long-term future exposure, when no changes in exposure conditions are expected. And with regard to supportive measurements, the representative time span of clay content (decades to centuries) and pH (1–10 year), for example, largely differ.

1.7.4 Laboratory Data Versus Field Data

Appropriate Risk Assessment is all about what is happening in the real world, that is, at contaminated sites. However, Risk Assessment would hardly be possible without the support of laboratory experiments. These experiments are used for at least three different purposes. First, for ethical and technical reasons, direct human toxicological effect data are rarely available. These data need to be derived from laboratory experiments with animals, following strict guidelines with regard to laboratory animal welfare. Also ecological effect data heavily depend on laboratory experiments, since it is difficult to investigate effects on one specific species in the field. Second, it is inconvenient to control standard environmental conditions in the field.

As a consequence, Human Health and Ecological Risk Assessment both strongly depend on experimental laboratory data. To transfer the laboratory effect data to human health, and ecological effect data and to real world applications (field conditions), the laboratory effect data are divided by *assessment factors* (often called extrapolation factors). These assessment factors, which are often in the range of 10 to values as high as 10,000, cover intraspecies and interspecies differences and, last but not least, uncertainties associated with laboratory studies, which always cover a limited amount of tests, for a limited time span. A specific case of intraspecies differences in Human Health Risk Assessment is the differing sensitivity within the human population. In this case, an assessment factor could be applied, when politically feasible, to also protect the most sensitive part of the human population. In case of effects data for a whole ecosystem, instead of a single species, uncertainties due to interspecies variation are dominant. In fact, a major problem in Ecological Risk Assessment is the extrapolation of observations from individual and population levels to the ecosystem level (Eijsackers et al. 2008). Again, assessment factors can be used to cover the corresponding uncertainties.

A second important use of laboratory studies relates to the assessment of input parameters. These input parameters may include 'supportive parameters' such as physico-chemical contaminant characteristics (vapour pressure, octanol-water partition coefficient (Kow), water-saturated permeation coefficients, etc.). But laboratory measurements can also focus on 'basic' (more lumped) input parameters, such as indoor air concentrations, or leachate concentrations. Often, such measurements are prescribed in manuals, guidance documents or Decision Support Systems. Examples of this are the measurement of the concentration in vegetables in Tier 3 of the Dutch tiered approach used to determine the risk due to the vegetable consumption

(Swartjes et al. 2007), or the performance of bioassays in several tiers of the TRIAD approach, to determine the site specific ecological risks (see Chapter 15 by Rutgers and Jensen, this book).

Finally, a third purpose of laboratory studies is validation of models or testing of technologies (pilot studies). An example of a validation study is the comparison of calculated with measured indoor air concentrations for 11 petroleum hydrocarbons and chlorinated solvents sites in the USA and Canada (Hers et al. 2003).

In short, to enable Human Health and Ecological Risk Assessment, experiments are simply a necessity. In principal, laboratory experiments for validating and testing and, to a lesser extent, for assessing input parameters could be replaced by field studies. However, the adage that 'field studies are always better than laboratory studies' requires a few nuances. The choice for field versus laboratory studies simply depends on the trade-off between the control of conditions in the laboratory versus the degree of reality in the field. Of course, financial arguments also must be included in this choice. Often a multitude of experiments can be performed in the laboratory for the same budget as for one single field study. Another, non-scientific, aspect that is in favour of field experiments is that they better connect with public perception, but also even sometimes with scientific perception.

1.7.5 Expert Judgement

It is a major inconvenience when in a scientific discipline the possibilities for a quantitative analysis are lacking. In such a case, *expert judgement* may offer an alternative. Expert judgement is the process in which experts determine an opinion, (partly) based on 'gut feeling'. In an optimal expert-judgment process, several experts are involved, and opinions are based on consensus.

A situation, in which expert judgement can be used, for example, is when a Soil Quality Standard for a specific contaminant is lacking and effect data are not available for this contaminant. Via expert judgement a 'substitute contaminant' with similar physico-chemical contaminant characteristics can be selected, for which it is expected that effects, and hence the Soil Quality Standard, are in the same order of magnitude. Another example relates to the optimal balance between the use of a few available measured contaminant concentrations for a specific vegetable and of many less appropriate data for a non-edible crop, in terms of determining a representative concentration for a specific vegetable.

In practice, expert judgement can vary from the opinion forming of a single expert up to striving towards consensus within a group of appropriate experts. A decent expert judgement must be performed within a well conceived group, that often includes experts from different disciplines and, preferably, individuals who approach the issue from a different angle. In any case, the risk assessor needs to describe the expert judgement process and the decisions criteria that were used so that scientists, regulators, and other stakeholders can decide how to weigh the outcome of the expert judgement.

1.7.6 Essential Metals

Several contaminants that are found in soil, mainly metals, are essential for the functioning of human beings or organisms. Twenty-five naturally occurring elements are believed to have an essential function in plants and animals. As a consequence, humans and/or organisms need to take these contaminants up, for example in their diet. Among these *essential metals* are zinc, copper and selenium. It is interesting to note that about 2 billion people worldwide, mainly children and women, suffer from zinc deficiency, partly due to low zinc concentrations in soils in Africa, Latin America and in some parts of Asia (Prasad 1998; Ramakrishnan 2002). Most organisms have a narrow range between optimal concentrations (toxicity) and deficiency.

From the perspective of essentiality, it seems a paradox to name essential metals 'contaminants'. However, analogous to the definition used in this book in Section 1.3.1, that is, that the toxicity of chemicals depends on the dose at which humans or organisms are exposed, the period that this exposure takes place and the frequency of exposure and the form (speciation) in which the chemical substance is available, these essential contaminants are not beneficial by definition. Generally speaking, however, essential contaminants are beneficial at specific low doses.

There has been a lot of debate about Risk Assessments in which essential metals are involved. However, at high concentrations and, hence, toxic levels, risk decisions must not be influenced by the fact that the same contaminants would have been useful at lower concentrations. Of course, in the definition of an end goal of Risk Management (e.g., remediation objectives), the essentiality of contaminants must play an important role.

1.7.7 Background Concentrations

Many definitions are used for *background concentrations* (aka: baseline values or Reference Values), often in combination with the adjective 'natural' (natural background concentrations). Generally speaking, background concentration refers to the concentration in soil or aquifer over a larger area (that is, on a larger scale than the site that is under investigation). These background concentrations can either be related to natural processes or to anthropogenic activities (that is, anthropogenic activities, which took place during years, decades, or even centuries, other than the activities that caused the contamination under investigation). In the context of this book, the distinction is followed as described by the US Environmental Protection Agency (US EPA 2002):

- Natural background concentrations: contaminants present in the environment in forms that have not been influenced by human activity.
- Anthropogenic background concentrations: human-made or natural contaminants, present in the environment as a result of human activities (but not specifically related to the site in question).

Anthropogenic background concentrations can relate to both man-made and natural contaminants. The reason for this is that human activities can be responsible for the release of man-made contaminants over larger areas, for example by large scale exhaust of man-made organic contaminants via the air, followed by atmospheric deposition. But, human activities also can also lead to the spread of natural contaminants in soils, for example, by the soil surface rising, in the case of deep polder areas with shallow groundwater tables, exposing soil material that has been naturally contaminated. So the distinction is based on the type of activity that caused the background concentrations and not on the origin of the contaminants.

Like every classification of background concentrations, this distinction is artificial, since there is a transition from one to the other, for example, human manipulation of the flow characteristics of a river that will impact the deposition of sediment with natural (or maybe also anthropogenic) contaminants. Moreover, some contaminants may contribute to the background concentration as a result of both natural processes and man-made activities, such as the combined presence of naturally occurring arsenic and arsenic from pesticide applications or smelting operations.

Naturally occurring contaminants are often metals. Most organic contaminants are man-made, although many exceptions are known such as PAHs (polycyclic aromatic hydrocarbons) that are formed through natural processes such as wood fires. Cyanide-containing chemicals are produced by a wide range of organisms and plants as part of their normal metabolism. Bacteria and fungi are known producers of cyanide. A few species of centipedes, millipedes, insects, beetles, moths and butterflies secrete cyanide for defensive purposes in repelling predators such as toads and birds (MERG 2001). Some of the common plants that contain cyanide are cassava, sweet potatoes, corn, lima beans, almonds, radishes, cabbage, kale, brussels sprouts, cauliflower, broccoli, turnips, lettuce, kidney beans, and it can be found in the pits or seeds of cherries, plums, apricots, pears and apples (MERG 2001).

Many statistical procedures exist to determine the background concentrations. In general, the type of contaminant (natural or anthropogenic) does not influence the statistical or technical method used to characterize background concentrations (US EPA 2002). In most countries, background concentrations have been established on a national or regional level (e.g., Lavado et al. (2004), who determined metals background concentrations in Pampas soils in Argentina).

