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REVIEW

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Remediating polluted soils

Abstract This review focuses on treatment-based remediation of soils and the acquisition of data to support and monitor this remediation. Only in the last two decades has significant progress been made in regulating for soil pollution, with a parallel development of methodologies for soil assessment and remediation. However, soil complexity remains a problem for pollutant measurements relevant to environmental risk and informative to the design or evaluation of remediation technologies. Understanding the distribution of pollutants between different soil phases and the kinetics of transfer between these pools is fundamental to prediction for these processes; further progress is needed to characterise less accessible pollutant pools and to develop guidelines for their analysis. Available remediation options include physical, chemical and biological treatments, and these options offer potential technical solutions to most soil pollution. However, selecting the most appropriate approach requires detailed information on how pollutants interact with soil physio-chemical properties. Only general information is available as to the effectiveness of specific treatment systems for particular soil type–pollutant combinations. Given the high degree of heterogeneity in physio-chemical characteristics and pollutant distribution of affected soils, prediction of treatment timescales and levels of residual contamination remains a problem. On sites with a range of organic and inorganic pollutants present, combinations of different treatment approaches may offer the best prospect for effective remediation. Further work is needed to provide evidence that residual contamination does not pose significant risk and to evaluate effects of treatments on general soil function in relation to this contamination.

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Introduction

Pollution can be defined as the introduction of elements, compounds or energy into the environment at levels that impair its functioning or that present an unacceptable risk to humans or other targets that use or are linked to that environment. Humans are at risk from polluted soils through dermal contact, ingestion, consumption of food grown on polluted areas and inhalation of dusts or vapours (Nathanail and Earl 2001). Soils may fail to support vegetation (Fig. 1) because of phytotoxic effects of pollutants (e.g. Chaineau et al. 1997; Siddiqui et al. 2001) or disrupted biological cycling of nutrients (e.g. Belyaeva et al. 2005; Khan and Scullion 1999). These soils may also affect the hydrosphere compromising the quality of drinking water resources and threatening the aquatic ecosystems. For example, an area of some 5,000 km^2 in Germany affected by lignite mining threatens surface water and aquifers with acidification (Bilek 2004).

Pollutants can build up in soils from several sources. The spreading of wastes such as sewage sludge (Bright and Healey 2003; SolerRovira et al. 1996) or other biosolids to land can be a problem especially where these wastes have been applied repeatedly over a number of years. Some wastes (e.g. dredgings, pulverised fuel ash or mine spoils) constitute new ‘soil-forming’ materials (e.g Bramley and Rimmer 1988; Shaw 1992; Winterhalder 1996). Soils may become polluted by atmospheric deposition from traffic, and incinerator or metal smelting emissions (Helmisaari et al. 1995) over a period of time. Soils may also be polluted through the spillage of liquids such as oil or industrial solvents (Collins et al. 2002), or through flooding or irrigation with polluted water (Siebe 1996).

Historically, pollution of soil has been of limited concern. As a result, there are examples of severe and widespread pollution. In the Sudbury region of Canada, for example, a combination of metal (primarily Cu and Ni) mine waste disposal and smelting emissions commencing in the late 19th century led to widespread devastation of land up to 30 km from the sources of pollution (Winterhalder 1996), more than 50,000 ha of land in-

Fig. 1 Former forest areas affected by acidic, metalliferous smelter emissions; trees in foreground were planted after remediation treatments—Sudbury, Canada

capable of supporting other than very limited vegetation cover and an ongoing programme of soil and water remediation over more than 40 years. More recently, Mulligan et al. (2001a) have estimated that 22,000 Mt of Cd and 1,372,000 Mt of Zn were deposited globally on soils during the 1980s; Fengxiang et al. (2003) estimated a cumulative increase in the As concentration of surface soil due to anthropogenic activity equivalent to 2.18 mg kg^{-1} and an accelerating trend in these inputs up to the year 2000. There have been numerous examples of tailing dam failure in Europe (e.g. Aznalcollar mine, Spain as reported by Lopez-Pamo et al. (1999) and elsewhere in recent years leading to widespread pollution of floodplain soils.

Whilst most of the early examples of soil pollution related to metals or other inorganic pollutants, there has been an increasing concern over the last few decades regarding organic contaminants, a reflection of their widespread use in industry as solvents, feedstocks and their presence in industrial wastes (e.g. Collins et al. 2002). Fuel hydrocarbons, for example, (Solano-Serena et al. 2001) are major pollutants of soils and aquifers. Combustion processes have led to widespread contamination of soils (vanBrummelen et al. 1996) with polycyclic aromatic hydrocarbons (PAHs).

Remediation approaches encompass applied physical, chemical and biological environmental sciences. The aim of this review will be to illustrate current understanding of the scientific principles underlying soil remediation and some of the challenges to their successful application. Remediation approaches that isolate treated soils are site rather than soil remediation technologies. These approaches, and the treatments that result in the destruction of soil function, will be referred to only in passing. The emphasis of this review will be on approaches that treat soils whilst retaining most of their essential functions. Recent progress in the science of soil remediation and challenges that lie ahead will be considered. More comprehensive detail on remediation technologies for polluted soils can be accessed in, for example, Armishaw et al. (1992) or Stegmann et al. (2001).

For the purposes of this review, soil will be taken to include all materials which have or are expected to form the surface layer and to support vegetation. On many sites, at least some contaminated materials do not meet this definition, and soil treatment is complicated by the presence of obstructions (e.g. building foundations and underground tanks or pipework) or by high proportions of oversized materials (e.g. building rubble). These are important issues for the practical implementation of remediation schemes, but are site specific and not strictly soil remediation, so will not be considered further.

Both soil and groundwater pollution are a problem on many sites, and their remediation is often linked. Several remediation approaches, particularly physical systems, involve the treatment of aqueous phase pollutants and, here, the distinction between soil and groundwater is of limited practical significance. Remediation approaches aimed primarily at treating or containing groundwater within 'geological' materials will be mentioned only briefly, whereas those commonly used for dual purposes will be considered in more detail. It is recognised that many of the scientific principles underpinning soil remediation have relevance for groundwater treatment.

Environmental regulation and remediation

In practical terms, a polluted soil is one where the concentration of a contaminant exceeds that set out in the relevant regulations. Whilst early systems of regulation were to some degree arbitrary, the process of establishing 'safe' concentrations of contaminants in soils has become increasingly risk based (Swartjes 1999). The definition of polluted soils is now embedded in this risk assessment process, although these definitions continue to evolve. Key to the assessment is an evaluation of the hazard-pathway–receptor linkage.

Whereas water and air have been subject to environmental regulation since the 19th century (RCEP 1998), equivalent measures for soil, or indeed the concept of polluted soils, were not considered until the later decades of the 20th century. Sheppard et al. (1992) reviewed the development of standards for polluted soils and related these developments to the emergence of contaminated land as an important environmental issue in the early 1980s. They listed a range of regulatory initiatives including those commencing in 1981 (USA), 1983 (Denmark, Netherlands and West Germany), 1984 (Canada and France), 1987 (UK) and 1990 (Australia and New Zealand). Most of these countries have seen significant changes in their regulatory regimes over the intervening period.

