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Soil health in agricultural systems

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Soil health is presented as an integrative property that reflects the capacity of soil to respond to agricultural intervention, so that it continues to support both the agricultural production and the provision of other ecosystem services. The major challenge within sustainable soil management is to conserve ecosystem service delivery while optimizing agricultural yields. It is proposed that soil health is dependent on the maintenance of four major functions: carbon transformations; nutrient cycles; soil structure maintenance; and the regulation of pests and diseases. Each of these functions is manifested as an aggregate of a variety of biological processes provided by a diversity of interacting soil organisms under the influence of the abiotic soil environment. Analysis of current models of the soil community under the impact of agricultural interventions (particularly those entailing substitution of biological processes with fossil fuel-derived energy or inputs) confirms the highly integrative pattern of interactions within each of these functions and leads to the conclusion that measurement of individual groups of organisms, processes or soil properties does not suffice to indicate the state of the soil health. A further conclusion is that quantifying the flow of energy and carbon between functions is an essential but non-trivial task for the assessment and management of soil health.

Keywords: soil health; agricultural impact; ecosystem services; biological processes and functions; indicators

1. INTRODUCTION

Soil health is a term which is widely used within discussions on sustainable agriculture to describe the general condition or quality of the soil resource. Soil management is fundamental to all agricultural systems, yet there is evidence for widespread degradation of agricultural soils in the form of erosion, loss of organic matter, contamination, compaction, increased salinity and other harms (European Commission 2002). This degradation sometimes occurs rapidly and obviously, for example when poor soil management leads to gully erosion. Often degradation is slower and more subtle, and may only impact on agricultural production and the wider environment over years. For this reason, research has been directed to devising measures of the health of soil, which could be used to monitor its condition and inform its management so that degradation is avoided. This has led to debate around the question 'What is soil health?'

There are two ways in which the concept of soil health (or the closely related concept of soil quality) has been considered, which can be termed either 'reductionist' or 'integrated'. The former is based on estimation of soil condition using a set of independent indicators of specific soil properties—physical, chemical and biological. This approach has been much discussed and well reviewed (e.g. Doran *et al.* 1994; Doran & Jones 1996; Van-Camp *et al.* 2004). This reductionist approach has much in common with conventional quality assessments in other fields, such as materials science. The alternative, integrated, approach makes the assumption that the health of a soil is more than simply the sum of the contributions from a set of specific components. It recognizes the possibility that there are emergent properties resulting from the interaction between different processes and properties. These aspects do not seem to have been explored to the same extent in recent literature (but see Harris *et al.* 1996). In this paper, we have taken the view that while the reductionist approach is an accessible and practical means of assessing soil condition, progress in understanding the interactions between management interventions and the capacity of the soil to respond depends on insights into its functioning as an integrated subsystem of the agroecosystem. We thus focus on analysing the extent to which soil can be seen to be responding as a living system to agricultural intervention and the implications of this for sustainable agricultural practices.

Our definition of soil health is derived from a context which we accept as an essential feature of sustainable agriculture, namely that agricultural production should not prejudice other ecosystem services that humans require from agricultural landscapes. Thus, our working definition is that 'a healthy agricultural soil is one that is capable of supporting the production of food and fibre, to a level and with a quality sufficient to meet human requirements, together with continued delivery of other ecosystem services that are essential for maintenance of the quality of life for humans and the conservation of biodiversity'.

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One contribution of 15 to a Theme Issue 'Sustainable agriculture II'.

We establish two contextual limits to the following account. Firstly, we make no attempt to review by giving examples and data from the vast amount of information relating to the performance of specific components, including specific biological populations, nutrients, physical conditions, etc. that contribute to soil health (i.e. the reductionist approach), except in so far as they shed light on an understanding of soil health as a system property. Secondly, we do not address socio-economic dimensions in any detail. Farmers, and the weight of factors that influence their decisions, always have a tangible presence in any agricultural intervention. We recognize the reality that any principles proposed for soil health must pass the tests not only of scientific validity but also of practicality and relevance, but we do not presume to evaluate them in this account.

We have considered agricultural systems in which crop and financial yields are maximized by optimal inputs of nutrients and other resources, whether artificial or organic, which we call 'industrial'; as well as those 'subsistence' systems which seek to achieve optimum yields where there is only limited access to industrially derived inputs, including fertilizers and power for tillage.

2. AN INTEGRATED CONCEPT OF SOIL HEALTH

(a) *Soil as a system*

Soil is undeniably a very complex system. It may be described as a multicomponent and multifunctional system, with definable operating limits and a characteristic spatial configuration. Within a continuum of possibilities, there are recognizable soil types that originate depending on variations in factors, such as parent material, climate and topography, which largely determine the dominant physical and chemical properties. These have often been altered, however, by agricultural interventions, such as drainage, irrigation, use of lime to alter soil reaction and additions of plant nutrients. The agricultural soil system is a subsystem of the agroecosystem, and the majority of its internal functions interact in a variety of ways across a range of spatial and temporal scales. As an example at the submillimetre scale, soil microbes modify soil structure by aggregating both mineral and organic constituents via production of extracellular compounds with adhesive properties. Such compounds are produced by bacteria and fungi as a feeding mechanism, to aid colony coalescence, as protective coatings against desiccation and as a means of attaching to surfaces—the aggregation of soil constituents is an inevitable consequence of such material being produced. However, there is an associated change in the local soil structure and topology of the pore network which is the microbial habitat that affects the distribution and availability of water, delivery of substrate and gases to the organisms and removal of metabolic products from their vicinity, with subsequent consequences for the microbial activity. Such microbial activity is fundamentally governed by the availability of fixed carbon (the major 'currency' of the soil system), which is amenable to manipulation via agronomic factors such as crop type, and residue and other organic waste management. Organic matter indubitably plays other important roles in modulating soil functions, for example via the provision of surface charges, expressed as the cation exchange capacity, or influencing hydrological properties such as wettability. However, while the chemistry (and physics) of the soil system provides the context, and indeed sets the limits, in which the biotic assemblages of soils operate, the unique and absolutely crucial feature of the biota is that it is *adaptive to changes in environmental circumstances, driven by processes of natural selection*, in ways that the abiotic systems of the soil are not.

