Robust biological descriptors of soil health for use in reclamation of brownfield land

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Abstract

Reclamation of contaminated land contributes to environmental health by improving the quality of soil and resolving contamination issues, but descriptors of soil health that can be used to recognise quality, identify problems and define endpoints are currently inadequate. There are limited guidelines as to what constitutes a healthy soil. Although the concept has been well discussed in the context of agricultural and forest soils, different indices are probably required for the environmental constraints associated with brownfield land remediation and the creation of new soils. Implicit to restored soil health is the existence of indicators or monitors of biodiversity, soil sustainability and acceptable risk management. This paper considers the relevant biological descriptors of soil health in the search for well-defined and practicable measures of the functional integrity of soils that can be used for management of brownfield sites undergoing restoration to soft end-uses. A current project in Liverpool aims to provide a toolbox of robust descriptors of soil health for practitioners.

Key words: biological indicators, brownfield, soil health, toolbox

INTRODUCTION

Regeneration of post-industrial landscapes in the UK is rapidly moving beyond hard redevelopment solely for industrial, commercial and housing end uses. There is now wide realisation that brownfield land has real potential to be re-developed for soft end uses such as amenity, recreation and nature conservation (Ling *et al.* 2003). One example of this is a 30 year programme in lowland England, initiated in 1989, to create community forests within the boundaries of which live one third of the country's population (Anon. 2003). The Mersey Forest in NW England, which is the largest of the Community Forests, is already half way toward achieving its target of increasing tree cover from 4% to 12%; more than 25% of the existing tree planting has taken place on brownfield land and this proportion is likely to increase in the future (Putwain *et al.* 2003). In this way, revitalisation of derelict, underused or neglected land that may or may not be contaminated provides effective *in situ* remediation (Dickinson 2000; Dickinson *et al.* 2000; Rawlinson *et al.* 2004; Dickinson and Pulford 2005), with objectives of creating better, cleaner soil, alongside many other potential benefits including support of biological processes and ecological diversity, watershed improvement and sequestration of atmospheric carbon (Bronick and Lal 2005).

There are numerous problems connected to the redevelopment of brownfield land associated with previous, often ill-defined, usage of the sites. Contamination from former industrial activities, waste disposal and aerial fallout presents varying levels of damage and risk. In many urban situations, high soil fertility (often originating from atmospheric N and S deposition), soil compaction (from heavy machinery) and high soil pH (from concrete and rubble) may prevent the successful establishment of native plant communities that are generally poorly adapted to these conditions (Dickinson 2003). In other cases, for example after mining, landfilling or removal of buildings, there is a lack of significant topsoil; these become 'create' sites that rely on the import of topsoil or soil-forming materials such as recycled green wastes, paper pulp and sewage sludge (Fasham 2000). In all these situations and others, practitioners involved in site management and reclamation follow guidelines and standards of best practice (eg. Environment Agency 2004c and BSI 2004) that are entirely driven by identifying and resolving physico-chemical constraints: soil disposal, other engineering solutions, materials import and addition of ameliorants are the *de facto* tools of the trade (Harris *et al.* 1996; Moore *et al.* 2003; Nathanail and Bardos 2004). There is an assumption, which may well be correct, that if the appropriate site conditions are provided, natural processes will take care of the rest (Dickinson 2003). Unfortunately 'the rest' is ill defined, but it is equated to an assemblage of the necessary ingredients for a good quality or healthy soil.

The concept of a healthy soil has been defined as 'the capacity of the soil to function within ecosystem boundaries and to interact positively with surrounding ecosystems' (Larson and Pierce 1991). This has been elaborated into, 'the capacity of soil to function within ecosystem boundaries, to sustain biological productivity, maintain environmental quality and promote plant and animal health' (Doran and Zeiss 2000). Moffat (2003) has comprehensively reviewed the application and evaluation of soil quality criteria in forest soils, while Moffat and Kennedy (2002) have addressed the requirement for the development of national indicators for soil quality. More recently, sublethal biological indicators have been proposed for contaminated land (Environment Agency 2004a), for use in context with a tiered ecological risk assessment framework developed (Environment Agency 2004b).