Background concentrations can, at the least, be used for two different purposes. First, in relation to Ecological Risk Assessment, it is often assumed that the background concentration does not pose any risk, or less risk, to the soil ecosystem, since the organisms are adapted to the long-term situation ('Added Risk Approach'). Therefore, Risk assessors need to exclude or nuance the risks caused by the background concentration. Second, background concentrations can be politically used for the definition of 'acceptable' (from a political perspective, not in term of risks) soil quality, for example, as an end goal of remediation. The substantiation for this is the fact that large scale remediation of areas of several square kilometres, for example, is nearly impossible for practical and financial reasons. Besides, it is not always defensible, from the standpoint of fairness, to upgrade the soil quality in one small part of the region (the contaminated site), while the surroundings of the site remain slightly contaminated. Therefore, the background concentrations are often declared to be of (politically) acceptable soil quality, independent of the risks involved.

1.7.8 Spatial Scale

It is important to realise on what scale soil contamination needs to be considered. When a backyard is contaminated, it is clear that the human health of the inhabitants needs to be protected. In fact, for every contaminated site, the health of humans in contact with the site must always be the primary focus. In many cases, ecological protection will also be of concern. It is generally wise, certainly for the definition of remediation objectives, to take the soil quality of the wider environment into consideration. It often happens, mostly in urban areas, that remediation of small contaminated sites results in a number of 'clean' sites in an otherwise (slightly) contaminated area. This raises the question of whether human health has been sufficiently protected in the slightly contaminated areas around the 'clean sites', since the experts have found it necessary to reduce the risk at this specific sites to lower levels than the individuals at the slightly contaminated sites are experiencing. The same issues play a role in the case of a remediation that has taken place for protection of the soil ecosystem.

Moreover, debate could arise about the cost-efficiency of several small-scale remediations in the same area. A regional-scale approach offers possibilities from a cost-efficiency perspective. It is often much less expensive, for example, to investigate and perform Risk Management options for a larger area, in one big project, than to do this in several smaller projects, at different moments in time. And in the case of groundwater contamination, as another example, it is efficient to investigate and manage all the sites that drain towards the same groundwater body and not just to focus on management of the individual contaminated plumes. Moreover, since contaminated groundwater migrates, groundwater plumes have often intermixed, which technically would not make a site-specific approach possible.

Another example in which regional-scale thinking is beneficial relates to *soil material transposition*. Soil material transposition from an intensively used residential site to a less intensively used business park in the same area, for example, would imply risk reduction for the whole area. Obviously, Risk Management is much more complicated than this simple example shows, since more elements are involved than just human health risks, for example, the effects of leaching into the groundwater at the residential site and the business park and juridical aspects in case the locations are situated in different municipalities. However, the example shows that a more regional approach offers practical possibilities for efficient contaminated site management.

A specific example of regional Risk Assessment and Risk Management is the dredging of sediment materials. When dredged materials are deposited on the site of the water courses, the overall contaminant load stays the same within the area. However, since the physico-chemical environment of the water-saturated sediments is completely different than in the water-unsaturated upper soil layer, there are often (but not always) possibilities for risk reduction.

It must be noted, however, that the interest of the stakeholders, as well as the public concern, diminishes when the area under investigation is approached from the perspective of a larger scale. For obvious reasons, the stakeholders and the general public are more committed to the site they own, live on or work on, than the wider area around the site. The consequence for contaminated site management at the scale of the site and the surroundings, and even more so on a regional scale, is that a policy becomes necessary in order to force stakeholders to follow a 'large scale thinking' approach.

1.7.9 Time Domain

In Section 1.1.4.2, it was explained that in the phase of the Problem definition it is very important to define the time frame for which the conclusions from the Risk Assessment applies, since factors that impact human health risks and ecological risks will change over time. Outcomes of the Risk Assessment should usually represent the risks over longer periods. When a site receives a positive risk appraisal, it often is assumed that the site is suited for its purpose for many decades. As a consequence, assumptions need to be made for factors that change over time, mainly with regard to the layout of the site, human behaviour characteristics and the bioavailability of the contaminants.

Moreover, the effects on humans and on bigger animals might only reveal themselves years or decades after exposure. Therefore, it is essential to focus on a toxicologically relevant time frame in Risk Assessment and Risk Management.

With regard to ecological Risk Management the time frame for which ecological restoration takes place is important. For Groundwater-related Risk Assessment, it must be realised that transport times of contaminants might take decades or even centuries. It is essential to be aware of these long time frames, both politically (what time frame is relevant?) and technically.

In fact, changes in the contaminant concentrations due to migration and degradation in the soil and groundwater also need to be considered in Risk Assessments. Remarkably, this is not often done; the concentrations measured are often considered constant in time. Since the concentrations in soil generally decrease over time due to leaching, volatilisation, and degradation, this can be considered as a worst-case approach. This is not always true for groundwater, however, since leaching from soil could increase contaminant concentrations in groundwater.

Many of the Risk Assessment factors also need to be adapted for the determination of exposure scenarios or risk estimates in case the Risk Assessment relates to a different future land use or a different layout of the site under the same land use. This situation frequently occurs, especially in densely populated areas such as in Northwest Europe, since land use transitions are common and are often preceded by a site investigation and, hence, a Risk Assessment.

1.7.10 Costs of Soil Contamination

Large amounts of money are involved in the investigation, and certainly remediation, of contaminated sites. Many developed countries spend amounts in the nine or ten digits (in euros or dollars) on contaminated sites. Today, development firms are used to incorporating cost for soil contamination in their overall costs estimates.

The general rule in most countries that have regulations concerning contaminated site management is that the persons or parties that cause the soil contamination today are fully responsible for the Risk Management solutions, often including recovery of the site to its original situation, that is, eliminating all contamination. This principle is called the *precautionary principle*, see Raffensberger and Tickner (1999) for an overview. Generally speaking, they are obligated to carry all the costs involved.

The cost aspect is much more complex for *historical soil contamination*, that is, contaminations that were caused before any regulations were available. In most countries, the *polluter-pays principle* is the foremost one. This means that the company, association, government or individual that caused a soil contamination is responsible for the costs of investigation and Risk Management. In many cases, however, it is difficult to identify the responsible party. In other cases, the polluter apparently responsible is known, but it is difficult to prove that they indeed caused the soil contamination. This has led to court cases with regard to the identification of the polluter responsible, often between a governmental body and a company.

From the perspective of fairness, the 'polluter-pays principle' can be rather harsh. In many situations, before the era of contaminated sites consciousness-raising (that is, roughly, before 1980 in many developed countries), many individuals or companies contributed to soil contamination without any negative intent. A good example are farmers who paid for waste materials in the 1960s for the purpose of elevating their soils in wetland regions, focusing on land improvement and unaware of any negative site effects due to the presence of contaminants in these waste materials. Today, they may be held responsible for the contaminants in their soils originating from these waste materials. It is a task for the politicians to find practical solutions for these cases. Fairness often demands part or complete governmental financial support.

In many countries, the owner of the estate can be held responsible for the site when no polluter can be identified.

The costs of site investigation, site management and, especially, soil remediation can be substantial. Therefore, many specific arrangements are designed in which the government will at least subsidise the actions that are needed to manage soil contamination, even when the polluter is known. Nowadays, it is generally recognised that an efficient solution of the problem of the many contaminated sites in industrialised countries requires a joint effort between the government and the business community. In other words, the budget must come from the tax payer and private initiatives combined.

1.7.11 Cost-Benefit Analyses

A different way of looking at Risk Management relies on *cost-benefit analyses*, weighing the expected costs against the expected benefits (e.g., Crettaz et al. 2002; Edejer et al. 2003). It is often difficult to quantify the benefits of Risk Management solutions. Grosse et al. 2002, for example, estimated the economic benefits from projected improvements in worker productivity, resulting from the reduction in children's exposure to lead in the United States since 1976. The authors showed that because of falling lead-blood levels, USA preschool-aged children in the late 1990s had IQs that were, on average, 2.2–4.7 points higher than they would have been if they had the blood lead distribution observed among preschool-aged children in the USA, in the late 1970s.

A cost-benefit analysis necessitates that costs and benefits should be expressed in the same units, usually in the unit of money (e.g., euros or US dollars). Generally, a Risk Management solution is beneficial when the value of the benefits is higher than the value of the costs. An optimal Risk Management solution seeks the most optimal (highest) 'value of benefits/value of costs' ratio. Since human health effects are difficult to monetarise, health effects often are expressed as DALYs, that is, a measure for the overall disease burden defined as the sum of the years of life lost due to premature mortality in the population and the years lost due to disability (WHO 2009).

Cost-benefit analyses are also used to evaluate Risk Management projects.

1.7.12 Integration of Human Health and Ecological Risk Assessment

Human Health and Ecological Risk Assessment mainly developed independently of each other. Certainly in the pioneer years of risk-based soil quality assessment, only experts from the respective disciplines were concerned with either Human Health or Ecological Risk Assessment. The role of generalists, who could have promoted integration, was limited during this phase. Another reason for the independent development is that in most countries Human Health Risk Assessment was developed earlier than Ecological Risk Assessment.