(b) *Services, functions and assemblages of soil*

Humans depend on both natural and managed (including agricultural) ecosystems for a range of what have been called 'environmental (or ecosystem) goods and services' (Daily 1997; Costanza *et al.* 1997; De Groot *et al.* 2002). These include the natural processes that support the production of food and fibre, such as nutrient cycles and the biological control of pests and diseases together with the regulation of water flow and quality, and influence on the gaseous composition of the atmosphere with its implications for the control of the global climate. In reality, these goods and services are functional outputs of biological processes. Soil is a living system and as such is distinguished from weathered rock (regolith) mainly by its biology. Nonetheless, it should be emphasized that these functions operate by complex interaction with the abiotic physical and chemical environment of soil. Both natural and agricultural soils are the habitat for many different organisms which collectively contribute to a variety of soil-based goods and services (Wall 2004). Figure 1 shows the relationships between these and the soil-based biological processes that deliver them. In the third column, these processes are aggregated into four ecosystem functions, which we propose collectively provide the basis for all the major services provided by soil, *viz*.

(i) Transformation of carbon through the decomposition of plant residues and other organic matter, including soil organic matter, together with the synthetic activities of the soil biota, including, and particularly, soil organic matter synthesis. Decomposition in itself is not only an essential ecosystem function and driver of nutrient cycles but also supports a detoxification and waste disposal service. Soil organic matter contributes to nutrient cycling and soil structure maintenance. Sequestration of C in soil also plays some role in regulating the emission of greenhouse gases such as methane and carbon dioxide.

(ii) Cycling of nutrients, for example nitrogen, phosphorus and sulphur, including regulation of nitrous oxide emissions.

(iii) Maintenance of the structure and fabric of the soil by aggregation and particle transport, and formation of biostructures and pore networks across many spatial scales. This function underpins the maintenance of the soil habitat and regulation of the soil-water cycle and sustains a favourable rooting medium for plants.

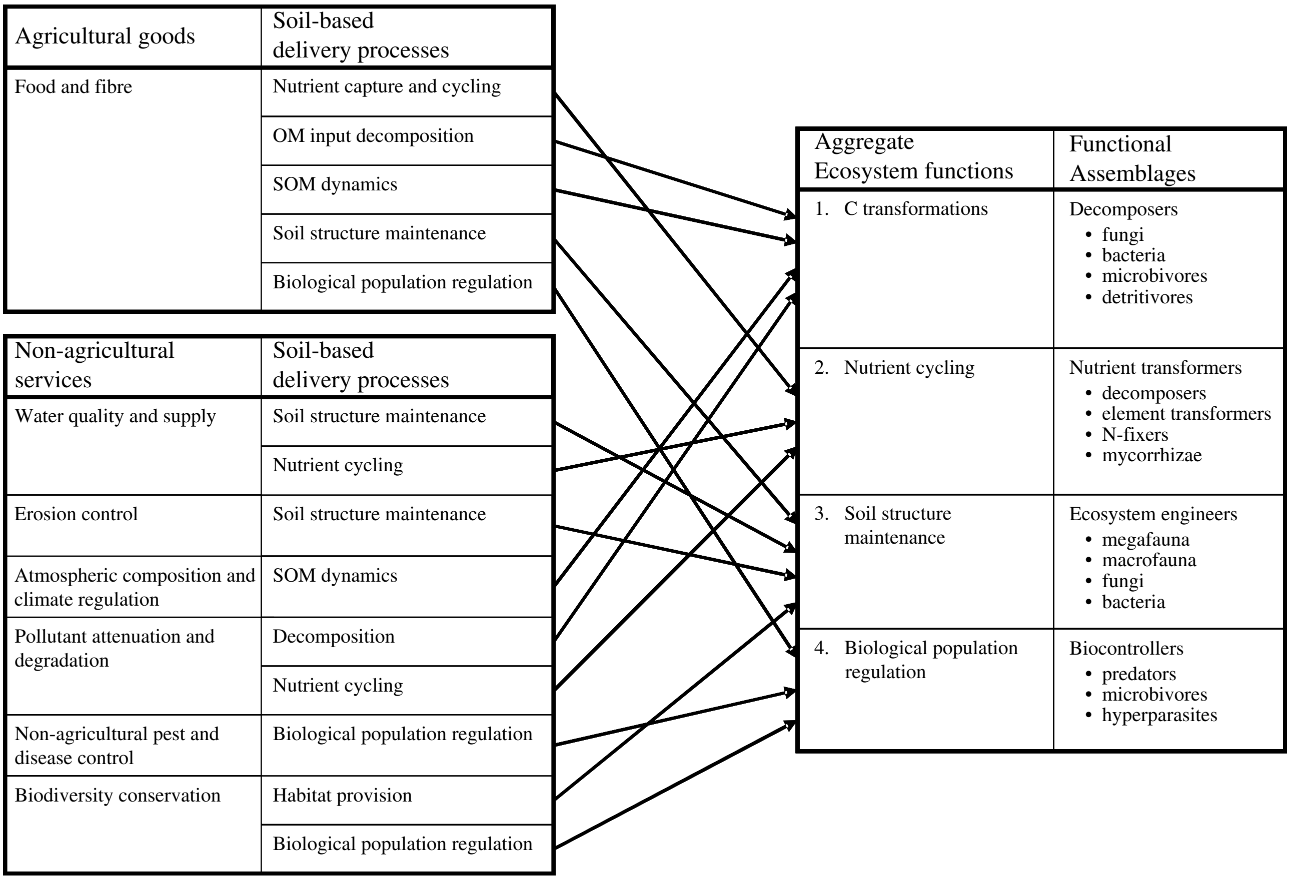

Figure 1. Relationships between the activities of the soil biological community and a range of ecosystem goods and services that society might expect from agricultural soils. OM, organic matter; SOM, soil organic matter.

(iv) Biological regulation of soil populations including organisms recognized as pests and diseases of agriculturally important plants and animals as well as humans.

The biological processes that contribute to these aggregate functions, such as carbon transformation and nutrient cycling, are provided by assemblages of interacting organisms (sometimes termed 'key functional groups'; Lavelle 1997; Swift *et al.* 2004), which are subsets of the full soil community. The major functional groups of organisms contributing to the four aggregate soil functions are given in the final column of figure 1. We propose that soil health is a direct expression of the condition of these assemblages, which, in turn, depends on the physical and chemical condition of the soil habitat.