A better knowledge of biological indicators of soil health is essential for a critical understanding of the relationships between biological, chemical and physical components of soils. In this paper we argue that practitioners require a toolbox of biological soil health descriptors and we consider the realistic ways in which this could be provided.

ASSESSMENT OF SOIL HEALTH: THE BIOLOGICAL REQUIREMENTS

The assessment of soil health can be based on measures of biodiversity or functional processes (Figure 1). Soil biodiversity is probably most important for maintaining ecosystem function in disturbed environments (Bradford and Newington 2002) and can be measured directly as species richness, or as a surrogate measure of biodiversity using standardized procedures (e.g. higher taxa richness, microbial community diversity, testate amoebae, nematode maturity indices). Functioning of soil processes can be measured as soil functional assessments (e.g. enzyme assays, nutrient mineralization, nitrification potential, soil respirometry).

The presence of soil organisms provides the most obvious visual indicator of soil health but surprisingly often even this is not a standard item of soil quality evaluations (van Straalen 2004; Bengtsson 1998; BSI 2004). There are justifiable reasons for this, particularly that (i) there is limited agreement on what organism or groups of organisms are most appropriate, (ii) most groups of invertebrates require high-level taxonomic skills, (iii) soil microbiology requires specialist equipment or approaches, (iv) the choice of potential indicators is immense, and (v) no indicators of soil quality are likely to be applicable to more than a very restricted range of soils. This is unfortunate because, despite a recent renaissance in the subject, soil ecology has received considerable scientific attention for decades as reviewed extensively elsewhere (Bardgett *et al*. 2005; Coleman *et al.* 2004; Doelman and Eijsackers 2004). However, Doran and Zeiss (2000) have suggested that any biological parameter used to assess soil quality or health should meet a number of conditions:

- Sensitive to changes in management practices.
- Correlated with practical soil functions.
- Useful for clarifying ecosystem processes.
- Comprehensible to practitioners.
- Low cost.

BIOLOGICAL INDICATORS

Biodiversity is higher below-ground than above-ground, and ecological processes in the soil are obviously essential for above-ground ecosystem functioning (Copley 2000). Potential biological indicators of soil health are numerous and diverse (Pankhurst *et al.* 1997). Schloter *et al.* (2003) specified that, in order to use faunal groups as indicators successfully, the group should be dominant in all soil types, having a high biodiversity and abundance. It must also have a significant role in the food web, and be both sensitive to contamination and well correlated with beneficial soil functions. The literature provides an extensive starting point in the search for bioindicators that may be most suitable for inclusion in a practitioner's toolbox.

Invertebrate assays

Examples of biological indicators that focus on invertebrate systems include assessment of earthworms, mites, enchytraeids, nematodes and protozoa. Using earthworms as biological indicators has involved more subtle analysis than simply species presence and biomass. Earthworms are the most obvious bioindicators because they are the most visible soil animals and they process large volumes of organic matter and therefore may be related to fertility and functioning of the soil (van Straalen 2004). Earthworm life-history strategies (Bouché (1977), Paoletti (1999)) classify worms according to their distribution and burrowing habits into three ecological groups, namely (a) epigeics; (pigmented surface dwellers and shallow burrowers), (b) endogeics, (produce extensive branching burrows in the organic and mineral layers) and (c) anecics (construct vertical burrows and only visit the surface to collect litter) (Edwards and Bohlen 1996). Epigeics have a high population turnover producing high numbers of cocoons, whilst anecics are at the opposite end of the scale.

Oribatid mite life-history classifications can be used in relation to their dispersal and reproduction and have been used to compare sites, which have experienced disturbances, such as clear felling of trees (Siepel 1996). Oribatid mites (Acari: Oribatida) are one of the most abundant microarthropod groups in soils acting as detritivores playing an essential role in nutrient cycling, particularly of calcium **(**O'Connor, 2003**)**.

Enchytraeid worms contribute to the cycling of carbon and nitrogen in soils, feeding indiscriminately on fungi. Enchytraeids decrease the mineralisation of persistent organic contaminants (polycyclic aromatic hydrocarbons), and appear to enhance contaminant sequestration to soil particles (Uffindell *et al.* 2005). Enchytraeids from unpolluted soils have shown avoidance behaviour to heavy metal pollution (Salminen and Haimi 2001). Thus, at sites where enchytraeids would be expected to be present, their absence may be related to degraded or stressed soils.