In the last few years, there has been a trend with regard to making a case for a stronger link between Human Health and Ecological Risk Assessment (WHO 2001; Suter et al. 2005). The UNEP/ILO/WHO International Programme on Chemical Safety (IPCS) formulated two fundamental reasons for the integration of Human Health and Ecological Risk Assessment, in the framework of the production, use, transport and disposal of chemicals (WHO 2001). First, it improves the quality and efficiency of assessments through the exchange of information between human health and environmental risk assessors. And, second, it provides more coherent input into the decision-making process. Indeed, in several risk-based frameworks different values are used for important input parameters, such as the Koc (organic carbon fraction-based partition coefficient), for Human Health and Ecological Risk Assessment applied to the same site. Moreover, as a third argument, it is wise to focus on the same degree of conservatism/precaution in a Human Health and Ecological Risk Assessment for the same site.

Actually, there is an important fourth reason for the integration of Human Health and Ecological Risk Assessment, and that is to improve the balance in terms of politically defined risk levels. The basic idea of the more integrated Risk Assessment framework proposed by the WHO (2001) is to treat the relationships among Risk Assessment, Risk Management, stakeholder input, and data-collection activities in a general, parallel and concurrent way.

1.7.13 Harmonisation of Risk Assessment Tools

Since the beginning of risk-based contaminated site management in the early 1980s, a large number of *Risk Assessment tools* have been developed in many countries. As a consequence, many Risk Assessment tools exist for the same purpose. By 'Risk Assessment tool' is meant a model, regression equation, table, protocol, graph or document, which can be used to determine variables that are used in Risk Assessment. These variables can vary from a 'supportive' parameter such as a Kow (octanol water partition coefficient), via the available fraction of an organic contaminant with regard to Ecological Risk Assessment, on up to 'direct' input parameters such as measured concentrations in body fluids or tissue.

Although the development of Risk Assessment tools was often based on studying existing Risk Assessment tools, the diversity of tools that are available worldwide for the same purpose is remarkable. This diversity is partly due to different geographical, cultural and social conditions, and sometimes due to differences in political points of view. However, lack of scientific consensus also explains many of the differences. An example of different Risk Assessment tools that serve the same purpose is the procedure to determine concentrations in vegetables for metals (essential for the calculation of exposure due to vegetable consumption). Many countries that have a procedure on contaminated site management derived BCFs (BioConcentration Factors) or regression equations for this purpose. In some Northern European countries, BCF values were adopted from other Northern European countries, but this is an exception rather than the rule. One reason that many countries derived their own Risk Assessment tools might have to do with the fact that the type of vegetables, and the specific genotype of that vegetable, that grow in different countries (and certainly in other climate zones) differs. This is a geographical difference. Another reason is that in different countries, apart from the *possibilities* of growing specific crops, different type of vegetables are grown because of cultural differences or traditions. However, there is no scientific, generally accepted protocol that is used in these countries with regard to the amount of data for each vegetable, quality of the data set, extrapolation margins outside the range of the input data, etc. In Europe, one of the major challenges in Risk

Assessment today is to move towards more consistency in the Risk Assessment tools used by the EU-member states (Swartjes et al. 2009).

Since geographical, cultural and social conditions indeed vary, a worldwide complete harmonisation of Risk Assessment tools is not applicable. However, for the sake of scientific integrity, a stronger convergence of Risk Assessment tools that do not include geographical, cultural, social or policy elements would be favourable (*standardised* Risks assessment tools). Risk Assessment tools that do include geographical, cultural or social elements must be applied with a certain level of flexibility so as to account for these geographical, cultural or social elements (*flexible* Risks assessment tools). Alternatively, guidance could be developed which would describe the requirements for these flexible Risk Assessment tools, which would take into account a higher degree of consistency on the part of the scientific elements of these Risk Assessment tools (Swartjes et al. 2009).

1.7.14 Brownfields

The CABERNET (Concerted Action on Brownfields and Economic Regeneration) network defines Brownfields as 'sites that have been affected by the former uses of the site and surrounding land; are derelict and underused; may have real or perceived contamination problems; are mainly in developed urban areas; and require intervention to bring them back to beneficial use' (Oliver et al. 2009). The US Environmental Protection Agency uses a different kind of definition in which the redevelopment or reuse is central, namely, 'a Brownfield site means real property, the expansion, redevelopment, or reuse of which may be complicated by the presence or potential presence of a hazardous substance, pollutant, or contaminant' (US EPA 2009). In Fig. 1.10 an example of a Brownfield in San Francisco, California, USA is shown as an aerial view.

Since the general approach for contaminated site management would imply long development time frames, large uncertainties and an over-proportional budget, a specific Risk Management approach is followed for these Brownfields. It is often claimed that Brownfield redevelopment is primarily financially driven. The general idea, however, is to make site redevelopment profitable, while at the same time protecting human health. Therefore, economic and socio-cultural factors are given greater weight than in 'normal' cases of soil contamination.

O'Reilly and Brink (2006) developed a simple screening procedure for Brownfield sites in New York State, USA, in which they classify human health risks in three categories on the basis of the concentration and toxicity of the contaminants, the location of the contaminant, the exposure route (oral, inhalative or dermal) and the type of site user (construction/utility worker, residents, industrial employees, visitors/shoppers). A popular, but very informative overview of Brownfield revitalisation, including examples from the city of Stuttgart in Germany, the Nantes metropolis in France, the cities of Tilburg and Hengelo in the Netherlands, the Medway Council and Torfaen County Borough Council areas in the UK, is given

Fig. 1.10 An aerial view of the San Francisco Bay area, with a Brownfield in the foreground (photo: San Francisco Bay regional water quality control board; reproduced with permission)

in REVIT (2008). A detailed account of the management of Brownfield sites is given in Nathanail (Chapter 20 of this book).

1.7.15 Risk Perception and Risk Communication

Risk Assessment and certainly Risk Management interacts with the daily life of the general public. The life of individuals who do not have knowledge of the effects of contaminants or of the fate and transport processes in soil are affected by contaminated sites, when they live or work on it or are in any other way associated to this site. There have been many cases in which contaminated sites raised enormous concern in the society. See Fig. 1.11, for example, which shows a notice board at which a connection between a landfill and an increased risk for cancer is presumed, in the Silvermines area in Ireland, in 2002. The general public has a much more intuitive approach towards contaminated sites than the experts have. Grasmuck and Scholz (2005) found that humans with higher scores in self-estimated knowledge tended to provide lower risk judgments, were less interested in further information, showed low emotional concern, and thus displayed higher risk acceptance.

The intuitive approach under laymen led to the consequence that the soil compartment does not really have a good reputation. For the general public, soil is a dark place, where some obscure organisms live (if any) and lugubrious decomposition processes take place. You cannot see what is happening in soils ('and maybe that is just as well'). A much more sophisticated view on soils relates to the soil as

Fig. 1.11 A notice board at which a connection between a landfill and an increased risk for cancer is presumed, in the silvermines area in Ireland, in 2002 (photo: F. Swartjes)

the source for plant production, and hence food production, and drinking water. On the other hand, many people, encouraged as a result of media attention, blame the soil for contaminating their food and drinking water.

Of course, perception of the contaminated sites problem is as varied as human character. Generally speaking, acceptability is less if the associated possible diseases are less known, manifest themselves in the longer term, and when cancer is involved (a small chance for cancer is often perceived as worse than a huge chance for another serious disease). Lima (2004), who investigated the Risk Perception of people living near an incinerator in Portugal, demonstrated that Risk Perception was initially more acute for persons living closer to the incinerator. After a while, however, the persons living closer to the incinerator showed a habituation effect. They developed less extreme attitudes and a lower estimate of the risk.

When laymen are confronted with contaminated sites, they often associate the contaminated site with the diseases that the contaminants *could* generate. Naturally, humans are often afraid of anything dangerous that they cannot comprehend nor control. It is much easier to accept something, even serious adverse effects, that humans can understand and even more so when they are able to control these effects.

For these reasons Risk Communication in an extremely important process. Typically, risk assessors often experience a situation where it is difficult to convince stakeholders of the fact that the risks are acceptable when contaminants are present, even though these are usually at low levels, simply because the diseases that these contaminants *could* generate are known. A well-known example is the presence of asbestos, maybe the best known carcinogenic contaminant in the environment due to intensive media attention, in the soil of a residential garden. The realisation alone that asbestos has been found in the immediate living environment could be cause for panic, independent of the amount, type and condition of the asbestos, let alone the risks involved.

For obvious reasons, the credibility of the risk communicator is of the utmost importance. Stakeholders' participation and intensive communication is crucial, for the purpose of putting the risks into realistic perspective and of supporting stakeholders in making the right decisions. Today, consortium building is therefore an important activity at the start of any contaminated site.

1.8 Approaches Towards Contaminated Site Assessment and Management

1.8.1 Evolution

Since the discovery of soil contamination in the late 1970s, the approach towards soil contamination has undergone a major evolution. Several reasons have contributed to this evolution, for example, the growth in understanding of the risks related to contaminated sites and of procedures for managing the risks. Moreover, the enormous increase in the number of contaminated sites that have been detected made more practical approaches necessary. A very important development in many developed countries is the more integrated approach of contaminated site management with spatial planning. And, finally, the public attitude towards the environment as a whole is constantly evolving. In recent years, the *concept of sustainability* has been advocated in many countries for the state of the environment as well as for activities that impact the environment, including soil quality.