These assemblages do not, however, operate in isolation but are part of an interactive soil system. The most fundamental integrating feature is that of the feeding relationships between organisms. Soil food web models are based on grouping organisms into assemblages with similar trophic roles (figure 2). These trophic groupings subsume a high degree of taxonomic diversity which also hides a significant extent of variation in functional behaviour. Nonetheless, the general pattern described in figure 2 is one which has been mirrored in a wide range of models of 'detritus-based' systems in the soil (e.g. Hunt *et al.* 1987; De Ruiter *et al.* 1994; Wardle 1995).

The primary agents of decomposition are fungi and bacteria which, in their turn, provide a food source for a variety of microbivorous predators occupying the lower row of the larger box in figure 2. A wide range of experimental studies have shown the importance of these organisms in regulating the rate of decomposition through the release of nutrients and by stimulating microbial population turnover through their feeding activity (e.g. papers in Coleman & Hendrix (2000)). This path of decomposition may be supplemented or diverted by the intervention of larger detritivorous fauna, such as earthworms (shown in the diagram) and other macroarthropods (such as termites in tropical soils) which consume both organic matter and microbes, often together with soil. In figure 2, the rate of decomposition is shown as being regulated by climate (particularly soil moisture content and temperature), soil conditions (pH and particularly habitat structure) and resource quality (particularly in relation to organic matter and the content of nutrients, lignin and polyphenols; Swift *et al.* 1979; Lavelle *et al.* 1997). Soil organic matter is not only a synthetic product of the primary decomposition process but also a substrate for decomposition and mineralization by a subset of the decomposer organisms. In figure 2, it is presented as two fractions differing in their resistance to decomposition, i.e. active (*sensu* labile) and slow passive. These fractions are also hypothesized to differ in their functional roles, with the former contributing more to nutrient cycling and the latter to soil structural features.

Detritus-based food webs have been used for quantitative assessment of transfers of nutrients, energy

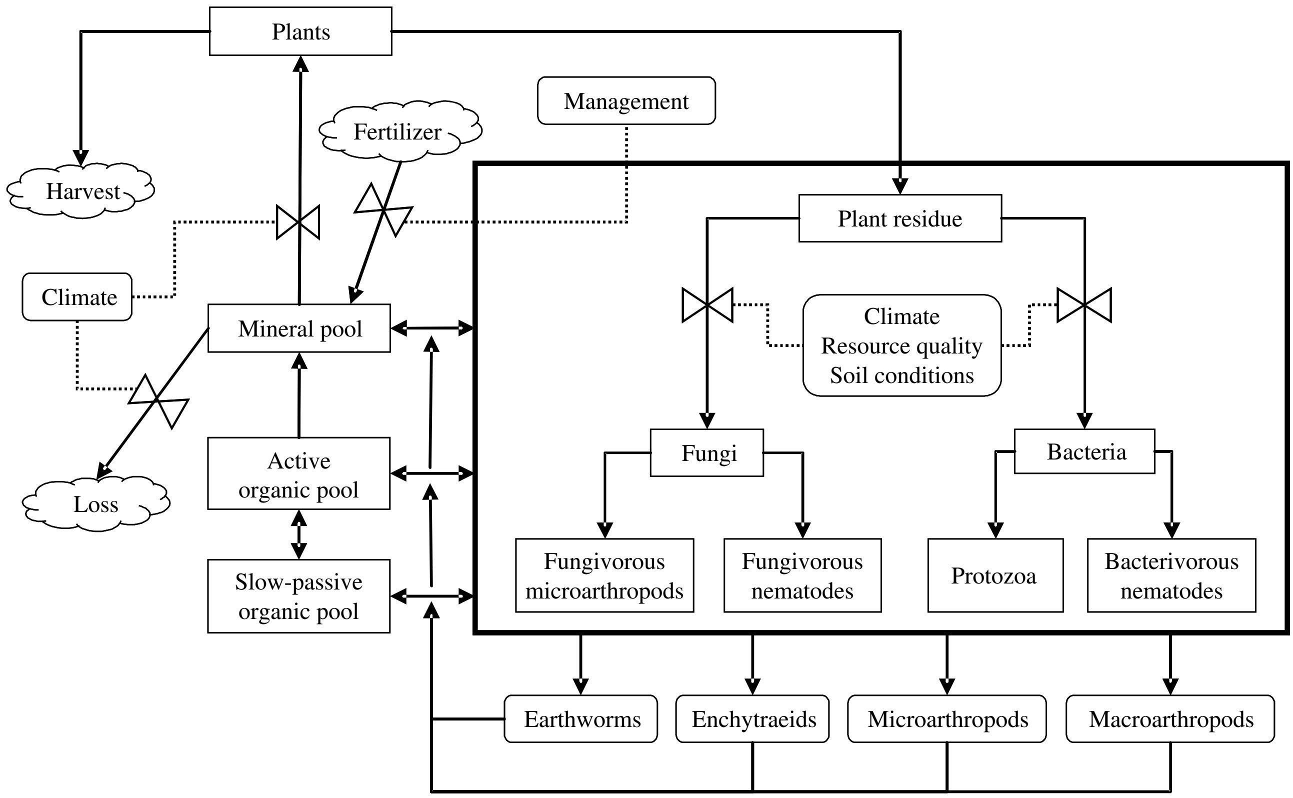

Figure 2. Major trophic relationships in the soil biological community of an agricultural soil under zero tillage (adapted with permission from Hendrix *et al.* (1986)).

or carbon between the designated compartments. However, they are not explicit with respect to other ecosystem functions. Figure 3 hypothesizes relationships between the four major ecosystem functions identified above. In this model the arrows represent flows of energy from the plant and dead organic matter to and between the four different functions. The concept incorporates the activity of parasitic and mutualistic micro-organisms and root herbivores as sources of flow directly from photosynthate and living plant cells into the soil system, features not shown in figure 2. The model implies a high degree of interconnectedness between the functions. This is due to the high frequency with which soil organisms contribute to more than one function, i.e. the assemblages of organisms involved in the different functions overlap to a considerable degree. Thus, while many of the organism groups implicated in figure 3 are the same as those that are explicitly shown in figure 2, the functional model does not map directly onto the trophic model. For instance, many of the species of fungi, bacteria and detritivores, such as earthworms and termites, that participate in decomposition also contribute to soil structure modification and/or nutrient cycling. This makes the simple but important point that decomposition of organic matter is not only a key ecosystem function in its own right but also the main source of energy for driving other functions such as nutrient cycling and soil structure maintenance.