Most nematode work has focused on indices, feeding groups or functional groups (Yeates and Bongers 1999). The Maturity Index (MI) (Bongers 1990) has been used to show that nematode species respond differently to disturbance and stress. The MI is similar to an ecological r-K dichotomy, ranging from fast colonisers to less invasive but more persistent groupings (c-p 1-5), thus classifying nematodes in relation to their generation times, reproduction rates and feeding groups. Group c-p1 has the fastest generation times egg production and metabolic rates, whilst c-p5 are the larger loner lived nematodes. Sensitivity to pollution increases with each group, with group c-p5 being most sensitive.

Protozoa are essential to the food web because they consume a significant amount of bacterial biomass (Foissner 1999). They are thought to have an essential role in the development of soil from sterile substrates, producing a vital link between prokaryote and ciliate pioneers and the later invertebrate colonisers (Smith 2002), whilst they are also important components of earthworm nutrition (Bonkowski and Schaefer 1997). Protozoa are only rarely used as indicators of soil health, but identification of particular groups to genera level such as testate amoeboid protozoa would still provide meaningful data that could be easily recorded by non-specialists (Wilkinson and Davis 2000).

A biological index of soil quality (Qualità Biologica del Suolo (QBS)) (Parisi *et al.* 2005) has been proposed that appoints scores to soil microarthropods (Figure 2). Scores are assigned to each microarthropod found in a soil sample of between 1-20, which denotes an eco-morphological index (EMI) according to its adaptation in the soil (Parisi *et al.* 2005). The higher the QBS index the greater will be the soil quality, as microarthropods will be better adapted to the soil conditions. The QBS focuses on the morphological characteristics of soil invertebrates, which show adaptations to the soil conditions in which they live. This approach does not require knowledge of identification to species level and is therefore accessible to non-specialists (Parisi *et al.* 2005).

Microbial assays

Fungi and bacteria are increasingly popular as bioindicators for the assessment of soil health because of their intimate relationships to soil and plant health (Figure 3) and in the sustainability of ecosystems (Doran *et al.* 1994). Conventional culturing methods, such as community level physiological profiles, have been employed in soil health investigations (Hill *et al.* 2000). More recently, culture independent techniques have been developed which use molecular analysis such as phospholipid fatty acids and denaturing gradient gel electrophoresis to determine microbial diversity (Kirk *et al.* 2004).

Numerous assessments exist for microbial indicator systems, which generally fall into measures of either biodiversity or soil function (Nannipieri *et al.* 2003). Biodiversity can simply describe species richness, and relative abundance (evenness) in soil communities (Nannipieri *et al.* 2003). Mycorrhizal activity in relation to root activity has also been assessed and Schloter *et al.* (2003) suggested that because of the complexity of the microbial community, indicator species such as arbuscular mycorrhiza and *Rhizobium* could be linked to soil quality.

Microbial biomass may be one of the few biologically meaningful fractions in soil (Schloter *et al.* 2003). A chloroform fumigation technique for determining biomass has been widely used although it may provide limited data for determination of microbial activity (Nannipieri *et al.* 2003). Using different indicators such as enzyme activity, bacterial diversity, richness and density appears to be more effective and revealing than the analysis of single parameters, and a combination of microbial indicators may be a more useful tool for measuring soil health (Avidano *et al.* 2005). More advanced biomolecular techniques such as PCR may provide useful tools in the future, but standardised procedures with wide applicability require further development.

Functional processes, perturbations and resilience

Functional processes can be assessed using soil respiration (CO_2 evolution) and enzyme activity. Differentiating between microbial respiration and plant roots in the field may be problematic, as well as high variation associated with season and system (Dilly *et al.* 2000). Examples of enzyme assays include protease, chitinase and polyphenol oxidase (for carbon cycling) and dehydrogenase and urease (for nitrification and N-fixation (N-cycling)).