Early efforts at environmental regulation focussed on the protection of human health (RCEP 1998). The link between soil pollution and human health was less obvious than that for air and water. The earliest measures to regulate soil pollution, relating to sewage disposal on agricultural land (Sewage Sludge Directive 86/227/EC), were linked to food production. This was followed by regulations aiming to protect water resources from pollutants leaching from soils. Only more recently have we recognised that there are multiple pathways by which polluted soils pose risks to humans and the broader environmental system. Even more recently, the need to protect soil as a resource has been widely recognised (Defra 2001; EC 2001), although individual countries had implemented related soil protection policies before this (e.g. Germany; Bachmann 1991).

The delay in regulating for soil pollution was partly due to the complexity of the medium affecting contaminant behaviour. For example, K_d or the equilibrium soil partition coefficient between solid and aqueous phases varies widely between different soil types; Sheppard and Thibault (1990) quote K_d values for lead ranging from 270 l kg^{-1} in a sandy soil with low organic content to 22,000 l kg^{-1} in a peaty soil. It follows that the phase distribution for Pb will vary with depth within a single soil profile as organic contents decrease typically with depth. Many important organic pollutants are non-polar and hydrophobic with very large octanol:water partition (K_{ow}) coefficients; these also have a high affinity for organic matter and can become occluded within the soil humus fraction. Partitioning between solid and liquid phases will also vary with contaminant concentration (Tiller 1996) and factors that show marked temporal and spatial variation such as pH, competitive adsorption and inputs of organic compounds (Staunton 2004). These soil-specific factors can have a major influence on pollutant impact; Khan and Scullion (2000), for example, demonstrated significant effects of biosolid metals in some soils but no effect in others.

The setting of guidelines or standards has often driven research and technological developments in soil remediation; for example, limits have been set at levels achievable with technologies that were then available (Sheppard et al. 1992). The regulatory framework often determines the viability of a particular remediation process. In Holland, policy initiatives aimed at discouraging disposal of polluted soils to landfill limited this amount to 19% of excavated soils during 2002 and encouraged the development of treatment based systems; in the UK, with a historically inexpensive disposal route, most polluted soils (1 to 5 Mt per annum; RCEP 1996) have been sent to landfill (Doak 2004).

There has been an evolution of strategies for managing polluted soil problems, away from objectives of multifunctionality and towards a fitness for use of concept based more on site-specific risk assessment (RCEP 1996); accompanying this evolution has been a broader concept of environmental protection that includes chronic effects on human health and impacts on broader ecosystem function (RCEP 1998). These trends have important implications for soil remediation. For example, remediation approaches based on immobilisation of pollutants in situ cannot meet requirements to reduce contaminant concentrations below a threshold value but may offer an effective means of risk reduction in many situations.

Measuring pollutant concentrations

The setting of soil pollution limits assumes an agreed method for measuring the concentration of a pollutant that is relevant to risk assessment across differing soil types. Limits are generally expressed in terms of 'total' concentrations as there is no consensus on alternative (McLaughlin et al. 2000) methods more directly related to biological or environmental risk. Yet, assessing the bioavailability of soil pollutants is an essential part of the process of risk assessment and of determining the most appropriate approach to remediation (Semple et al. 2003).

In the case of metals, there is a scarcity of information about the kinetics of metal reactions in soils (Helmke and Naidu 1996), of desorption reactions for ingested soil in the human digestive system (Tiller 1996), and uncertainty about the bioavailability of metal complexes (Slaveykova and Wilkinson 2005). Sparks (2000) considered that the use of molecular scale in situ analytical techniques had significantly advanced the determination of speciation and sorption-release mechanisms of metals and oxyanions in soils. Techniques such as electron probe analysis and X-ray absorption fine structure spectroscopy have improved our ability to quantify the distribution of elements between various operationally defined pools. Species distribution may also be predicted on the basis of theoretical chemical concepts using computer modelling (Adriano 2004).

There are similar problems with developing non-exhaustive solvent extraction procedures that consistently predict the bioavailability of organic contaminants across a range of soil conditions (Semple et al. 2003). As an alternative to extraction, solid-phase micro-extraction uses adsorbents added to soil–water slurries aiming to mimic the accessibility of organic contaminants to microorganisms (Cornelissen et al. 1998). In relation to the assessment of risks to human health, much work is currently underway to develop physiologically based extraction tests; however, progress made in this respect for inorganic pollutants has not been matched by that for organic pollutants (Ruby 2004).

In recent years, there has been a growth in the use of on-site assays to improve decision making regarding the extent of pollution in batches of potentially polluted materials and, therefore, the need for treatment or disposal. In many cases, these new measurements are based on enzyme-linked immunosorbent assays linked to spectroscopy. Specific assays have, for example, been developed for pentachlorophenol (Li et al. 2001) and PAHs (Knopp et al. 2000). Whilst these methodologies can provide useful supplementary and 'real-time' information on pollutant concentration variability in the field, care must be taken when extrapolating findings from the very small samples used in these assays to bulk soil properties.

Various microbiological assays have been proposed as indicators of pollutant bioavailability. Biosensors (Rodriguez-Mozaz et al. 2005) have been widely deployed to provide fast, cost-effective monitoring of pollutants and their biological toxicity. McGrath et al. (1999) was able to show that lux-modified *Pseudomonas fluorescens* responded to ‘bioavailable’, rather than total, concentrations of sewage metals in soil and argued that this approach allowed toxicity assessments be undertaken without the significant modification to soil conditions involved in Microtox (Azur Environmental, Berkshire, UK) assays. Preston et al. (2000) used a similar whole organism biosensor approach to investigate the toxicity of Zn, Cu and Cd combinations in solution. They noted that exposure time had a significant effect on the assessment of toxicity. These findings emphasise both the importance of standardised protocols in the use of biosensors and some potential complexities in interpreting their responses. An alternative approach is to evaluate pollutant impacts on whole community response to stress based on analysis of phospholipid fatty acid profiles or substrate-induced respiratory responses to different carbon substrates, as reviewed by Harris (2004).

Soil standards are meaningful only if they are used in conjunction with prescribed methodologies for sample preparation and analysis. Assays used to measure total concentrations of soil pollutants vary between administrative authorities. This point is well illustrated by Hortensius and Nortcliff (1991) who compared total cyanide concentrations in soil and groundwater as measured by then current Dutch (NEN 6489) and US Environmental Protection Agency (EPA 335.3) procedures. Concentrations varied markedly between the two assays and they did not even rank the samples tested in the same order of pollution. Pollard et al. (2001) similarly reported findings from inter-laboratory comparisons of arsenic concentrations which varied significantly for different methods and even between different laboratories using the same analytical technique.