Figure 3 shows that the environmental service of pest and disease control originates in an energy flow path from plant photosynthate, rather than through dead organic matter, although these two energy flow paths converge at higher trophic levels. Predators and hyperparasites that are important in regulating herbivores and parasites may also feed on decomposers including those that influence soil structure modification and nutrient cycling (figure 3). Similarly there is a direct link between these two major energy flow paths through the key roles in nutrient cycling played by the root symbiotic mycorrhizal fungi and nitrogen-fixing bacteria.

This brief analysis of biotic interactions in soil-based ecosystem functions reveals two key points with respect to the management of soil health. First, the overlapping nature of the biotic relationships makes it clear that soil-based functions do indeed comprise a highly integrated system, and that intervention which disturbs any one function will inevitably alter the dynamics of others. The degree of interrelatedness increases at higher trophic levels, suggesting that changes in the biota of these trophic groups may have significant regulatory impacts on the organisms and the processes they perform at the lower levels. This is most obvious with respect to a service such as pest control and also applies to the other functions. In contrast, a second conclusion is that organic matter decomposition, which lies at the lowest trophic level, underpins and is crucial to all the other functions, so any ‘damage’ at this level is likely to have wide implications. This particular connection is least obvious with respect to pest control, but if it is correct that energy from decomposition partially supports populations of animals that can also be consumers of pests, then a decrease or diversion of energy flow from decomposition will also influence pest regulation.

The understanding of the mechanistic basis of soil health would be greatly enhanced by quantifying the

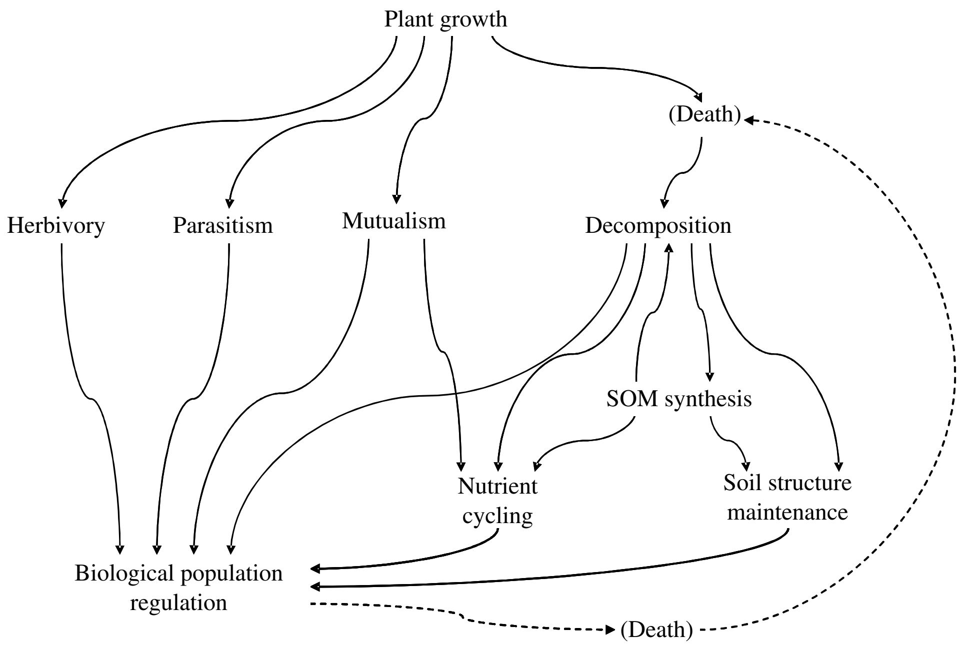

Figure 3. Interconnectedness between the major ecosystem functions of soil. The arrows hypothesize two flows of energy from the plant to the major functions of the soil biota; either directly through the actions of herbivory, parasitism and mutualistic symbiosis, or indirectly via heterotrophic carbon-transforming processes in the soil. Soil organic matter (SOM) synthesis is pictured as supported by energy flowing from the decomposition of plant residues and contributing energy in its turn directly (i.e. by virtue of its properties) to soil structure maintenance and indirectly, through its own decomposition, to nutrient cycling and biological population regulation. See text for further explanation.

flows of energy to and between each of the four aggregate functions and demonstrating how these allocations change under different circumstances, particularly those of agricultural management. The complexity of the functional interactions makes this a major challenge. In order to do this, it would be necessary to build energy flow models for functions such as soil structural dynamics and biological control of soil-borne pests and diseases (as discussed in §5) and then map them onto the established models of nutrient cycling and soil organic matter dynamics to identify the overlaps and convergences. This might prove particularly valuable in identifying keystone species or functional groups which may be susceptible to particular types of soil management. This type of argument has been used to identify organisms such as the Collembola as indicators of change in below-ground food webs (Edwards 2000).

(c) *Soil as a habitat*

The ecosystem services provided by soil are driven by soil biological processes, but our concept of soil health embraces not only the soil biota and the myriad of biotic interactions that occur, but also the soil as a habitat (Young & Ritz 2005). The key concept here is that soil provides a living space for the biota, which is defined by the architecture of the pore networks. Indeed, it is the porous nature of soils that governs so much of their function since the physical framework defines the spatial and temporal dynamics of gases, liquids, solutes, particulates and organisms within the matrix, and without such dynamics there would be no function. The walls of soil pore networks provide surfaces for colonization, and their labyrinthine nature defines how, and to large extent where, organisms can move through the total soil volume. The enormous range in pore sizes affords physical protection mechanisms for prey from their larger predators and organic matter from microbial decomposition. Hence the capacity of the soil biota to deliver ecosystem services may be compromised not only by loss of diversity or impairment of function but also by destruction of the habitat via changes in soil structure and physical–chemical properties. Organisms aggregate the solid constituents of soil, and hence generate structure and associated pore networks. These mechanisms occur across orders of magnitude in scale and involve processes of adhesion, coating, enmeshment, particle alignment and gross movement (Tisdall & Oades 1983; Lavelle *et al.* 1997; Ritz & Young 2004). Biotic activity can also degenerate structural integrity, primarily through the decomposition of organic material that, while it may be a binding agent, also represents energy-rich substrate to a predominantly C-limited biota. The community and the habitat therefore have a two-directional interactive relationship, which encompasses both feed-forward and feedback interactions between the biota and architecture of the soil. These mechanisms lead to the concept that soil may be a self-organizing system (Young & Crawford 2004). The capacity for self-organization can be recognized as an essential component of soil health, which relies on the presence of appropriate constituents and sources of energy to drive biological processes.