Applying stress factors (wetting/drying, excess nutrients or heavy metals) to soil samples whilst monitoring changes that occur in their microbial populations, may provide a better indication of soil health than measurements obtained in more steady-state conditions (van Bruggen and Semenov 2000). After applying a stress factor observation of succession could be monitored such as r- to K- strategists. The authors also suggested that traditional methods for analysing microbial communities, for example fumigation-extraction (biomass) and respiration (microbial activity) were not effective, and overlooked important factors such as the shift from eutrophic to oligotrophic conditions, which they considered to be an important attribute of soil health. By considering the ratio of copiotrophic (fast growing microbes, 2 days) to oligotrophic (slower growing microbes, 7 days) bacteria after stress factors had been applied, changes in bacterial populations could be observed, increasing with stress and decreasing with a return to stable conditions (van Bruggen and Semenov 2000).

CONTAMINATION AND RISK ASSESSMENT

Brownfield soils are often contaminated and this introduces an element of ecotoxicology and risk assessment into the derivation of soil health criteria. The working definition of contaminated land is concerned with identification and remediation of land where contamination poses an unacceptable risk to human health or the environment (DETR 2000). Management of this risk largely concerns breaking the source-pathway-receptor linkage, as discussed elsewhere (Kearney and Herbert 1999; Dickinson *et al.* 2000). Currently we rely to a large extent on chemical testing prior to reclamation, but analyses of total concentrations of heavy metals frequently provide a crude and inaccurate estimate of risk; it is the availability not the total amount of metal that is important. Despite this, there is still little standardisation of protocols to determine bioavailability (Kearney and Herbert 1999; Nolan *et al.* 2003). Current guidelines largely refer to total metals and probably originate from those derived for agricultural soils subjected to cumulative inputs from agricultural chemicals and waste disposal.

In the context of designating an end-point for the remediation of contaminated soils, we really should determine whether the soil created is healthy and sustainable; a soil within which any residual contamination poses negligible risk to human health or the wider environment (Oliver 1997; Dickinson *et al*. 2000). Ecological Risk Assessment is a relatively new approach to quantifying the risk of significant harm to organisms and their ecosystems with particular focus on nature conservation areas (Environment Agency 2004). The assessment consists of a tiered framework for determining the risk of harm to defined eco-receptors and uses biological and ecotoxicological tests for measuring harm. However evaluation of soil health on brownfield land really requires a broader perspective.

THE WAY FORWARD

The contribution of large-scale field experiments is required to provide vital evidence of what constitutes a healthy soil in relation to biological or ecological indicators but, for many reasons, any derived index of health must be approached with caution. For example, contaminated or degraded sites may sustain a community of edaphic organisms adapted or acclimated to the stress but different to those of a cleaner or undisturbed soil. In the future it may be possible to provide a workable definition of our concept of a statutory healthy soil for brownfield sites remediated to soft end-uses to differentiate from more natural ecosystems.

Guidelines need to take account of land-use criteria; different soil types quite naturally have different biotic components and assemblages of organisms. What constitutes a healthy soil on agricultural land may be different to that for forestry, conservation or housing development. Baseline surveys can provide basic information about soil quality, (e.g. pH, organic matter, nutrient status, heavy metals) and more refined chemical analyses such as leaching tests (Hartley *et al*. 2004). Further to this, a suite of bioindicators may determine soil health in the context of intended final land-use, or suitable-for-use.

Indicators of soil health are required that can be routinely measured, providing quantifiable data that can be understood and used by practitioners involved in land reclamation and policy making. Relatively simple indicators that are credible and realistic; identification of invertebrate groups to species level, or state-of-the-art microbial techniques may be somewhat out of reach in relation to practicality and cost.

A tiered approach

A tiered approach to the use of soil health criteria by practitioners is clearly required based on a sequence of physico-chemical analysis, ecological surveys and bioassays, similar to that used for ERA of contaminated land (Environment Agency

2004). Preliminary investigations using established physico-chemical analyses would determine whether to proceed with biological tests.