Spatial heterogeneity and behaviour of pollutants within soils

Patterns of soil pollution may be difficult to predict and this presents a challenge to risk assessment, design of remediation schemes and validation of remediation outcomes. On many industrial sites, pollutant concentrations are highly variable, especially if the pollutants are immobile and there are multiple phases of polluting activity. Under such conditions, it may be inappropriate to interpolate between points, and the concept of a soil pollution map needs to be reconsidered.

The problem of site heterogeneity is well illustrated by a study (Smith and Ellis 1986) where samples were taken at 1-m intervals along two trenches on a former gasworks site and from several depths at each sampling point. For many pollutants, concentrations at neighbouring sampling points, or indeed from within layers of similar appearance, varied by up to two orders of magnitude. Whilst gasworks sites present one of the greater challenges in this respect (see Fig. 2), high variability is a ubiquitous feature of soils on former industrial land. The hydrogeochemical complexities of many sites add to these problems (Rao et al. 1996).

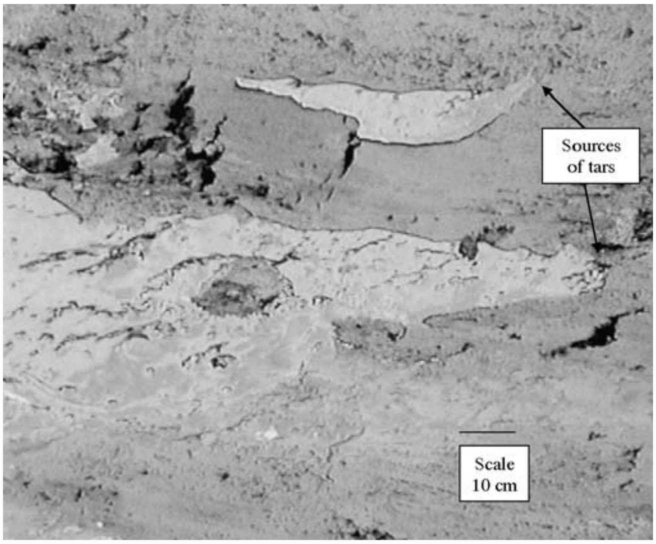

Fig. 2 Tars oozing from point sources in otherwise uncontaminated soils illustrate the heterogeneous nature of contaminant distribution. Former coal-gas site—Aberystwyth, UK

At a micro-scale, the distribution of pollutants can be equally heterogeneous (Fig. 3). Their distribution between phases and rates of exchange between these phases determine the extent of pathways by which pollutant hazards link with potential human or environmental targets. Sorption and desorption processes control behaviour of less polar and non-polar organic contaminants (Huang et al. 2003). Uptake and degradation of organic pollutants by microorganisms is far greater from liquid than from solid phases (e.g. Ogram et al. 1985), so phase equilibria have an important influence on their persistence and how amenable they are to remediation. However, there is evidence (e.g. for pentachlorophenol Dudal et al. 2004) that aqueous-phase concentration does not adequately reflect the dynamics of contaminant availability to microbes and that they can induce desorption in some circumstances. It should also be noted that sorption equilibria are rarely achieved in soil macro-pores because of the rapid passage of liquids through these systems (e.g. Rahman et al. 2004).

Variations in sorption capacity, in sorption–desorption kinetics and sorption–desorption hysteresis, particularly for organic contaminants, have been linked to differences in soil organic matter characteristics. Beck et al. (1995) emphasised the biphasic nature of contaminant release from the solid phase; where compounds are not degraded or lost fairly quickly from soil, their chemical and biological availability decreases rapidly over periods of minutes to hours, then slowly over periods of weeks to months through sorption and diffusion processes referred to as ‘ageing’ (Semple et al. 2003). This pattern of pollutant behaviour has been attributed to intra-particle diffusion, intrasorbent diffusion and chemisorption processes. Sequestration of some organic contaminants is considered to involve their diffusion into the complex structures that make up soil organic matter. Sorption-retarded diffusion may also occur within micro-pore systems. Zimmerman et al. (2004) demonstrated that mineral pores (2- to 50-nm diameter)

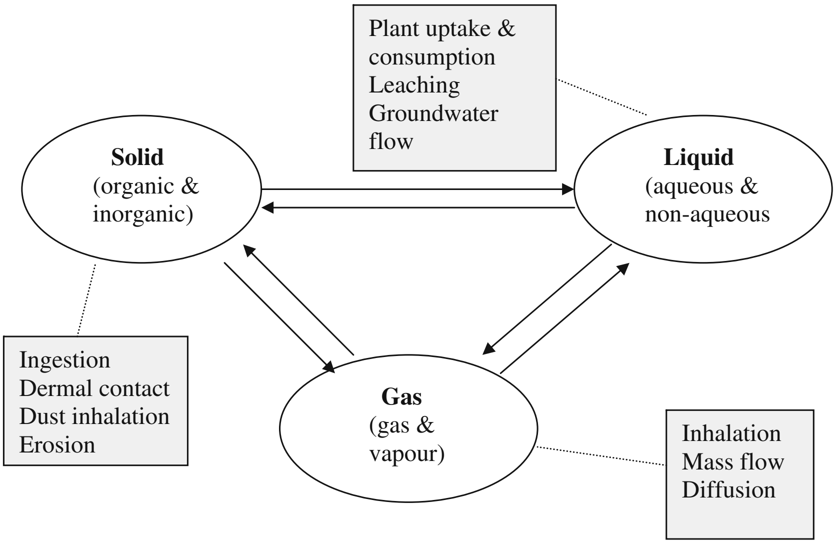

Fig. 3 Distribution of pollutants between soil phases affects pathways linking hazards to receptors. Phase concentrations tend towards equilibrium ratios but the dynamic nature of soil conditions means that full equilibria are rarely achieved

protected a model humic compound from enzymatic degradation. Understanding these processes is fundamental to the prediction of remediation success; desorption kinetics may be a critical factor, for example, determining the potential for groundwater quality to deteriorate after soil remediation, often referred to as 'rebound' (e.g. Aksoy and Culver 2000), due to slow transfer from the solid-phase pool.

The evolution of polluted soil remediation practice

Polluted soils have traditionally been either excavated for disposal to landfill or isolated in situ by the use of various 'barriers' that prevented movement of pollutants off-site or to contact between humans and these pollutants. In terms of risk management, these approaches aimed to control the pathways linking hazard and receptor without treating the source of the hazard. Remediation practices emphasised containment rather than treatment. Some of the early in situ containment approaches, for example at Love Canal in the USA and Lekkerkerk in Holland (Pollard et al. 2001), were based on fairly limited understanding of contaminant behaviour and proved to be seriously inadequate.