3. FACTORS CONTROLLING SOIL HEALTH

(a) *Soil type*

Particular soil types form in response to the nature of parent material, topography and environmental factors, such as climate and natural vegetation. Past land management by humans can alter natural soils considerably, for example by loss of surface horizons due to erosion, alteration of soil water regime via artificial drainage, salinization due to poor irrigation practices, loss of natural soil organic matter caused by arable production or contamination. Thus, land-use and management are the controlling factors for soil health. A set of fixed characteristics such as texture, stone content, etc. combine with climate to set an envelope of possible soil habitat conditions, especially those relating to the soil water regime. Variable factors such as pH, bulk density and soil organic matter content, which are influenced by land-use and management, then determine the prevailing condition of the habitat within the range for a particular soil. These fixed and variable abiotic factors interact with biotic ones to determine the overall condition of the soil system and its associated health. Primary biological factors will include the presence or absence of specific assemblages and types of organisms, the availability of carbon substrate and nutrients, and the concentrations of toxic materials.

(b) *Organisms and functions*

The relationships between community structure and function are inevitably complex and a prevalent theme in contemporary soil ecology (e.g. Schulze & Mooney 1994; Wardle 2002; Bardgett *et al.* 2005). They are underwritten by the three principles of repertoire (i.e. the 'toolkit' of available functions), interaction and redundancy (Ritz 2005). Relationships between diversity and function have been postulated to follow a number of forms (Swift *et al.* 1996; Gaston & Spicer 1998), but rigorous experimental demonstration of these issues is relatively scarce, not least owing to the difficulty in manipulating soil biodiversity as the sole factor (Ritz & Griffiths 2000). There is some experimental evidence that there may be threshold levels of soil biodiversity below which functions decline (e.g. van der Heijden *et al.* 1998; Liiri *et al.* 2002; Setälä & McLean 2004). However, in many instances, this is at experimentally prescribed unrealistically low levels of diversity that rarely prevail in nature. Many studies demonstrate high levels of functional redundancy in soil communities (e.g. Setälä *et al.* (2005) for review). It can be argued that high biodiversity within trophic groups is advantageous since the group is likely to function more efficiently under a variety of environmental circumstances, due to an inherently wider potential. More diverse systems may be more resilient to perturbation since if a proportion of components are removed or compromised in some way, others that prevail will be able to compensate. However, more diverse systems may be less efficient since a greater proportion of available energy is used in generating and countering competitive interactions between components, a situation which may be exacerbated by the similarity in functional properties and hence potential niche competition. All these factors are likely to influence the patterns of interaction between the soil-based functions illustrated in figure 3, and thence the status of soil health.

(c) *Carbon and energy*

The energy that drives soil systems is derived from reduced carbon that is ultimately derived from net primary productivity (figure 3). Carbon is the common currency of the soil system, and its transfer with associated energy flows is the main integrating factor. This suggests that the quantities and quality of different organic matter pools may be indicative of the state of the soil system, while the flows and allocations of carbon between assemblages of organisms may provide information about their relationships to ecosystem functions. However, as shown above, food web models as presently constructed do not explain how different assemblages use carbon to support these functions. Existing models of soil carbon dynamics (Jenkinson & Rayner 1977; Parton 1996) assume the presence of pools of carbon that turn over at different rates. Rapid and medium turnover fractions provide immediate and short-term sources of carbon substrate for the soil biota. More recalcitrant forms that turn over slowly represent long-term reservoirs of energy that serve to sustain the system in the longer term, as well as provide some structural stability. These models are not, however, based on measurable carbon pools or those used by particular assemblages of organisms. Neither are they explicit with respect to allocation to different soil functions. Consequently, their utility for assessing soil health appears to be limited.

The limitations to current models may be diminished by the answers to a range of research questions that emerge from the previous discussion: how might the allocation of soil carbon regulate functional outputs? What quantities and qualities of organic matter are needed to support soil system performance and are the levels and forms of soil organic carbon indicative of soil health? Are there minimum levels of carbon reserves below which long-term soil health is compromised? Are there levels of readily decomposed organic matter which if present continually might indicate reduced carbon transformation and so a lack of soil health? How do the forms and flows of soil carbon to and between different assemblages exert control over the physical condition of the soil habitat, and can this condition be used to infer soil health? In §4 we propose some principles for defining the soil system and assessing its health, as a basis for directing these and other research questions.

(d) *Nutrients*

Nutrients are a controlling input to the soil system and the processes within it. Their levels and transformations are critical to soil health. After carbon, the cycling of nitrogen and phosphorus to, from and within the soil system most affects its dynamics and the delivery of ecosystem services, including agricultural production. Manipulation of nutrient supplies to increase productive outputs from the soil system by the addition of fertilizers has been one of the keystones of agriculture for centuries. Nonetheless, knowledge is limited about the impacts of nutrient additions on the condition of different assemblages of soil organisms and thence on their functions.

Generally, while it is considered that the availability of carbon substrate is normally the primary limiting factor on microbial activity in soils, this is not necessarily the case, and there is accumulating evidence that soil microbes may frequently be N limited (Schimel *et al.* 2005). Where demand for nitrogen is higher than its supply, the functional capacity of the soil system will be strongly influenced by N availability. In undisturbed natural soil systems, inputs from the atmosphere are usually low and losses through leaching or gaseous emissions are also slight because biological demand for mineral nitrogen is high and any that is rendered available is quickly assimilated. When the soil system is disturbed, for example by tillage, losses via leaching or to the atmosphere are increased because mixing of the soil leads to more rapid decomposition of organic matter and the conversion rate of organic nitrogen to mineral forms may exceed the biological demand, particularly where balancing of available nitrogen to plant requirements is poorly managed. Moreover, nitrogen is removed in agricultural produce, often in considerable excess of that which can be replaced by natural inputs. So without balancing additional inputs of nitrogen, and particularly without due consideration of the associated carbon (energy) requirements of the biomass, soil health declines in agricultural systems owing to progressive reductions in the pool of nitrogen available to organisms supporting soil functions and for plant growth. Similarly, the natural pool of phosphorus in soil is reduced through cropping and via other pathways such as erosion, causing a decline in soil health in the absence of any balancing replacement that inherent mineralization (i.e. weathering) can deliver. Agricultural strategies based on additions of animal manures and the use of mineral fertilizers counter losses of nitrogen, phosphorus and other nutrients with the aim of restoring and sustaining soil health. In well-managed systems employing high levels of manufactured, processed or mechanized inputs, where these strategies are implemented effectively, productivity is maintained, but it may be compromised in subsistence agriculture where nutrient additions are inadequate or absent. In industrial agriculture, on the other hand, additions of nutrients beyond that which can be used by the soil–plant system lead to their damaging leakage from the soil system into other environmental compartments via leaching and gaseous emissions. In this case the soil system is polluted and unhealthy.