1.8.2 Multifunctionality

In the pioneer years of contaminated site management, contaminated sites were considered to be an incomprehensible threat beyond human control, and for which the adage was rather straightforward: eliminate the whole problem in such a way that every kind of use of the site is possible. This *multifunctional approach* was advocated in many countries in the world and seemed economically feasible as long as the number of sites was limited. The advantage of the multifunctional approach was that no elaborate administrative procedures were needed for keeping an account of the possibilities for and restrictions on the use of a site. Moreover, no complicated Risk Assessment procedures were needed, since risks were, for all intents and purposes, reduced to zero. Regulators did not have to bother about acceptable risk levels.

However, the multifunctional strategy of site management did not free regulators from aftercare activities and costs. 'Dig-and-Dump', that is, removing the contaminated soil from the site and depositing it in landfills, was a popular remediation technology in the early days of contaminated site management. For contaminated groundwater 'Pump-and-Treat', that is, extracting contaminated groundwater from the aquifer and cleaning it in water treatment facilities, was often applied. But for bigger sites 'Dig-and-Dump' was often too comprehensive and expensive. And extraction of contaminated groundwater did not always explicitly solve the problem, since it resulted in contamination of clean groundwater through desorption of contaminants from the solid phase of the aquifer. For these reasons, mainly financial, technologies that required long-term aftercare were usually applied. An example of such a technology is the *boxing in* of contaminated parts of an aquifer by means of sheet pile walls.

1.8.3 Fitness-for-Use

A much more (cost-)efficient alternative for contaminated site management is the concept of *Fitness-for-Use* (or: Fit-for-Purpose or Suitable-for-Use). This concept implies that the assessment and management of the contaminated site relates to a specific type of land use. This could either be the present or the future land use. The latter is often an option when an alternative type of land use would fit the present soil quality better. Since the late 1980s, the concept of Fitness-for-Use has gained in popularity and has gradually become the leading concept in most countries.

The advantage of Fitness-for-Use is simply that in most cases less strict requirements can be applied. This is much more efficient in terms of the time frames needed for Risk Management activities and costs. Besides that, for many scientists, consultants and regulators, but also for the general public, Fitness-for-Use is a rather logical concept. The idea behind this conception is that, such as most common things in life, things need to be suited for a specific, appropriate purpose. A garage, for example, needs to be suitable for parking a car and not as a playground for children.

The disadvantage of the Fitness-for-Use approach is that aftercare is often needed. Humans can live, work or recreate at a specific site, without experiencing unacceptable human health effects. And when the soil ecosystem is considered as a protection target, the soil ecosystem can be sufficiently protected under specific conditions. However, contaminants might threaten clean groundwater through leaching and migrate to places with a more sensitive land use for human beings or the soil ecosystem. Another drawback of the Fitness-for-Use concept is that intensive administration procedures are needed in order to keep an on-going account of the state of the soil contamination and of the restrictions for the use of a site. Moreover, compared to the multifunctional approach, intensive investigations using Risk Assessment procedures and defining appropriate Risk Management solutions, often including remediation plans, are needed.

In the framework of the concerted action known as *CLARINET* (Contaminated Land Rehabilitation Network for Environmental Technologies) a concept of risk-based soil quality management has been advocated so as to be able to guide the Fitness-for-Use, called *Risk-based Land Management* (*RBLM*).

1.8.4 A More Pragmatic Approach

1.8.4.1 Mentality Change

Since the mid 1990s a significant change in mentality in terms of contaminated site management has taken place in most developed countries. Gradually, the general idea has settled in that the present philosophy on contaminated site management was solid, but too rigid and not efficient enough to make the desired progress. Desired progress, in this perspective, is mainly expressed in terms such as percentage of the total contaminated site load that has been restored. Mainly within the larger municipalities, contaminated sites had resulted in stagnation in building activity.

As a response, the basic concept of management of contaminated sites has evolved into the adage ‘environmentally acceptable and financially feasible’. Fitness-for-Use and cost-efficiency have become important boundary conditions in contaminated site management. It has become widely accepted that not every risk means an *unacceptable* risk. Therefore, the remediation objectives, at least for immobile contaminants in the upper soil layer, need not relate to ‘no risk’, but to ‘an acceptable risk’ for the specific land use (land use-specific remediation). Moreover, it was widely accepted that more cost-efficient methodologies should become available.

At the same time, many governments have developed procedures for financial support, such as sharing financial risks, the provision of subsidy grants, co-financing structures, tax benefits and ‘green’ investments.

1.8.4.2 Natural Attenuation

The concept of using biodegradation as a Risk Management solution, which generated enormous interest in the 1990s and could count on broad support from decision-makers, initiated a very important innovation, that is, *extensive* in situ *remediation technologies* (bio-restoration). Although this is the oldest remediation technology, since organisms have been breaking down organic contaminants ever since contaminants were present in the soil, it was not greatly accepted in the early days of contaminated site management. The reason for this was the relatively long time span that was needed for complete restoration of the site. Moreover, it was difficult to predict the progress of this kind of remediation. However, this change in mentality has come up with the general idea that ‘contaminants that have been in soil for many decades need not necessarily be removed within a time-span of months on up to a few years’.

This insight led to numerous investigations, mainly in the second half of the 1990s, for the purpose of understanding the processes better along with improving efficiency and predicting the time span needed. The process is now called *Natural Attenuation* (aka: intrinsic remediation), which often also includes dilution by transport processes such as molecular diffusion and hydrodynamic dispersion. Natural Attenuation is often considered as a Risk Management solution rather than as a remediation technology. Despite sometimes high starting costs, the overall budget for this Risk Management procedure is relatively low. Besides this, it results in a minimal disturbance of the natural conditions at the site. Today, these extensive remediation technologies, often in combination with ex situ techniques such as removal of the source, are widely accepted. As specific applications, 'bio-screens' are used; these are zones with an active, often stimulated, degradation at strategic positions in the soil system, or Funnel-and-Gates techniques, in which contaminants are led to zones with an active degradation. The basic principle is: 'use the natural self-cleaning capacities of the soil as much as possible, stimulate natural conditions when necessary and use ex situ remediation technologies only when strictly needed'.

1.8.5 Market-Oriented Approach to Site Development

Construction work, certainly within urban areas, is big business. Ironically, most of the interesting locations for residential developments often coincide with contaminated sites. Several former industrial sites, which were created close to the former limits of the municipality, now lie within the expanded cities. These sites, although often contaminated, offer excellent possibilities for residential development. Other examples for residential development are former harbour sites, and former waterfront storage depots or warehouses, which offer exclusive housing and recreation opportunities. Again, these sites are often contaminated. An interesting adage from site contractors involved in building activities such as developing residential areas is: 'turn a threat into an asset'. Risk Management of contaminated sites in these areas could result in a profitable rise in value of the site for different building purposes. Moreover, consultancies have proven to be experts in finding creative solutions for risk reduction.

More and more it has become accepted fact that contaminated site management is part of the integral complex package of site development (see Fig. 1.12 as an example, which shows a former with DDT (DichloorDifenylTrichloorethaan) and HCH (HexaChloroCyclohexane) contaminated site in Wuxi, China, which has been developed into a residential area after remediation). Traditionally, contractors had to adapt the physical state of a site through grading, providing drainage and guaranteeing the supporting foundation of buildings by the use of piles. One could argue that maybe contaminated site management is just another aspect of making the site suitable for building activities. Contaminated site management is just another aspect that should be included in a cost-benefit analysis. As a consequence, market-driven financing is contributing more and more to solving the problem of contaminated sites.

The development of a more market-oriented approach has gone hand in hand with the mentality change towards a more flexible way of contaminated site management and a more intensified use of Natural Attenuation techniques (Section 1.8.4.2). As a consequence, 'the market' has taken on the responsibility for a cost-efficient risk reduction at many contaminated sites.

Schelwald-Van der Kley et al. (Chapter 24 of this book) describe the philosophy on cost-efficient Risk Management solutions of industrially contaminated sites, at the same time discussing the protection of human health, ecology and groundwater.