There has been greater interest more recently in treatment of polluted soils (e.g. Rulkens et al. 1998), or mitigation of the hazard, a trend partly driven by regulatory changes, such as the European Union (EU) Landfill Directive (99/31/EC), that discourage disposal to landfill. In some countries, in situ containment has been viewed as a waste disposal process and, therefore, subject to the same stringent regulations, permitting processes and liabilities (Doak 2004). In a very recent European Court decision, polluted soil has been classified as waste, even before any remediation effort (ENDS 2004).

There has also been a marked improvement in the dissemination of information relating to remediation treatments. There are now well-established sources of data on treatment capabilities and costs in programmes such as the US EPA Superfund Demonstration Projects (http://www.epa.gov/tio/) and through EU networks (e.g. EUGRIS http://www.eugris.org).

Options for remediating polluted soils

Treatment approaches may be classified as biological, chemical or physical, although it should be emphasised that in many cases, a combination of processes may offer the most effective remediation (Armishaw et al. 1992). Treatments may be applied in situ (treating relatively undisturbed soils) or ex situ (treating excavated soils) either on site or in designated soil treatment facilities. Treating soils in situ has the advantage of minimal disruption to activities on site or on adjacent land. However, ex situ approaches generally offer greater scope for managing conditions to optimise treatment efficiency and for controlling potential spread of pollutants. In situ processes usually involve the movement of air or water through the polluted soil and so are favoured by more permeable media and by lower heterogeneity of physical conditions and pollution distribution.

Remediation may have several outcomes in terms of the pollutants in soils. These outcomes include (based on Nathanail and Bardos 2004):

- Complete or substantial destruction/degradation of the pollutants
- Extraction of pollutants for further treatment or disposal
- Stabilisation of pollutants in forms less mobile or toxic
- Separation of non-contaminated materials, and their recycling, from polluted materials that require further treatment
- Containment of the polluted material to restrict exposure of the wider environment

Two key issues have to be addressed in designing remediation programmes. Firstly, which of the available remediation options are capable of achieving the required level of risk reduction? Secondly, which of these options can deliver this reduction within the time, cost and practical constraints applying to a particular site? The first of these issues addresses scientific and technical aspects of remediation practice and is the main focus of this review.

Physical treatment systems

A detailed consideration of containment systems is beyond the scope of this review. However, it should be noted that these systems may be used in conjunction with attenuation approaches or, in the case of polluted groundwater, can take the form of permeable reactive barriers that transform pollutants passing through them into environmentally acceptable forms (Jefferis et al. 1997). Although these systems represent site or groundwater remediation approaches, the long-term outcome may be a remediated soil.

Thermal treatment of soils for the destruction of organic pollutants involves high-temperature incineration (>1,000°C) or a two-stage process in which pollutants are desorbed at lower (<600°C) temperatures and then combusted; some essential functions of soil can be retained with low-temperature thermal desorption (Norris et al. 1999). Vitrification also involves high-temperature (>1,000°C) treatment of soils to form a solid, ceramic-like material in which inorganic pollutants are trapped. Solidification of polluted soils can be achieved by mixing them with cements or similar materials, again to form a solid mass. The end products are rarely considered soil-forming materials, more usually being disposed to landfill or made into alternative products, so these processes will not be considered further. It should be noted, however, that thermal treatments such as microwave heating (George et al. 1992) or injection of steam/hot air have been used to enhance vapour extraction at temperatures (~100°C) that do not irretrievably damage soil function.

Most physical treatment processes remove pollutants from the soil–water complex for further treatment or disposal in a more concentrated form as summarised in Table 1. In some cases, physical treatments can enhance the effectiveness of, for example, biological degradation of contaminants or indirectly cause their destruction. Removal of pollutants from soil relies on an understanding of their physical behaviour in this environment. Physical remediation processes are most effective in coarser-textured soils, although fracturing of finer-textured soils may extend their applicability, and for pollutants that are more soluble or volatile.

Table 1 Main physical remediation methodologies and their effects on soil pollutants (✓ = main process, (✓) = subsidiary process limited in extent or in the range of pollutants affected)

Process treatment	Destruction/ degradation	Solid separation	Extraction/ loss	Stabilisation
Thermal	✓		✓	
Solidification	(✓)			✓
Vapour extraction			✓	
Air sparging	(✓)		✓	
Washing/pump and treat	(✓)		✓	
Electroremediation	(✓)		✓	
Particle sorting		✓		

Vapour extraction and air sparging are treatments based on manipulation of the pollutant distribution between liquid and vapour phases. These treatments promote the volatilisation of pollutants (e.g. benzene, toluene, ethylbenzene and xylene and chlorobenzenes) in the unsaturated or saturated zones. Extracted gases or vapours may be sorbed onto activated carbon or treated (e.g. by oxidation). The effectiveness of vapour extraction systems may be extended to semi-volatile pollutants by injection of heated air or heating by microwave/ radiowave (George et al. 1992), and rates of extraction may be improved by increasing air flow rates to a point when mass transfer limits volatilisation (Park et al. 2005). Air sparging induces partitioning of dissolved and free-phase contaminants into the vapour phase, and increases in dissolved oxygen can stimulate aerobic biodegradation. Although Adams and Reddy (2003) found that volatilisation was the dominant process in benzene removal during air sparging, significant biodegradation did occur at rates controlled by oxygen concentrations.

Both of the above approaches are less well suited to fine-textured soils because of restricted rates of movement in the mobile phases and increased distances over which volatile organic contaminants have to diffuse through an aqueous phase. Also, treatment rates are slower in soils with higher organic contents (Gomez-Lahoz et al. 1995). In soils with heterogeneous physical conditions, preferential pathways may develop such that the mobile phase bypasses bulk soil, a particular risk where high-pressure sparging systems propagate fracturing.

Nathanail and Bardos (2004) stated that 17% of the soil volume treated in the Netherlands during the 1980s was subject to soil washing. The effectiveness of this treatment approach can be high for hydrophilic pollutants such as aniline and phenols (Rajput et al. 1994). Where adopted in situ, it can involve extraction and treatment of polluted groundwater which is then recirculated (pump and treat) to progressively desorb pollutants from the soil matrix. In principle, the effectiveness of this process should decline as the total concentration of pollutants falls. Also, the rate at which water can be effectively circulated depends on the kinetics of desorption; as noted previously, information on these kinetics is limited for many pollutants (Tiller 1996). Where cycles of soil flushing are too rapid to allow desorption to re-establish phase equilibria concentrations, pollutant concentrations in groundwater can increase once treatment ceases (Aksoy and Culver 2000). This is a particular limitation for pollutants with low aqueous solubility and a strong affinity for particular soil fractions (e.g. PAHs and organic matter). The balance of the equilibria can be changed in favour of the liquid phase by altering soil chemical properties, for example, through the use of surfactants (in trichlorobenzene extraction—Rajput et al. 1994) or manipulation of pH (see chemical systems).