4. ASSESSING SOIL HEALTH

(a) *Principles*

Assessment of soil health across agricultural systems, soil types and climatic zones presents major scientific and policy challenges. Given the multicomponent nature of soil systems, the breadth of goods, services and functions that they are called upon to provide, and their spatial variability, a complex debate is to be expected about appropriate methods for soil assessment. Clearly, no single indicator will encompass all aspects of soil health, nor would it be feasible (or necessary) to measure all possible indicators.

Emergent proposals for soil assessment are linked to the establishment of legal frameworks for the protection of soil at national (e.g. Defra 2004) and international levels. For example, the European Commission is implementing a Thematic Strategy for Soil Protection in Europe (European Commission 2002) which identifies erosion, declining organic matter, contamination, compaction, salinization, loss in biodiversity, soil sealing, landslides and flooding as the main threats to soil. In response, the ENVASSO project (European Commission 2007) has been established, which is an initiative to harmonize existing datasets to form a European-wide reference to assess current and future soil status. This and other initiatives can provide valuable information about the physical and chemical state of soil, but a rationale is needed to relate this state to soil health. The problem is that an integrative conceptual framework is lacking for identifying diagnostic indicators.

To reiterate, our definition of a healthy agricultural soil is essentially one that is capable of supporting both an adequate production of food and fibre and also the continued delivery of other essential ecosystem services. Using concepts taken from operations management theory (Slack 1997), a healthy soil could thus be described as one that presents a satisfactory system performance. A set of system performance curves can be imagined describing the relationship between potential rates of outputs from a soil system to rates of inputs (Kibblewhite 2005). Such curves define the current capacity of a soil to support required ecosystem services, including agricultural production and correspond to the familiar form of response curves (figure 4). The 'working range' of the soil system is that over which there is no degradation of system performance in terms of input-to-output conversion efficiency with increasing outputs. Above this range, performance deteriorates as indicated by falling efficiency, but as long as the level of inputs do not exceed the working range greatly, no permanent damage occurs. However, if the loading is excessive, outputs fall as the soil system becomes increasingly compromised and at some point permanent degradation results. The capacity of the soil system to deliver goods and services is defined by the extent of the working range and the input-to-output conversion ratio at the upper limit of the working range.

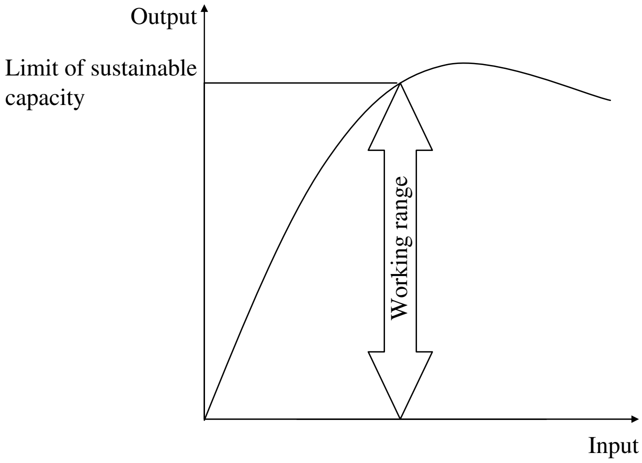

Figure 4. Performance curve showing an idealized relationship between input(s) to the soil system and delivery of an output of an ecosystem service. The capacity of the system is the output above which process performance deteriorates.

Thus, the 'healthiness' of a soil is indicated by its actual capacity relative to that within a population of soils. Healthy soils will have more extended working ranges and higher conversion ratios compared with less healthy ones.

The soil system is an open one and its health is affected by external environmental and anthropogenic pressures. The reaction of the soil system to these pressures can be described in terms of resistance and resilience (Griffiths *et al.* 2000; Orwin & Wardle 2004). Resistance is denoted by the magnitude of the change in state for a given level of perturbation. It further indicates a change in conversion ratio, for example a reduction in the respiration rate arising from compaction. Resilience describes the capacity of the system to return to its original state following perturbation and reflects the 'self-healing' capacity of the soil system, a concept that maps onto that of self-organization. Indeed, resilience may be a way of measuring the capacity for self-organization in soils. Some formally demonstrated examples of soil resilience are where the soil structure rejuvenates following compaction (Griffiths *et al.* 2005), microbial biomass reverts to antecedent concentrations following a drying cycle (Orwin & Wardle 2004) or decomposition potential is restored following a temperature perturbation (Griffiths *et al.* 2004). If the perturbation is within the capacity, the soil system can recover to its original condition, but if not, a permanent loss of soil health is expected. For example, in the latter study, while the grassland soil under study was resilient to a heat perturbation, this was not the case where the soils were subjected to copper (Griffiths *et al.* 2004).

(b) *Measurement*

There are, however, practical difficulties which make assessment of soil system health by measurement of performance curves and reactions to external pressures rather problematic. First, there is difficulty in rigorously defining at least some of the ecosystem services other than food-and-fibre production. Second, soil systems are multifunctional and able to deliver a variety of combinations and levels of services, so that full evaluation of the system performance would require very extensive testing. Third, soil systems are open systems and their performance is variable and interactive with environmental factors, such as air temperature and precipitation, which are not easily controlled. Fourth, soil system performance does not respond instantaneously to altered conditions, and its assessment has to be made over significant time periods. In truth, field assessment of whole soil system performance requires long-term, complex and detailed experimentation which can pragmatically only be conducted at a restricted number of sites. While an alternative within-laboratory assessment may provide useful information that helps to understand better how the system operates under well-controlled experimental conditions, it cannot provide an assessment which is indicative of whole system performance in the field.