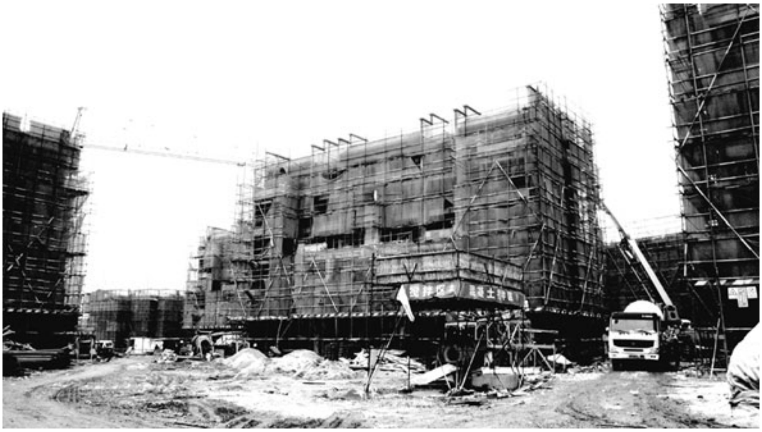

Fig. 1.12 A former with DDT (DichloorDifenylTrichloorethaan) and HCH (HexaChloro Cyclohexane) contaminated site in Wuxi, China, which has been developed into a residential area after remediation (photo F. Swartjes)

1.8.6 Integrated Approaches

Some individuals involved in risk-based soil quality assessment believe that the main purpose of the term 'integrated', mainly in the 1990s, was to 'dress up' political reports and letters. Although this opinion does not reflect reality, it is a fact that plans for and ideas about integration always overruled concrete application. Today, several variations on integration offer immense benefits in contaminated site management.

1.8.6.1 Interdepartmental

In all countries in the world, various ministries have a relationship with soil quality assessment and management. Political themes that have a relationship with soil include environment, agriculture, water resources, nature protection, and spatial planning. A balanced *interdepartmental approach*, however, would practically be impossible. Therefore, it is essential that laws and acts that influence soil policy do not permit actions to conflict, or, still better, that they actually strengthen each other. The same conclusions hold for national versus international regulations. In Europe, for example, many environmental acts, such as the Water Framework Directive, overlap with national regulations.

1.8.6.2 Spatial Planning

Traditionally, spatial planning is a process with a two-dimensional scope, that is, it is related to the arrangement of the soil surface, usually on the scale of a region. Since the late 1990s, the idea of including the third dimension, that is, soil aspects, into spatial planning, has become a point of interest. The reason for this is that the soil, that is, both the upper layer and the groundwater, impose limitations as well as opportunities for functions at the soil surface. An example of these limitations concerns the establishment of a new estate district or an industrial park on soils with limited bearing capacity (e.g., peat soils). An example of an opportunity is the planning of a nature reserve, in combination with additional water storage possibilities in wet agricultural areas, in case of high water recharges, in anticipation of climate change. Although humans are capable of changing the environment, including many soil factors, in accordance with their own requirements, these examples show that this is not always possible, or at least not without substantial additional costs. Therefore, several soil issues, including soil quality assessment and management must become part of any integrated spatial planning process.

Moreover, in densely populated regions in the world, sub-surface construction works have gained in popularity. From this perspective it becomes unavoidable to include the third (depth) dimension, i.e., the soil quality, in spatial planning.

1.8.6.3 Chemical, Physical and Biological Soil Quality Assessment

Traditionally, since contaminated sites have become a political issue, soil quality is approached from a chemical perspective, that is, focused on contaminants in soil. During the last few years the philosophy of considering overall soil quality, that is, chemical, physical and biological soil quality, gained in interest. A concrete example is the determination of so-called Soil Ambitions for local soil quality in the Netherlands (Otte et al. 2009). These Soil Ambitions can be assessed at the level of municipalities, with the use of a so-called Route planner. Although the chemical component of the assessment is the most mature, physical elements (such as sealing, or bearing power), or biological elements (such as Biodiversity) could also be included in Soil Ambitions.

1.8.6.4 Environmental, Socio-Cultural and Economic Assessment

Another interesting integration is to combine risk-based soil quality assessment with social and economic factors. In a way, economic factors always have been included, since no Risk Management activities have had access to unlimited financial resources. Today, however, cost-efficiency has become an important element of modern risk-based soil quality assessment. Socio/cultural factors have also implicitly played a role of some importance. This is reflected, for example, in the different approach to contaminated site management in urban areas than that applied in rural areas.

In the procedure for dealing with Brownfields, environmental, socio-cultural and economic factors are assessed and weighed in a systematic way.

1.8.6.5 Life Cycle Assessment

Life Cycle Assessment (LCA; aka: *Life Cycle Analyses* or *Life Cycle Impact assessment*) is the holistic evaluation of the environmental impacts of a specific product or service, from the time that the product or service comes into existence until the product is deposed of or the service is ended. For products, this usually relates to the overall environmental impacts of the product in all stages, that is, the raw material production, manufacturing, distribution, use in practice and disposal. In all these stages there is a possibility that soils are impacted. This impact relates to contaminants from the product under investigation, but also from other products that are needed for manufacturing (e.g., degreasing liquids), and that are released during distribution (e.g., from fuels during road transport) or during use (e.g., from detergents during use). In a Life Cycle Inventory the pathways of the contaminants, including the immissions to the soil compartment, are investigated. Finally, the impact is investigated in the Life Cycle Assessment, which overlaps with risk-based soil quality assessment. Risk Assessment could be an important process, and is in fact one of the steps of the more integral approach of Life Cycle Assessment.

Life Cycle Assessment can also support the selection of Risk Management solutions. Suèr et al. (2004), for example, described nine case studies in which Life Cycle Assessment was used to evaluate alternative remediation technologies. As criteria they use the spatial scale that is affected by the remediation, the time scale at which positive and negative effects reveal themselves and secondary processes such as the production of tubing, electricity use, materials used for treatment of contaminated groundwater, the production of iron fillings for reactive walls, and of active carbon, nitrate, and hydrogen peroxide for bioremediation purposes.

1.8.7 Technical Approaches

1.8.7.1 Risk Assessment Methodologies

During the last few years, many easily accessible Risk Assessment methodologies have become available. These methodologies often concern spreadsheet-like models in a Windows environment. Many of these models are readily available, for example, on the Internet. The huge advantage of this is that the Risk Assessment process has gained in popularity and is followed more often, which generally improves the procedure on contaminated site management. However, there also is a serious disadvantage, since engineers can use these models without much knowledge of Risk Assessment. It is important to realize that these models may not be used as black box models. Even more disturbing is the fact that these procedures, though in fact any Risk Assessment methodology, are easy to manipulate according to the wishes of a specific stakeholder. Therefore, any Risk Assessment needs to be accompanied by some kind of certification process. Usually regulators are primarily responsible for the objectivity and quality control of the Risk Assessment.

Another matter of concern is that individuals who are not expert often believe in models that they do not understand. Ironically, it sometimes happens that laymen trust models more when they appear to be more complex. It is the responsibility of the risk assessor and the regulators to provide transparency in the possibilities and the limitations of Risk Assessment methodologies.

An interesting and difficult issue in Risk Assessment methodologies concerns the balance between uniformity and flexibility. Uniformity is an important regulatory aspect. Two different risk assessors must come to the same conclusion about the risk appraisal of a specific site, if only from the perspective of fairness. In general, uniformity improves when Risk Assessment procedures are more rigid, with more fixed input parameters. However, a higher degree of freedom in the use of Risk Assessment methodologies stimulates the incorporation of more site-specific information, which improves the assessment, at the expense of uniformity. It is the task of both the scientists and the regulators to strive towards an optimal balance between uniformity and flexibility when making Risk Assessment methodologies accessible.

Another challenge is to find a good balance between the scientific foundation and the possibilities for pragmatic applicability. A scientist might claim that many difficult theories cannot be put into a practical format. However, practical methodologies which are scientifically not completely mature are often better than no methodologies at all. When this kind of methodology is used, the scientific limitations need to be made transparent. These scientific limitations are the basis of the interpretation of Risk Assessment results, at the same time offering the possibilities for improvement of the methodology. It takes courage, however, to follow this practical approach, since scientists thereby make themselves vulnerable.

1.8.7.2 Conceptual Model

Every Risk Assessment is somehow related to a source-pathway-receptor approach. With regard to risk-based soil quality assessment, it often pays off to start with a Conceptual Model, especially in the case of contaminated aquifers. Such a Conceptual Model gives a (usually visual) presentation of relationships between the source, all pathways involved and the receptor. A cross-section of the contaminated site is the most common format of a Conceptual Model. See Fig. 1.13 in which a Conceptual Model for groundwater-to-indoor-air mass flux analysis is shown, as an example.

In fact the Conceptual Model represents a two-dimensional contamination pattern and includes all relevant pathways involved. It relates to the migration to other compartments, for example, from soil to aquifer or to the migration within a compartment, for example, the migration of a contaminant plume within the aquifer. It also represents migration of contaminants into contact media, for example, upward transfer of volatile contaminants from the upper aquifer into a building. The source generally relates to a location, or locations, in soil that are contaminated. The receptor is a specification of the protection targets. It can relate, for example, to humans living on the site, or to the soil ecosystem in a downstream nature reserve that might be threatened by a lateral and upward flow pattern in the aquifer and upper soil layers.