Soil washing ex situ often combines extraction of more soluble pollutants, followed by the separation of solids with different degrees of pollution on the basis of size, density or surface chemistry (Mulligan et al. 2001a). The separation of different solid components is not a treatment per se but is a means of reducing the volume of polluted materials requiring further treatment. Some pollutants concentrate in certain soil fractions (e.g. clay minerals or organic matter). Where other fractions are relatively 'clean', separation before treatment can offer significant cost savings and present a medium that is more uniform for subsequent treatment. Many of these treatment systems are based on practices well established in mining and mineral-processing industries (Armishaw et al. 1992).

A further 'physical' treatment process involves the mobilisation and migration of pollutants in an applied electric field through electrolysis, electro-osmosis and electrophoresis. Pollutants are then collected and treated. Electro-remediation has been used primarily to aid extraction of ionic compounds, especially metals and inorganic anions such as sulphates, but can be effective for polar organic compounds. Virkutyte et al. (2002) considered electro-remediation to be most effective in treating near saturated, clay soils polluted with metals (removal >90%). It can be targeted to specific locations without the need for excavation. However, it is less effective in organic- and carbonate-rich media and control of soil pH is a key management factor determining solubility of pollutants and thus, treatment rates. Saichek and Reddy (2005) reviewed the use of surfactants/cosolvents to enhance desorption and extend the use of electro-remediation to the extraction of hydrophobic organic contaminants. They noted, however, that the reactions involved were complex and that optimisation of performance depended on an understanding of these reactions. In addition, the effectiveness of electro-remediation can be compromised by the presence of metal objects within the treated soil.

Table 2 Main chemical remediation methodologies and their effects on soil pollutants (✓ = main process, (✓) = subsidiary process limited in extent or in the range of pollutants affected)

Process treatment	Destruction/ degradation	Extraction/loss	Stabilisation
Oxidation	✓	✓	✓
Reduction	(✓)	✓	✓
Hydrolysis	✓	✓	
Solubilisation	(✓)	✓	
Dechlorination	(✓)		
pH manipulation	(✓)	✓	✓

Chemical treatment systems

Chemical treatments are applied more commonly to polluted groundwater, although soil slurries are amenable to this approach (Armishaw et al. 1992). A range of chemical processes have been applied to soil (Table 2) to destroy or convert pollutants into less toxic forms, to extract them or to immobilise them. Recent research has focussed on developing two of these options, oxidation/reduction and extraction of pollutants (Mulligan et al. 2001b).

Wood (2001) suggested that chemical treatments can be highly specific for some pollutants [e.g. dechlorination of polychlorinated biphenyls (PCBs) and halogenated alkanes] and are applicable to a range of matrices provided effective mixing can be achieved. However, introduced chemicals can pollute soils if not completely reacted and cannot provide effective treatment for a range of common pollutants that are similar in chemical structure to indigenous soil organic compounds.

In several studies, Liang et al. (2003, 2004) investigated the oxidation of trichloroethylene (TCE) in soil water by the sulphate free radical (SO_4^{-}). High temperatures (>40°C) and chelated ferrous iron were found to promote persulphate decomposition and, therefore, TCE degradation. However, it was noted that the effectiveness was reduced at higher soil organic contents. This interaction between chemical oxidants and indigenous organic matter is not a simple one. Thus, Bogan and Trbovic (2003) found that the susceptibility of PAHs to chemical oxidation was related to indigenous organic matter levels in soils with higher organic contents, but that soil porosity was an important factor in low organic matter soils. They also noted that the relative contribution of porosity-mediated protection to sequestration increased with ageing and was greater for three to four ring PAHs than for higher molecular mass PAHs.

Ozone has also been evaluated for contaminant oxidation. Pierpoint et al. (2003) found that ozone rapidly degraded aniline and to a lesser extent trifluralin in moist soils; ozonated water improved treatment rates for trifluralin in soil columns but had little additional effect on pollutants with higher aqueous solubility such as aniline. Transport of gas-phase ozone can be retarded, however, by reaction with soil organic matter and by dissolution (Kim and Choi 2002). Under real site conditions, heterogeneity in these factors and in the distribution of contaminants is likely to lengthen treatment times.

A further oxidation treatment used in aqueous or slurry systems is based on Fenton's (hydrogen peroxide catalysed by iron) reaction, which produces hydroxyl radicals. Weeks et al. (2000) used this approach to treat TCE-contaminated soils, but as with other chemical oxidation processes, they found that indigenous soil carbon 'competed' with TCE in the oxidation process.

In some cases, reduction reactions can contribute to remediation of pollutants in soils. Chemical-reductive dechlorination of common organic pollutants (e.g. PCBs) involves the cleavage of chlorine atoms from the compound by reagents such as alkali polyethylene glycols (Felsot 1996). Reduction of chromium [Cr(VI) to Cr(III)] eliminates the toxicity of this soil pollutant and can be achieved by addition of labile organic carbon that stimulates reduction reactions mediated by microbial activity (Tokunaga et al. 2003).

As noted previously, addition of surfactants can enhance extraction of pollutants. Wang and Mulligan (2004) reviewed the potential of surfactant foams for soil remediation, emphasising the importance of surfactant type and concentration; they also noted that these approaches may be used alone or as a means of augmenting other treatment systems. Pentachlorophenol removal by Triton X-100 was more effective than for a biosurfactant (JBR425), with optimum application rate of 1%; surfactant foam was twice as effective as liquid surfactant solutions (Mulligan and Eftekhari 2003). Shin et al. (2005) investigated desorption of cadmium using various non-ionic surfactants combined with an iodide ligand. Their findings suggested that desorption was achieved primarily by the ligand but that the extracted cadmium was then stabilised within the surfactant micelle; at excess concentrations, the surfactant appeared to block ligand access to soil surfaces.

Organic and other solvents have also been used to extract pollutants before subsequent treatment. Nam et al. (2001) obtained extraction efficiencies >90% for PCBs and other persistent organic pollutants using mixtures of alkanes and alcohols; PCBs were subsequently degraded by either chemical dehalogenation or gamma irradiation. Khodadoust et al. (1999) used ethanol to achieve 98% removal of pentachlorophenol from soil at a wood treatment plant. The application of sodium perborate solutions to soils enhanced hydrolysis and degradation of some organophosphate compounds and was found (David and Seiber 1999) to be more effective than the surfactant Triton X-100.Supercritical fluid extraction, using highly compressed gases (e.g. propane and butane), exploits their improved viscosity and diffusivity compared with other liquids to achieve efficient pollutant extraction (Armishaw et al. 1992). Sahle-Demessie and Richardson (2000) found that supercritical fluid extraction (using CO_2) was as effective as methanol extraction and low-temperature, thermal desorption in removing a range of pesticides from soil.