The condition of complex systems is often assessed using diagnostic tests to support operational asset management because whole system performance assessment is either impossible or too costly. We propose that the same approach can be applied to soil systems to indicate their healthiness. Diagnostic tests offer a means to provide ongoing information about the condition, connection and configuration of those components or component processes that are critical to overall system performance.

We have identified critical processes in the soil system as transformations of carbon, cycling of nutrients, maintenance of the structure and fabric of the soil, and biological regulation of soil populations. There are existing techniques for assessing the performance of specific processes linked to these functions, such as respiration rates following organic matter addition, organic nitrogen mineralization rates during incubation, etc. While providing useful information about specific processes, these reflect the current activity within soil rather than any intrinsic capacity to support ecosystem services. An indication of the health of the soil system as a whole requires a more integrative approach. Individual processes are not related solely in a linear fashion, but within a network of interactions leading to a nonlinear system with associated feed-forward and feedback loops. Building on our conceptual model for the soil system as a living and reactive system whose performance is interactive with habitat condition, we propose that assessment of soil system health may be achieved using diagnostic tests, for example, abiotic ones that are indicative of the state of the habitat (i.e. physical and chemical conditions such as bulk density, aggregate stability, pH, cation exchange capacity, etc.), and the levels of key energy and nutrient reservoirs (e.g. ratios of organic matter fractions and nutrient balances); as well as biotic measures, such as those which are discussed below, which describe the community composition and populations of key functional groups of organisms (earthworms, N fixers, pest-control populations, etc.).

Although there are many similarities between all soil systems, differences in soil forming factors over space and time have led to distinct soil populations with characteristic properties. Any scientific assessment of soil health has to be made with due regard to these different populations. Soils that are intrinsically very fertile, for example because they are deep, well drained and have a favourable texture and background nutrient content, may be in good or bad health, and in the latter case may only be able to support delivery of ecosystem services at levels below that of a less fertile soil that is in excellent (healthy) condition. The question then arises 'What levels of such measures are indicative of soil health?' The 'agricultural equilibrium' that agroecosystems reach following conversion from natural vegetation could theoretically be used as a baseline for assessing the health of similar soils. It is difficult however, given its dynamic nature, to establish what that equilibrium should be for any given combination of soil type, climate and agroecosystem design. Thus, relative comparisons of soil health have to be made between modified soil systems (derived from the same type of soil) that are, or have been, subject to different agricultural systems and/or practices. It seems unlikely, and is scientifically naive, that exact thresholds for individual measures can be set objectively that define where the impact of substitutions or other management practices lowers efficiency below an ecologically or economically acceptable level. However, an approach would be to measure chosen indicators for soils representative of the whole population of a soil type and evaluate changes, trends and ranges within these populations. From this flows the conclusion that the concept of soil health and related diagnostic testing is most useful when applied to soil populations at the landscape scale, when the concern is about the general impact of natural pressures or agricultural practices on populations of similar soils at a landscape or regional scale. A more instrumental approach is needed at the individual field level to support operational decision making about nutrient additions, pesticide applications, cultivation timing, etc. Nonetheless, assessment of soil health by analysis of changes, trends and ranges using diagnostic tests for soil habitat condition and biological community structure offers a powerful means for evaluating the impacts of climate, land use change and altered agronomic practices on the valuable natural capital represented by soil.

(c) *Biotic indicators*

We have argued that soil health is fundamentally underwritten by the assemblages that carry out the various key processes. These assemblages are predominantly biological in origin, but actually involve a particular configuration of the biology, physics and chemistry of the soil constituents. This configuration is undoubtedly complex and unlikely to be directly measurable, so surrogates must be sought. The status of the soil biota may be a surrogate measure, notwithstanding there is a panoply of ways of measuring soil community attributes (Kirk *et al.* 2004; Leckie 2005). Community-level measurements can be classified as those based on genotype, phenotype and function (Ritz *et al.* 2004), which forms a logical series from the fundamental information required to create organisms and biological molecules, through its physical expression to its environmental manifestation. The genetic structure of the soil biomass, particularly at the prokaryotic level, is remarkably complex (Curtis & Stoan 2005). To date, there is little evidence that the soil genome relates particularly to the environmental circumstances from which it is derived, although the biogeography of soil microbial communities is not well researched. Community-level genetic profiles tend to show great variation both within and between soils under both similar and disparate ecosystems (e.g. Grayston *et al.* 2004; Fierer & Jackson 2006). At a whole-community microbial level, while everything might not be literally everywhere, extreme genetic complexity is apparently universal and genetic profiling has yet to be proven as a useful diagnostic of soil health. The much-vaunted belief that the gene is the appropriate level to measure the state of the system (e.g. reviews by Insam 2001; Torsvik & Ovreas 2002; Kirk *et al.* 2004; O'Donnell 2005) may be misguided, distorted by a strong contemporary focus on (predominantly organismal) genomics in biology. In particular, it may be an inappropriate paradigm for soil communities due to their extraordinary diversity. The potential for the phenotypic state of the soil community to be an effective indicator of biotic status is apparently greater. This makes ecological sense, since the phenotype, by definition, reflects the manner in which the environment interacts with the complex soil genome and impacts upon the expressed biota—an 'environmental sieve' through which the soil genome passes and is manifest as the attendant phenotype. There is increasing evidence that soil microbial phenotypic profiles, such as those based on membrane-lipid composition, are generally coherent and consistent, for example between different soil–land-use combinations, under different management practices or in soils exposed to pollutants (e.g. Kelly *et al.* 2003; Grayston *et al.* 2004; Abaye *et al.* 2005; Bossio *et al.* 2005). Functional profiling is a looser concept, not least because there are such a variety of functions that can be defined and measured. The so-called 'community-level physiological profiling' (CLPP) concept, which measures the ability of the soil community to metabolize a prescribed range of diverse carbon sources, shows potential, particularly when based upon the use of carbon substrates added directly to soil (Degens & Harris 1997; Campbell *et al.* 2003). This is because the suite of substrates employed can be prescribed according to a variety of ecologically pertinent factors such as synergy with C inputs that the system normally encounters (or not), energetic status, molecular complexity and so forth. Coherent studies applied at appropriate spatial and temporal scales to truly test these issues of genotype versus phenotype versus functional profiling as interpretable biotic indicators of soil health are lacking. What is quite clear is that any measure of soil health must be multivariate—single properties will not adequately encompass or integrate the features or issues that underwrite soil health.