The Liverpool Healthy Soils Project has received its first years funding from WRAP (the Waste Recycling Action Programme). Within the next 6 months at Liverpool John Moores University we aim to compile and provide a robust and relatively simple toolbox of soil biological indicators appropriate for assessment of land that is either contaminated or despoiled which is undergoing restoration to soft end-uses, for example to community forestry. The derived toolbox will be tested on a range of established brownfield sites at different stages of restoration in NW England. If successful this will be a step towards providing guidance to environmental practitioners that can be used in conjunction with established site investigation methodologies. Hopefully it will also contribute to the broader national agenda concerned with healthy soils (Environment Agency, 2002; National Trust, 2003; DEFRA, 2004; European Commission, 2004).

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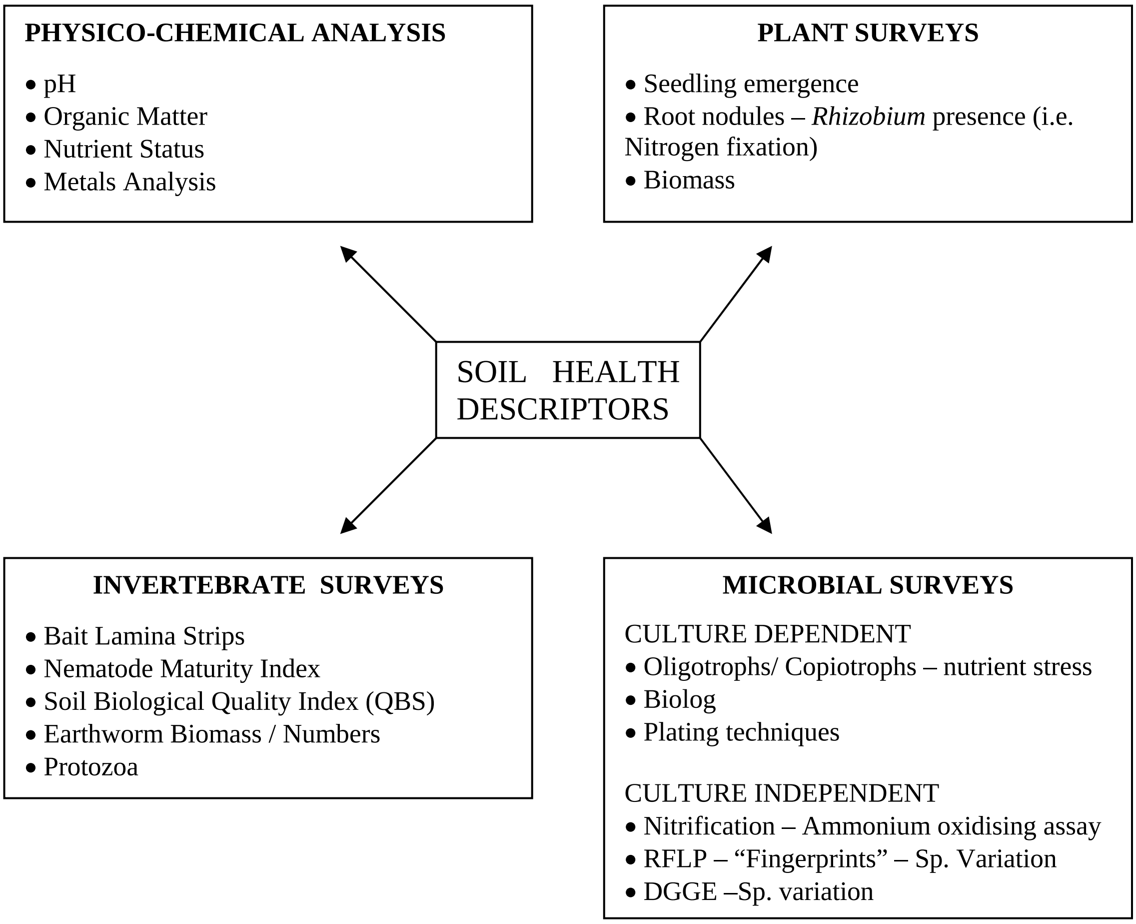

Figure 1. Examples of indicators that may be useful to assess the health of a soil.

Figure 2 A collembolan (springtail) common in UK soils. Photograph, L.A.Uffindell

Figure 3 Fungus (*Cortinarius* sp.) with Scots pine roots. Photograph, L.A.Uffindell