Other chemical approaches aim to manage the risk associated with soil pollutants by manipulating soil chemistry or providing additional binding sites to immobilise pollutants. A range of amendments has been adopted to manage metal behaviour, including addition of liming materials, phosphate compounds and biosolids as reviewed by Bolan and Duraisamy (2003). Mechanisms include increasing metal adsorption through higher surface charge, formation of insoluble metal complexes, precipitation and redox reactions leading to immobile valency forms. Whilst such approaches do not secure pollution ‘clean-up’, they offer a cost-effective means of managing pollutant behaviour to reduce risk of water pollution and to establish vegetation cover. This cover will stabilise the soil surface but may act as a pathway for transfer of contaminants into the wider ecosystem via grazing herbivores.

Immobilisation approaches are favoured where pollution covers an extensive area and where the main targets to be protected are water resources and plants. Aguilar et al. (2004), for example, considered that the extensive nature of soil pollution after the Aznalcollar mine incident in Spain meant that spoil removal then chemical immobilisation of metals in underlying soils was the only feasible option. Monitoring over a 3-year period indicated that applications of alkaline and $CaCO_3$-rich sugar refinery waste caused a marked reduction in water soluble Cd, Cu and Zn but that immobilisation of As required additions of materials rich in clays and iron oxides. In coarse-textured soils, there was a marked increase over time in soluble metals associated with decreases in soil pH, indicating a need for ongoing management of acidity in these soils. Controlling metal mobility by pH management presents particular problems where pyrite and other metal sulphides have the potential to generate acidity (Bilek 2004). Also, the limited range of targets protected by these approaches is underlined by findings of Basta et al. (2001). Working with smelter wastes polluted with Cd, Pb or Zn, they found that a range of soil amendments, especially alkaline biosolids, reduced the extractability and phytotoxicity of these wastes. However, only rock phosphate reduced gastrointestinal availability and then only for Pb.

In soils polluted with both metals and organic compounds, reductions in metal bioavailability may allow more effective microbial degradation of the organic contaminants. Thus, Yoo et al. (2004) demonstrated that binding to organobentonite moderated metal effects on microorganisms, thus, promoting phenol degradation.

Biological treatment systems

Microorganisms, soil invertebrates and plants have all been exploited as potential agents of soil bioremediation, although most treatments have been based on microbial

Table 3 Main biological remediation methodologies and their effects on soil pollutants (✓ = main process, (✓) = subsidiary process limited in extent or in the range of pollutants affected)

Process treatment	Destruction/ degradation	Extraction/loss	Stabilisation
Microbial activity			
Landfarming	✓	(✓)	✓
Biopiling	✓	(✓)	✓
Composting	✓	(✓)	✓
Bioreactor	✓		(✓)
Bioleaching		✓	
Plant activity			
Phytostabilisation	(✓)	(✓)	✓
Phytoextraction	(✓)	✓	(✓)
Phytodegradation	✓	(✓)	(✓)

activity. Bioremediation may involve augmentation of indigenous populations or the manipulation of environmental conditions to enhance the effectiveness of these populations (Ritter and Scarborough 1995). Zhou and Hua (2004) have reviewed in some detail progress and future developments in bioremediation. Although biological solutions to inorganic pollution exist, the main emphasis has been on the treatment of organic compounds.

The ability of microorganisms to degrade xenobiotic organic compounds derives from their co-evolution with naturally occurring compounds that have analogous molecular structures and by the development of catabolic activity through adaptation on sites subject to extended periods of pollution (Semple et al. 2003). Degradation mostly involves consortia of microorganisms and may be achieved using in situ, on-site or bioreactor approaches (Table 3). On-site processes are likely to involve solid-phase approaches such as landfarming (spreading of polluted materials on soils usually combined with cultivation and/or nutrient inputs) or more 'engineered' solutions such as composting or the use of biopiles (piles of polluted soils constructed to facilitate aeration and addition of nutrients). The management of the treatment environment is easiest in bioreactors and most difficult with in situ approaches, with compost and biopile approaches intermediate in this respect. In most of these treatments, there will be some loss to the atmosphere through volatilisation and some physical or chemical stabilisation of pollutants.

Where pollutants are accessible to microorganisms, in situ approaches can be effective but as this accessibility declines, bioreactor treatments are increasingly favoured. Wilson and Jones (1993) considered in situ techniques to have limited effect for most PAHs, on-site solid-phase treatments such as landfarming or biopiles to be capable of treating PAHs with up to three to four rings, whereas even soil slurry treatment in bioreactors cannot always achieve treatment of some of the higher molecular mass and more hydrophobic PAHs. Degradation of high molecular mass PAHs may rely on fungal activity, whereas lower molecular mass compounds are degraded by bacteria; optimal conditions will vary for these microbial groups. Zhang et al. (1995) also concluded that slow desorption rates could restrict rates of aromatic hydrocarbon degradation during in situ bioremediation. However, the extent to which biodegradation of organic pollutants in soils is controlled by desorption rates varies with different compounds and soil organic contents (Huesemann et al. 2003). Whereas *n*-alkanes appeared to be susceptible to microbial metabolism even when not in the aqueous phase, the degradation rates of most two- and three-ring PAHs were controlled by desorption kinetics. Johnsen et al. (2005) emphasised the dynamic nature of bioavailability for pollutants with low aqueous solubility and high solid–water distribution ratios such as PAHs. They argued that biodegradation rate is controlled by microbial catabolic capacity when mass transfer from inaccessible to bioavailable pools is high; in many soils, PAH catabolic capacity is high but degradation rates are limited by PAH flux to the microorganisms. Microbes can increase this flux by excretion of biosurfactants or bioemulsifiers and the formation of biofilms which might also accumulate organic pollutant pulses for subsequent degradation. Fungal exoenzymes probably initiate the degradation of high molecular weight PAHs and these may diffuse to PAHs.

Bioaugmentation (addition of cultured microorganisms with the capacity to degrade target contaminants) or biostimulation (addition of nutrients to increase indigenous biomass or of substrates to promote co-metabolism) of soil microbial populations may provide a means of accelerating pollutant degradation. Singer et al. (2005) stated that inoculum survival remains the 'Achilles' heel' of bioaugmentation. Gentry et al. (2004) proposed several ways of improving survival, including encapsulating cells in a carrier, incorporating them with a plant that serves as a niche for the inoculant's growth and engineering remediation genes directly into a plant without microbial inoculation.

Whilst biostimulation with nutrients has often resulted in a more rapid onset of biodegradation (e.g. Margesin et al. 2003), several studies (e.g. Sarkar et al. 2005) have found that treatment rates converge with time, with no marked improvement in overall treatment outcome compared with natural attenuation. Other studies have found inconsistent responses to both bioaugmentation and biostimulation compared with natural attenuation (e.g. Bento et al. 2005). However, for the more recalcitrant pollutants such as benzo (*a*)pyrene, Kanaly and Bartha (1999) found that substantial mineralisation occurred only with suitable hydrocarbon supplementation. These hydrocarbons served as primary substrates but also enhanced bioavailability through dissolution of the pollutant.