A Conceptual Model might serve two important purposes. First, it supports a systematic investigation of all possible pathways, and subsequently helps to identify all necessary Risk Assessment tools. Second, it makes the whole Risk Assessment

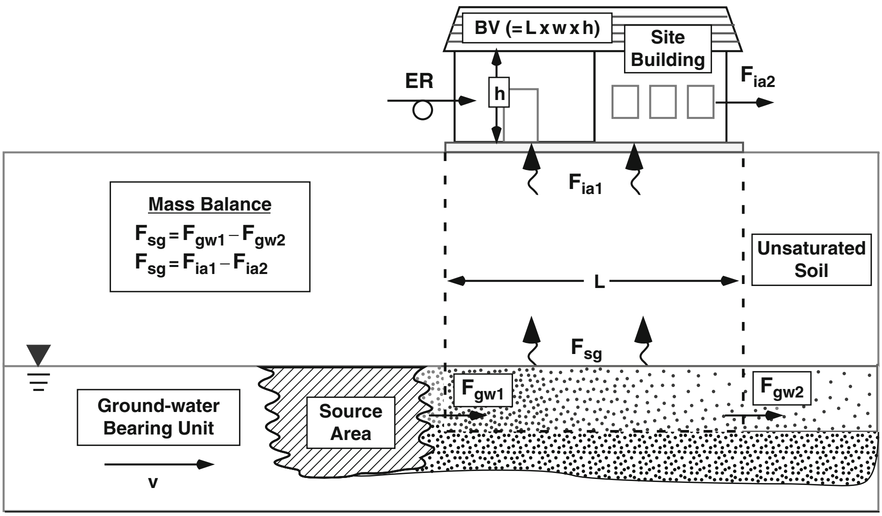

Fig. 1.13 A Conceptual Model for groundwater-to-indoor-air mass flux analysis, as an example of a Conceptual Model (source: McHugh et al. 2003; reproduced with permission). Fgw1 = mass flux through groundwater at upgradient edge of building, Fgw2 = mass flux through groundwater at downgradient edge of building, Fsg = mass flux from groundwater to soil gas under building, Fia1 = mass flux from soil gas to building, Fia2 = mass flux from building to outdoor air

process accessible, even to non-experts. For this reason, a Conceptual Model is an excellent basis for discussions in an early stage of any Risk Assessment project, along with the involvement of all stakeholders.

1.8.7.3 Tiered Approach

The most elegant way of dealing with Risk Assessment is in the form of a tiered approach. In such an approach several assessment steps (*tiers*) are described. In each tier, an assessment is performed with generally two possible outcomes: either a judgment of the absence of unacceptable risks can be given, and the total assessment is finished, or unacceptable risks cannot be excluded and the assessment has to be followed into the next tier. When the presence of unacceptable risks cannot be reputed in the final tier, unacceptable risks cannot be excluded, the total assessment is also finished and Risk Management needs to follow the tier-based Risk Assessment. Given the nature of a tiered approach, in each step the assessment becomes less conservative, is based on more site-specific information and, hence, is more complex, time-consuming and often more expensive. The philosophy behind this is: *simple when possible (only the first tier) and more complex when necessary (higher tiers)*. A tiered approach is an efficient way of risk assessing, without compromising scientific integrity.

A tiered approach often starts with a generic Risk Assessment, that is, comparing measured value with Soil Quality Standards (screening levels). The purpose could be, for example, to separate risk factors into two classes, namely, 'no unacceptable risks' and 'further research needed'. In this further research stage, fine-tuning of risks is performed in one or more higher tiers. Another option is the tripartition of risk classes in the first tier, namely, 'clearly no unacceptable risk', 'clearly an unacceptable risk' and an intermediate class. Obviously, the fine-tuning in higher tiers needs to occur in the last (intermediate) class.

1.8.7.4 Weight of Evidence

There is no uniform definition of *Weight of Evidence* (WOE), but it refers to multiple lines of investigation, for which the results are combined to improve the reliability of an assessment. The principle behind the Weight of Evidence approach is that when several uncertain results are combined, the overall result is less uncertain. Weight of Evidence is a popular tool used in forensics (e.g., Balding 2005). Since Risk Assessment typically is associated with uncertainties, WOE is often used, although the term 'evidence' usually must not be taken literally. Most approaches concern a more qualitative way of combining the results of the multiple lines of evidence. Smith et al. (2002), for example, described a quantitative Weight of Evidence approach to predict the risk at potentially contaminated sites on the Great Lakes in the USA.

Weed (2005) described three kinds of 'evidence' in the Weight of Evidence approach. The first is metaphorical where Weight of Evidence refers to a collection of studies. The second is methodological where Weight of Evidence points to different methodologies. In the third, the author referred to theoretical evidence where Weight of Evidence serves as a label for a conceptual framework. As a practical recommendation, it was stated that the WOE concept and its associated methods should be fully described when used.

1.8.7.5 Decision Support Systems

In standard works (e.g., Finlay 1994) a *Decision Support System* (DSS) is often rather broadly defined as a computer-based system that aids the process of decision-making. In the field of risk-based soil quality assessment, a Decision Support System is a methodology, often in the format of a user-friendly computer program, for which the generated outcomes are linked to concrete (regulatory) consequences for contaminated site management. It generally combines Risk Assessment tools with political positions. Decision Support Systems often have a relatively large degree of objectivity and are, hence, rather rigid to use.

DSSs have some advantages. First, using DSSs is more efficient than an 'open' site-specific Risk Assessment, since many of the choices on Risk Assessment tools (models, equations, input parameters) have already been made. For the same reason, it improves uniformity. Second, it facilitates communication, at least when DSSs have been described in detail.

An example of a DSS is the procedure used in the Netherlands (Dutch Soil Protection Act) to determine the urgency of remediation, based on site-specific risks for human health, the ecosystem and contaminant migration in the groundwater (Swartjes 1999).

For more detailed frameworks for contaminated site management, see Vegter and Kasamas (Chapter 23 of this book) from the viewpoint of policy makers and regulators; Schelwald-Van der Kley et al. (Chapter 24 of this book), from the viewpoint of the industry as an important stakeholder; and Nathanail (Chapter 25 of this book) with regard to the management of Brownfields.

1.9 Sustainability

Sustainability relates to a specific state that is able to continue indefinitely. A very workable and often-cited definition of a sustainable development is the definition of the General Assembly of the United Nations (United Nations 1987), often referred to as the Brundtland Commission, that is, 'a development that meets the needs of the present without compromising the ability of future generations to meet their own needs'. The good thing about this definition is that it focuses on future continuation (future generations must meet their own needs), but also offers a realistic perspective for the present (meeting the needs of the present).

This view of sustainability accounts for the major pitfall for contaminated site management, that is, the sole focus on short-term benefits, since spreading the benefits over longer time spans is a luxury that is not always permitted. In the so-called 'Rio declaration on environment and development', the outcome of the 1992 United Nations Conference on Environment and Development (UNCED) in Rio de Janeiro (often called the Earth Summit) (United Nations 1992), the principle of sustainability was further stressed on the widest international scale possible.

The *Cradle to Cradle Concept*, which was introduced by a designer-chemist duo (Braungart and McDonough 2002), goes one step further. This concept is not limited to preservation, but rather focuses on improvement in the quality of life. The motto 'the waste of today must be the food for tomorrow' illustrates the idea of the Cradle to Cradle Concept. Although the authors aroused a lot of scepticism—according to criticasters the idea was too general and the authors did not specify the road map towards this ideal world perspective—their philosophy opened the way to many discussions on improving long-term care for the planet, including the world's soils.

With regard to contaminated site management, it seems wise not only to focus on the time factor, but also to incorporate the undesired shifting of problems to other places. In fact, the character of contaminated sites (more specifically, contaminated groundwater), that is, a dynamic system with migration of contaminants, forces the risk assessor to account for off-site risks. As a consequence, the wider definition of sustainability used in this book is: a development that meets the needs of the present *at a particular site* and without compromising (or with improvement on) the ability of future generations to meet their own needs, and *also in the wider surroundings*. From this perspective the plea for a regional-scale approach (see Section 1.7.8) is justified.

Parallel with the increase in anthropogenic pressure on the environment, including the soil, sustainability has gained enormously in popularity. The basic reason for this is that history has proven that successful cultures have eventually been wiped out by their own success. This phenomenon seems to relate currently to most developed countries, since human presence, activities and production have already impacted many natural resources which were in equilibrium for long time frames. A highly contemporary example is the increase in the immission of substances into the atmosphere, leading to climate change. It seems that without sensible human interference the indefinite maintenance of a stable climate suited for human survival is not guaranteed.

Like the climate, soils are also strongly impacted by human activities. Obviously, human interaction with soil has a huge economic potential and should not be avoided. Sustainable soil management, however, implies that human interventions should adapt to the rhythm of natural soil processes. *A healthy soil ecosystem*, with appropriate resilience and recovery abilities, is generally considered to be an indicator of sustainable soil quality. Many processes that lead to soil contamination typically are not sustainable, since the soil ecosystem is often not able to respond properly. Without regulations, important soil processes will not continue optimally. More than that, several human activities might lead to an irreversible elimination of specific soil organisms and, hence, block ecological processes, more or less the opposite of sustainability.

An important aspect of sustainable contaminated site management relates to *prevention* of soil contamination. Jenck et al. (2004), for example, claimed that 'industrial sustainable chemistry', with process design and new equipment leading to minimal immissions, is seen worldwide today. They illustrated this statement with several current industrial case studies.