Romantschuk et al. (2000) have listed several environmental factors that limit microbial biodegradation of soil pollutants including low temperatures, restricted activity under anaerobic conditions, low levels of available nutrients or co-substrates and limited bioavailability of pollutants. Yeung et al. (1997) found that heating soils contaminated with crude oil in a bioreactor from 20 to 35°C reduced the half-life of hydrocarbons from 248 to 105 days; forced aeration caused a reduction to 182 days at 20°C but had no effect at 35°C where aeration was not presumably a limiting factor. Whilst most organic soil pollutants are degraded faster under aerobic conditions, reductive dehalogenation of chlorinated aliphatic and aromatic compounds appears to be an important initial step in their decomposition that is mediated by anaerobic microorganisms (Ritter and Scarborough 1995). Acting as a microbial consortium, fermentative bacteria generate hydrogen which acts as an energy substrate, via oxidation, for dehalogenating bacteria.

Whilst most applications based on microbial activities have focussed on organic pollutants, several studies have emphasised their influence on the behaviour of metals. Gadd (2004) reviewed this topic and concluded that microbial production of complexing metabolites or siderphores and methylation can aid extraction of some pollutants; this process of 'bioleaching' was considered an effective alternative to chemical extraction processes. Immobilisation by formation of insoluble salts (e.g.

sulphides) has been widely applied. For example, Jong and Parry (2005) investigated the stability and leachability of As immobilised by microbial sulphate reduction. Their results indicated that As released from solids was <300 μg l^{-1}, below the current maximum Australian leachate test limits; leaching over 68 days indicated a relatively slow release of As with concentrations below guideline concentrations in drinking water.

Bioremediation approaches can be combined with physical treatments (in effect biostimulation) which increase contaminant accessibility or improve physical conditions to enhance their degradation. Volkering et al. (1998) reviewed the role of surfactants which accelerated the breakdown of hydrophobic organic pollutants such as PAHs in soil. Whilst surfactants generally stimulate mass transfer from solid to aqueous phases, they noted the complex nature of microbial responses with examples ranging from growth inhibition to their use as co-substrates. Similarly, electrokinetic treatment of polluted soils can enhance biodegradation by mobilising organic contaminants, spreading indigenous bacteria and added nutrients and raising soil temperature (Alshawabkeh et al. 1999). In a novel approach to counteracting sequestration of pollutants by sorption or partitioning into non-aqueous phase liquids, Tungittiplakorn et al. (2005) mixed nanoparticles made from poly(ethylene) glycol-modified urethane acrylate with phenanthrene polluted soils; treatment increased the effective solubility of this pollutant and its mineralisation rate.

Although soil animals are not thought to have a significant direct role in the biological degradation of contaminants (Haimi 2000), their activities may stimulate microorganisms and improve the soil environment for microbial degradation. Soils on many polluted sites are physically degraded and macro-fauna such as earthworms can improve these conditions (Scullion and Malik 2000). There has been interest recently in the role of earthworms as ‘bioreactors’ for degradation of contaminants such as trinitrotoluene (Renoux et al. 2000). However, given the small size of many contaminated sites (aiding natural colonisation) and the timescales for treatment, it seems unlikely that inoculation of soil fauna would be included in remediation programmes other than those based on longer-term attenuation.

The term phytoremediation has been applied to a range of processes involving the use of plants to stabilise, extract or promote the degradation of soil pollutants (Table 3); in practice, plants will promote all three processes to a greater or lesser extent depending on species. It should be emphasised that phytoremediation is a longer-term approach to soil remediation, having much in common with enhanced or monitored natural attenuation options. It is not normally applicable in situations where contaminated sites are being remediated for redevelopment or where there is an acute risk to humans or the environment.

Much of the early research in this area related to phytoextraction of metals with foliage harvested and processed elsewhere, as reviewed by Khan et al. (2000). To be a viable strategy, plant species with a rapid biomass gain and high metal uptake are needed. Metal uptake rates are often restricted by their limited availability (Scullion 2003), although this may be enhanced by release of phytochelators (McGrath et al. 1997) or by plant root mycorrhizal associations (Oudeh et al. 2002). Susarla et al. (2002) emphasised that phytoremediation is potentially cost-effective, but that prescriptions must be site-specific.

One method of increasing metal availability and improving phytoextraction is through the application of chelating compounds to soils. Both Blaylock et al. (1997) and Epstein et al. (1999) reported markedly enhanced uptake of lead by Indian mustard (*Brassica juncea*) after soil treatment with ethylenediamine tetraacetic acid. However, several studies have observed enhanced metal leaching, often in excess of metal uptake, with applications of chelating agents to soil (Romkens et al. 2002; Scullion 2003), indicating their potential for polluting groundwater. An alternative approach is to employ the so-called hyperaccumulator plants (Robinson et al. 1997) that actively enhance metal availability in their rhizosphere (McGrath et al. 1997). Several genera of plants (e.g. *Thlaspi* and *Alyssum*) contain species capable of accumulating percentage concentrations of metals. Many of these are unfortunately small and fairly slow growing. It is also uncertain whether the high rates of metal offtake reported in short-term studies can be sustained over the time taken for effective remediation of polluted soils and few plant species have developed tolerance to a range of potentially phytotoxic metals.

There has been interest more recently in the effects of plants on degradation of organic contaminants (Alkorta and Garbisu 2001; Collins et al. 2002). Kirk et al. (2005) measured an increase in rhizosphere bacteria in petroleum-polluted soils planted with perennial ryegrass and alfalfa; there were also evidence from denaturing gradient gel electrophoresis analysis of shifts in microbial composition promoting hydrocarbon degradation. Kim et al. (2004) found a relationship between enhanced degradation of anthracene and the effectiveness of root exudates from different plant species in mobilising this pollutant. The effects of plants on degradation of organic pollutants can vary with soil and pollutant type; Chekol et al. (2002), for example, found that trinitrotoluene degradation was enhanced by plants only in a lower organic matter soil but that pyrene was unaffected. Also, Pradhan et al. (1998) found that plants growing in former gasworks soils caused a much more pronounced reduction in PAHs for moderately contaminated soils without pre-treatment than when used as a ‘polishing’ treatment for more heavily contaminated soils pre-treated to reduce their initial PAH concentrations. These findings may indicate that phytoremediation is less effective in promoting degradation of the more inaccessible contaminant fraction.

The use of coppice-harvested trees, particularly *Salix* and *Populus* species, in phytoremediation has been proposed especially where pollution is present at depths below the normal root zone of alternative species. A number of studies (e.g. Pulford et al. 2002; vanDecasteele et al. 2005) have shown the potential of selected clones of *Salix* for phytoextraction of certain metals. However, as with other phytoextractors, tolerance was not demonstrated for all metals investigated. Laureysens et al. (2005) found significant variation between *Populus* clones and between different metals as to where they accumulated and whether they were accessible for harvesting. Also, as noted earlier, the poor physical condition of many polluted soils may restrict the depth and proliferation of plant roots (Scullion and Malik 2000).