1.10 Actors Involved

1.10.1 Decision-Makers and Regulators

Ideally, contaminated site management needs to be regulated in acts or laws. Therefore, decision-makers must form the basis of any (national or regional) framework for contaminated site management, which is generally based on risk-based soil quality assessment nowadays. Many European countries, the USA, Australia, Canada, and some Asian, South American and African countries have enacted legislation on contaminated sites. The structure of involvement and responsibilities of the government differ among countries, not least of all because the size of the countries differs. Bigger countries, such as Canada or Spain, for example, have legislation at the provincial level. There also are countries that have national legislation with specifications at the provincial level.

A typical task for decision-makers, in close cooperation with scientists, is the selection of protection targets and the determination of protection levels. Another important task for decision-makers concerns the indication of the level of desired conservatism/precaution, for example, protection of the average human being, or protection of all human beings. Usually, the protection levels and the level of desired conservatism need to be determined in agreement with other existing laws. Although decision-makers play a crucial role in the definition of boundary conditions and degree of desired conservatism, it is essential that they unbiasedly commit themselves to the scientific part of the Risk Assessment and Risk Management.

Except for following fixed protection levels, decision-makers have other options for soil quality appraisal. These options are mainly implemented when defining the end goal of Risk Management, for example, through remediation. First, they can relate the acceptable soil quality to background concentrations, independent of associated risks. From the point of view of efficiency, an elegant procedure is to base the end goal on the *ALARA* (*As Low As Reasonably Achievable) principle*, using *Best Available Technologies (BAT).* This means that risk reduction is performed up to a concentration level, possibly with a maximum value that is ‘good enough’, for which the costs ‘are reasonable’. Obviously, the determination of ‘reasonable costs’ in relation to the improvement of soil quality is a subjective process. A special level of protection, not directly related to risks, is based on the *stand-still principle*, generally applied to assess the appraisal of groundwater quality or soil material transposition. According to this principle the soil quality may not deteriorate, in other words, contaminant inputs must equal contaminant outputs.

An approach that decision-makers can choose to follow is the *precautionary principle* (Raffensberger and Tickner 1999), especially in case of lack of scientific consensus. This principle is morally and politically based, and states that if an action or process has adversely impacted human health, the burden of solving the problem falls on those who caused the problem. At the 1992 United Nations Conference on Environment and Development (UNCED) in Rio de Janeiro (often called the Earth Summit) (United Nations 1992), the precautionary principle was advocated with regard to environmental protection. When applying the precautionary principle to contaminated sites, decision-makers can decide, for example, in the case of doubt about the risks involved, that the polluter is responsible for (financing the) elimination of the contaminants.

Along with the definition of protection levels, the decision-makers also have the duty to communicate them. It is often asserted that the political underpinning of these protection levels is based on coincidental aspects and very often not made transparent. In the most extreme case, protection levels that the scientists (arbitrarily) derive are implemented in soil quality assessment frameworks, since ‘they are the only levels available’ in the view of the decision-maker. Moreover, there is a risk that a specific degree of conservatism ‘slips’ into the Risk Assessment, since the scientists use the parameters that are available, without initially focussing on a specified degree of conservatism. It is important that all policy decisions are clearly specified in reports underlying, for example, Soil Quality Standards, Decision Support Systems, etc.

1.10.2 Scientists

Successful management of contaminated sites, whether on a local, regional or national scale, relies on understanding and applying a large and multi-disciplinary knowledge base that straddles the natural, physical, engineering and social sciences within a practical, commercial and political context (Pollard et al. 2002b). Scientists play a principal role in Risk Assessment procedures. Human Health Risk Assessment, Ecological Risk Assessment, Groundwater-related Risk Assessment and Food Safety-related Risk Assessment are based on a number of scientific Risk Assessment tools; these are an equation, a description, a database, a model, an instrument, a protocol, or a table. Basically, the scientists are responsible for the objective development and application of Risk Assessment and Risk Management tools.

It is difficult to profile the ideal risk assessor, since in an overall Risk Assessment several technical disciplines are needed. Moreover, specific social competencies are required. For this reason, most Risk Assessment projects are performed by a multi-disciplinary team, rather than by one single person. The basic kinds of expertise that are essential for risk-based soil quality assessment are soil science, (soil) chemistry/geochemistry, (geo)hydrology, toxicology and biology. Other kinds of expertise that support the Risk Assessment and Risk Management process are mathematics, information technology, statistics, geology, geography, and law. Moreover, any Risk Assessment team benefits from people with a social science background; these are people with communication knowledge or skills. Moreover, people with creative qualifications can make significant contributions to original site-specific Risk Management solutions, with a good balance between quality and cost-efficiency. In rare cases, technical expertise and communicative and creative qualities are combined in the same person.

1.10.3 Decision-Makers Versus Scientists

Procedures for risk-based soil quality assessment are based on scientific Risk Assessment tools and policy decisions. Since these elements are interwoven, the derivation of these procedures concerns a co-production between scientists and policy makers. As early as 1983, a good partnership between science and decision-makers was seen as an essential element of Risk Assessment (US National Research Council 1983). In most countries, however, there is no clear boundary between the tasks of policy makers and scientists. However, the relationship between decision-makers and scientists is crucial for a successful and efficient procedure for contaminated site management. Scientists need the policy makers for the definition of the problem formulation and the definition of the boundary conditions for Risk Assessments. Decision-makers, for their part, need the scientists to explain the possibilities for and exact meaning of protection levels, and the uncertainties involved, and to elaborate on technical procedures, models, protocols and related Soil Quality Standards. Although decision-makers and scientists usually have a common background, that is, academic study in a related technical field, the interest and mentality of both groups is often very different. Moreover, optimal co-ordination requires frequent communication, which is hampered by a physical separation of working places. Even more importantly, mutual communication is not implicitly productive and the significance of it, therefore, is often underestimated.

Souren (2006) investigated the role of scientists and policy makers in the derivation of Soil Quality Standards in the Netherlands in the period 1971–2000. She concluded that in general policy makers do have sufficient scientific knowledge to understand the scientific procedures used to derive Soil Quality Standards. At the same time, the author concluded that most scientists understand policy-making.

1.10.4 The Risk Assessor

Risk Assessment is often said to be the scientific part of the contaminated site management framework and an objective procedure. That might be true for a large part. However, compared to other scientific disciplines, Risk Assessment involves quite as many subjective decisions. An important example of a subjective judgement is the interpretation of measured soil concentrations, certainly in the case of heterogeneously contaminated sites. One risk assessor might focus on the highest measured value, while another assessor will use the average of all samples in the Risk Assessment. In the latter case, it might make a difference whether this individual decides on the arithmetic or geometric mean, which might differ with a factor of up to 10. Ideally, these decisions are protocolised. However, using margins in many decisions (such as the decision on which value to take for the 'representative soil concentration' in the example shown above) could actually improve the Risk Assessment. For example, in an early tier of Risk Assessment and when the risk assessor's estimate is that risks are acceptable, the risk assessor might deliberately focus on the highest concentration. In that case, the most likely outcome is 'risk acceptable and case closed', on the basis of a limited effort and budget. In other cases, for example, when the layout of the site shows different uses such as green shoulders and paved areas at a business park, the risk assessor might derive an average representative concentration, only valid for the green shoulders. In this case, since no vegetables will be cultivated on the shoulder, this average soil concentration will be good enough to investigate the possibilities for the development of typical shoulder vegetation. Using the margins in an intelligent and responsible way, in fact, makes Risk Assessment a creative and challenging process.

The risk assessor needs to accompany the technical Risk Assessment with three very important activities. First, the risk assessor should always relate the purpose of the Risk Assessment to the choices he or she has to make. Second, the risk assessor should incorporate uncertainties in every step of the Risk Assessment. Third, and last but not least, the risk assessors should make every choice, as well as the exact meaning of the Risk Assessment results, transparent. Moreover, during the whole process, communication with decision-makers and stakeholders is essential.

The quality of a Risk Assessment and the uniformity of the results are of utmost importance for a justified risk appraisal and a sensible way of spending the available resources. Clearly, all stakeholders benefit from sound decision-making, based on systematic and 'smart' Risk Assessment. This 'smart' Risk Assessment requires a knowledge base and creativity. Creativity is hard to steer, but the knowledge level of risk assessors could be approved by, for example, auditing systems, courses and exchange of state-of-the-art Risk Assessment tools. Some countries have auditing systems and most countries that have a policy on soil quality organise courses on Risk Assessment.

1.10.5 Project Managers

Project managers are, of course, of crucial importance in any project. The role of contaminated site project managers, certainly for bigger projects, requires special attention. This role is complex, though interesting, for several reasons. First, a project manager needs to be able to at least have an overview of the scientific knowledge of Risk Assessment tools. Because of the multi-disciplinary character of Risk Assessment, this overview often requires the ability to make a broad scientific interpretation on the part of the project manager. Second, the relationship between scientists and decision-makers or regulators traverses the whole Risk Assessment and even more the Risk Management process. Some projects lack efficiency, and often quality, because of the fact that decision-makers are unable or unwilling to use the scientific knowledge they need for sound decision-making. Analogously, scientists often are not capable, nor trained, to include 'soft' decision-making factors into their investigations, such as decisions on boundary conditions related to the degree of conservatism or specific political requirements in their Risk Assessments. It is the role of a project manager to bring together the political starting points and boundary conditions and the scientific elaboration of Risk Assessment tools, throughout the whole development of the project. Third, the project manager needs to have above-average communication abilities in order to be able to deal with the multi-stakeholder character of contaminated site projects. This has two different purposes, that is, the project must benefit from the knowledge and interest of all available stakeholders, and the stakeholders with less knowledge, such as land owners or inhabitants, need to be informed about developments that often affect their living environment and their well-being. Management involves different roles, both technical and social, which include aspects of work relationships as well as personal relationships. This includes safe-guarding (e.g., of the time schedule, the budget), motivating participants, team building and the optimal distribution of working tasks. It is often claimed that the creation of a good social atmosphere also stimulates the more technical performances of a project.

One interesting issue concerns the role of women in Risk Assessment and Risk Management, and, particularly, in project management. In the 'Rio Declaration on Environment and Development', the outcome of the 1992 United Nations Conference on Environment and Development (UNCED) in Rio de Janeiro (United

Nations 1992), the vital role of women in environmental management and development was stressed. Many studies addressed and many books have been written about the role of women in project management. The success of a female manager very much depends on factors other than gender, such as personality, age, experience, team structure, complexity and interest in the project. Of course, female managers also have disadvantages compared to male managers. Some men, for example, find it difficult to work under the supervision of a female (Fairhurst 1993), while some women are uncomfortable supervising men (Williams and Locke 1999). But, otherwise, women tend to be more relationship-oriented (Fairhurst 1993). Although it would be beyond the scope of this book to make a full analysis of all these factors, it can be stated that some typical female characteristics are beneficial for team building and successful communication and, hence, for Project Management.

1.10.6 Major International Institutions

Some major international institutions provide crucial information on Risk Assessment and Risk Management related to contaminated site management. Their reports are important for at least two reasons. First, the scientific basis of these reports is generally high. Second, using procedures from these institutions improves general use and harmonisation of Risk Assessment tools, while at the same time large-scale use improves the status of these procedures. Without the pretence of being complete, some crucial institutions are mentioned in this section. Since it is hardly possible within the scope of this book to enlarge upon all the relevant topics these institutions have dealt with and are dealing with and to describe the many crucial reports available, it is simply advisable to browse the Internet sites for information on specific topics. To facilitate this process, the institutions are briefly introduced in the footnotes in this section.

Some of the important international institutions are the World Health Organisation (WHO),[2] the International Programme on Chemical Safety (IPCS),[3] and the European Centre for Ecotoxicology and Toxicology of Chemicals

[2]The *World Health Organisation (WHO)* is the directing and coordinating authority for health within the United Nations system. It is responsible for providing leadership on global health matters, shaping the health research agenda, setting norms and standards, articulating evidence-based policy options, providing technical support to countries and monitoring and assessing health trends (http://www.who.int).

[3]The *International Programme on Chemical Safety (IPCS)* was developed and structured on the basis of recommendations of the United Nations Conference on the Human Environment (1972). It is a cooperative venture between WHO, International Labour Organisation (ILO) and the United Nations Environment Programme (UNEP) ILO and UNEP. The two main roles of the IPCS are to establish the scientific basis for the safe use of chemicals and to strengthen national capabilities and capacities for chemical safety (http://www.who.int/ipcs/en/).

(ECETOC).[4] Some authoritative national institutions in the USA that relate to contaminated site management are the US Environmental Protection Agency (US EPA),[5] the Agency for Toxic Substances and Disease Registry (ATSDR)[6] and The National Environmental Health Association (NEHA).[7] In Europe, The Joint Research Centre (JRC)[8] acts at the level of the European Union. Some important authoritative national institutions are the National Institute of Public Health and the Environment (RIVM)[9] in the Netherlands, The Environment Agency (EA) in the

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[4] *ECETOC* is a scientific, non-profit, non-commercial trade association with a mission to act as an independent, credible, peer-reviewed technical resource to all concerned with the identification of research needs and provision of scientific rationale for the assessment of health effects and environmental impact, and thereby to justify the industry's license and freedom to operate (www.ECETOC.org).

[5] The *U.S. Environmental Protection Agency (US EPA)* leads the Unitd States' environmental science, research, education and assessment efforts. The mission of the Environmental Protection Agency is to protect human health and the environment. Since 1970, EPA has been working for a cleaner, healthier environment for the American people. The EPA headquarters are in Washington, DC, but there are many other locations, such as regional offices, regional visitor guides, laboratories and research centres (http://www.EPA.gov).

[6] The *Agency for Toxic Substances and Disease Registry (ATSDR)* is an agency of the U.S. Department of Health and Human Services. The mission is to serve the public by using the best science, taking responsive public health actions, and providing trusted health information to prevent harmful exposures and disease related to toxic substances. ATSDR is directed by congressional mandate to perform specific functions concerning the effect on public health of hazardous substances in the environment. (http://www.atsdr.cdc.gov/about/index.html).

[7] The *National Environmental Health Association (NEHA)* had its origins in the state of California, USA, where it was incorporated in 1937. The original impetus behind the creation of a national professional society for environmental health practitioners was the desire by professionals of that day to establish a standard of excellence for this developing profession. NEHA's mission is 'to advance the environmental health and protection professional for the purpose of providing a healthful environment for all' is as relevant today as it was when the organisation was founded (http://www.neha.org/about/neha.html).

[8] The *Joint Research Centre (JRC)* is a research based policy support organisation, and an integral part of the European Commission. The JRC is providing the scientific advice and technical know-how to support a wide range of EU policies. Their status as a Commission service, which guarantees their independence from private or national interests, is crucial for pursuing our mission. This mission is 'to provide customer-driven scientific and technical support for the conception, development, implementation and monitoring of EU policies'. As a service of the European Commission, the JRC functions as a reference centre of science and technology for the Union. Close to the policy-making process, it serves the common interest of the Member States, while being independent of special interests, whether private or national.

[9] The *National Institute for Public Health and the Environment (RIVM)* is a recognised leading centre of expertise in the fields of health, nutrition and environmental protection. RIVM's mission is to benefit people, society and the environment, matching thier expertise, knowledge and research with that of colleagues from around the world. The institute works primarily for the Dutch government, but shares their knowledge with governments and supranational bodies around the world. The results of RIVM's research, monitoring, modelling and Risk Assessment are used to underpin policy on public health, food, safety and the environment (http://www.rivm.nl).

UK and Wales[10] and the Flemish Institute for Technological Research (VITO)[11] in Flanders, Belgium.

1.11 Scope of the Book

To sum up the previous sections, this book focuses on the possibilities for investigating whether 'there is a problem' with potentially contaminated sites (Risk Assessment) and, if so, how to deal with this situation (Risk Management). The scope of the book is limited to the extent of that part of the earth's crust that impacts human health and the ecosystem, that is, the water-saturated upper soil layer and the groundwater that is within human reach. The book is primarily focused on procedures for dealing with *terrestrial* contaminated sites, not on surface water or sediments.

The book deals with contaminated sites, either diffusely contaminated or locally contaminated (although since most chapters deal with tools that can be used for any type of contaminated site, this distinction is not always relevant). Physical quality of the soil is not within the scope of the book. Also the risks of radioactive contaminants, endocrine disruptors, microbial contaminants and nanoparticles in soil or groundwater are not included in this book, since they are of a different nature and require a different kind of Risk Assessment. This book does not focus primarily on agricultural practices, therefore, no further attention is given to (the consequences of) (macro) nutrients (nitrogen, phosphorus, potassium, and sulphur). However, Risk Assessment tools are described that can be used to assess the risk for Food Safety.

Since in most countries separate legislation exists for the assessment and control of occupational risks, this book does not implicitly focus on the risks of humans that are exposed to soil contaminants during working activities such as digging or other ground construction works. Many of the tools for Human Health Risk Assessment that are described in this book, however, could in principle be used for this purpose.

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[10]The *Environment Agency (EA)* is an executive non-departmental public body responsible to the Secretary of State for Environment, Food and Rural Affairs and an Assembly-Sponsored Public Body responsible to the National Assembly for Wales. Their principal aims are to protect and improve the environment, and to promote sustainable development. The EA is going through changing times, both in terms of the physical and business environments. Their new strategy builds on the improvements they have delivered in recent years, but it provides a new framework to guide our work to protect and improve the environment of England and Wales in the challenging climate of the next 5 years (www.environment-agency.gov.uk).

[11]The *Flemish Institute for Technological Research (VITO)* has, in its 18 years of existence, achieved the status of being a key player in the European world of research and development. In the research domains of environment, energy, materials and remote sensing, VITO's strength has been its pursuit of applied and practical research and development which is relevant for industry and public authorities. VITO expresses its 'vision on technology' through the recommendations given to clients vis-à-vis technological developments, as well as in the way in which the VITO experts develop new technology and support companies with their innovation (http://www.vito.be).

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