Karenlampi et al. (2000) reviewed the potential for genetic engineering of plants for improving phytoextraction of metals. They note that there have been few attempts to develop these technologies, partly due to the complexity of the process in practice, and emphasise that metal tolerance and accumulation are independent properties, so both would need to be engineered for a successful outcome. In principle, similar approaches might be adopted to promote degradation of organic pollutants (Gentry et al. 2004). The use of genetically modified plants for remediation purposes may avoid many of the perceived risks associated with crops, in that genes from these modified plants are unlikely to confer any competitive advantage should they pass into general plant communities.

Improvements in phytoremediation performance, particularly for water-soluble and/or volatile organic pollutants, have been attributed to the activities of plant endophytic bacteria. These bacteria found predominantly in plant roots are thought to degrade pollutants present in the transpiration flow and, therefore, reduce their phytotoxic effect. However, Newman and Reynolds (2005) noted that naturally occurring bacteria do not necessarily possess specific pollutant-degradative capability nor are those with this capacity capable of thriving as endophytes. Several investigators have used endophytic bacteria engineered to degrade pollutants. For example, Barac et al. (2004) were able to demonstrate increased degradation and host plant tolerance of toluene, coupled with a 50–70% reduction in toluene evapotranspiration using this approach. Newman and Reynolds (2005) emphasise the promise of these developments, in that genetic engineering of bacteria is easier than for plants and that competitive pressure might be less within plants compared with soil. However, they note that engineered bacteria would have to persist for the prolonged periods needed for phytoremediation.

Conclusions

Effective remediation of polluted soils requires accurate information on the distribution and behaviour of pollutants as they interact with soils and the broader environment. Zhang et al. (1998) considered that predicting bioavailability and mass transfer limitations on bioremediation in heterogeneous natural systems remains a major challenge. Indeed, collecting data for such predictions is difficult for many sites. Assessing remediation progress (Ritter and Scarborough 1995) and efficiency is also important, especially because pollutants are rarely completely destroyed or removed from polluted soils. For example, although treatments for petroleum-polluted soils are well developed and effective, Pollard et al. (2005) identified a need to characterise the risks associated with residual contamination. Adriano et al. (2004) discussed this problem in the context of metals and noted that there are no standard protocols for monitoring affected sites. In recognition of the need to develop a coherent approach, International Organization for Standardization (2005) is working towards the publication of a Draft International Standard providing guidelines for assessing the bioavailability of soil contaminants.

Although chemical tests are widely used in assessing specific risk pathways (e.g. US-EPA landfill (1990) and precipitation (1995) leaching tests), these are accepted as having their limitations in predicting general biological and environmental impacts. If ‘availability’ assessments can be developed that are reliable and relevant for a wide range of soils, treatment approaches based on ‘bioavailable contaminant stripping’ (e.g. Puschenreiter et al. 2005) might be more widely accepted. However, as noted by Semple et al. (2005), it is important to distinguish between *bioavailability* (freely available at a given point in time) and *bioaccessibility* (bioavailability plus what is potentially bioavailable). The latter is probably the key parameter for regulatory purposes and may be more difficult to quantify than the former. Some of the difficulties inherent to these measurements are illustrated by the findings of Hickman and Reid (2005) who demonstrated close relationships between two extraction procedures and microbial mineralisation of phenanthrene but no such associations with earthworm accumulation of this compound.

Ritter and Scarborough (1995) identified a number of bioremediation research needs including improved understanding of the degradation patterns of recalcitrant organic pollutants and identification of microorganisms capable of promoting their degradation, and better systems for delivering microbes and nutrients to pollutants. For in situ bioremediation, dissemination of soil amendments and accessibility of contaminants are the key factors in determining the success or otherwise of soil remediation (Romantschuk et al. 2000). Many of these comments could be applied equally to chemical and physical treatment approaches including the particular difficulties associated with in situ treatment.

There is also a need to develop new treatment sequences or ‘trains’, particularly on sites with multiple pollution phases and types; these are likely to involve combinations of biological, physical and/or chemical processes designed to improve the reliability, predictability and efficiency of soil remediation. For example, Huang et al. (2004) emphasised the synergistic effect of a multistage approach to remediation of creosote pollution involving various landfarming and phytoremediation treatment phases. This may be particularly important where multiple pollutants are present, not least because of potential inhibitory effects of, for example, metals on microbial activity (Khan and Scullion 1999) and their consequences for bioremediation of organic pollutants.

Many polluted sites do not pose acute risks to humans or the environment and can be remediated over longer time periods. These approaches have been termed ‘extensive treatment technologies’ (Bardos and van Veen 1996) or ‘monitored’ (US-EPA 1999) and ‘assisted’ (Adriano et al. 2004) natural attenuation. Remediation involves many of the techniques discussed earlier (e.g. soil amendments and phytoremediation) and may offer cost-effective risk management coupled with lesser effects on soil quality or ecological function. However, in many cases (e.g. metal pollution), their viability may depend on the concept of ‘bioavailable contaminant stripping’, initially proposed by Hamon and McLaughlin (1999), being accepted by regulators. They also require detailed site characterisation, a clear understanding of pollutant behaviour and ongoing monitoring over extended periods (Mulligan and Yong 2004); they should not be regarded as a strategy for avoiding action. In some situations short term, intensive treatment of pollution ‘hotspots’ combined with attenuation of residual and more diffuse contamination may prove the most sustainable approach.

As noted previously, treatment-based systems rarely achieve complete elimination of contamination. Residual contamination is a particular problem in fine-textured, organic-rich soils, and where contaminants have low aqueous solubility or volatility and high K_{ow} because of their limited accessibility. It is arguable that contaminants in this form pose a limited risk in many situations. Indeed, the outcome after effective treatment of accessible contaminants may be viewed as analogous to that where contaminants have been immobilised.

The evaluation of remediation techniques has focussed on the extent to which they achieve acceptable reductions in the risks posed by pollutants. In many countries, risk assessment takes account of the proposed use of a remediated site either by quantifying variations in risk or by subjective judgement as to these variations; for example, exposure pathways, and therefore risk, are more limited on industrial as compared with residential land. Less attention has been given to impacts on broader soil function and recovery in these characteristics. Yet, restoration of soil function should be the ultimate goal in many remediation schemes, not least in that this is likely to favour long-term control of any residual contamination. The importance of restoring ecological function to soils will of course be subservient to the need to mitigate pollutant risk. Its importance will also vary with land usage and the broader environmental role that remediated soils are expected to perform.

Finally, it must be recognised that soil remediation is not carried out in isolation from economic and social factors. Regulatory regimes have an important role to play in determining economic viability and should take into account principles of sustainability. There remains a challenge in demonstrating that soil treatment-based approaches deliver effective risk reduction, both to the regulatory authorities and to the general public (Syms 1998), if efforts to encourage re-use of affected land are to succeed.

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