5. IMPACTS OF AGRICULTURAL PRACTICE ON SOIL HEALTH

To maximize yields of food and fibre, a variety of agricultural management processes are imposed on the ecosystem, including artificial inputs such as chemicals and tillage. These practices and inputs supplement or even 'substitute' for biological functions that are seen as inadequate or inefficient for achieving required levels of production. This distorts the natural balance of the ecosystem and may compromise the output of other environmental services. The loss of non-productive services may affect farmers directly but often has effects which are distant in space and time. For example, nutrient leakage from the soil–plant system may lead to degradation of surface and ground waters and pollute drinking water supplies, while fine seed bed preparation on some land may increase the risk of soil erosion and sediment transfer to streams, or lead to surface capping, rapid surface water runoff and increased flood risk. In these and other cases, the costs of remediation or lost services are not borne by the farmer but elsewhere in the economy (Environment Agency 2002). Achievement of sustainable development requires that such externalities are contained, and new legal frameworks are being constructed that attach economic value to natural functions (i.e. by recognizing them as services), particularly those that relate to the water cycle, through legislation, incentives, trading mechanisms, etc. (e.g. European Commission 2005). Thus, one essential component of sustainable agriculture is (as embedded in our definition of soil health) to balance the ecosystem functions in such a way as to secure the target of agricultural production without compromising other ecosystem functions with respect to both present and future needs. In this section we examine the impacts of agricultural practice on the soil health system as described above, as a basis for deriving some principles for managing soil for sustainable soil health.

All agricultural soils have been altered from their natural state by human interventions which are aimed at maximizing production functions and which, to some degree, always result in a loss of other ecosystem functions. After clearing the natural vegetation to establish agricultural fields, all the major soil properties whereby we describe its health are changed, largely negatively. After a period of continuous cultivation they reach a new, dynamic, equilibrium. This has been most substantially documented in terms of the decline in soil organic matter content over the years immediately following clearing and the initiation of cultivation (e.g. papers in Leigh & Johnston (1994)). If there are no additions of nutrients to replace those lost by release during the transition to a new equilibrium and subsequently in crop offtake, the capacity of the soil ecosystem to deliver production and other services declines, and according to our proposed definition so does the health of the soil. Furthermore, the loss of ion exchange capacity which is concomitant with a decline in soil organic matter reduces the capacity of the soil system to retain nutrients that would otherwise be leached to groundwater.

The soil food web may also be substantially changed. In the Brazilian Amazon, large areas of forest have been converted to cattle pasture. Studies of change in the soil fauna showed that many of the main species of macrofauna present in the forest soils are not found in the pastures. In particular, the earthworm community changed from one commonly characterized by about six endemic species to one dominated by the opportunistic exotic species, *Pontoscolex corethrurus*. This is a species which, in contrast to many of the native worms, produces highly compact casts which have the effect of decreasing soil macroporosity, resulting in a surface layer which quickly becomes saturated and develops anaerobic conditions in the rainy season, which in turn stimulates methane emission and denitrification. Subsequent plant growth is also inhibited by the unfavourable soil physical conditions. Experiments showed that inoculation with a diverse group of the native soil macrofauna resulted in the 're-engineering' of this soil to reproduce a friable soil structure (Chauvel *et al.* 1999; Barros *et al.* 2004).

The agricultural management practised in the years immediately subsequent to clearing may serve to either exacerbate or ameliorate the processes of change put in place during the conversion phase. The intensity of agricultural intervention varies enormously across different farming systems, and may be expected to have both quantitatively and qualitatively different impacts on the soil health system. Different soils in different climatic and topographic situations may be more or less resilient to the introduction of agriculture. Flat alluvial soils in areas without extremes of climate are less likely to degrade quickly compared with shallow soils on steep slopes where rainfall may be intense.

The form and extent of substitution is a potential hazard to soil health, with the three most frequent practices being industrial pesticides (substituting for biological pest control), mechanical tillage (substituting for biological regulation of soil structure) and inorganic fertilizers (substituting for organically and biologically driven nutrient cycles). In view of the high degree of interconnectedness between functions described earlier, the use of energy and/or chemical products to replace, bypass or modify any particular biological function can be expected also to have significant consequences for other functions that have not been targeted.

The effects of intensive mechanical tillage on soil food webs provide a most instructive insight into the impacts of agricultural intervention on the integrated functioning and health of the soil system. The adoption first of animal-drawn and then of fossil fuel-driven tillage was one of the most significant steps in the history of agricultural intensification, enabling huge savings in human labour and increased efficiency, through improved timing in other agricultural operations, as well as the guarantee of a well-prepared seed bed. However, over the past two decades or so, there has been a substantial reversion to reduced tillage practices in many parts of the world, particularly in North and South America (Landers *et al.* 2001). The main perceived benefits driving the adoption of reduced or even zero tillage regimes were improved water and soil conservation, consequent on improved soil protection from the retained crop residues as well as reduced costs in terms of fuel (Van Doren & Allmaras 1978).

A number of major and long-term studies comparing soil food webs under intensive and reduced or zero tillage conditions have been made since the mid-1980s. Wardle (1995) reviewed and analysed more than 100 papers reporting on these studies and was able to derive a number of generalizations with respect to the impacts of tillage on the soil food web and a variety of processes mediated by the soil biota. The most obvious effect is a relationship between the size of the organism and the inhibitory effect of tillage (figure 5). This is indeed not unexpected, since mechanical tillage disrupts the spatial integrity of the soil fabric, particularly at meso- and macrofaunal scales. To some extent, tillage is intended to substitute for biological ploughing and it is well known that earthworms are killed during this process. In no-till, however, the enhanced activity of the macrofaunal engineers in soil structure modification 're-substitutes' for the withdrawal of intensive tillage. The origin of changes to the water regime under no-tillage, such as reduced runoff, increased infiltration and storage, are significantly physical in origin but the results of the food web studies show that enhanced activity of the macrofaunal ecosystem engineers also plays a

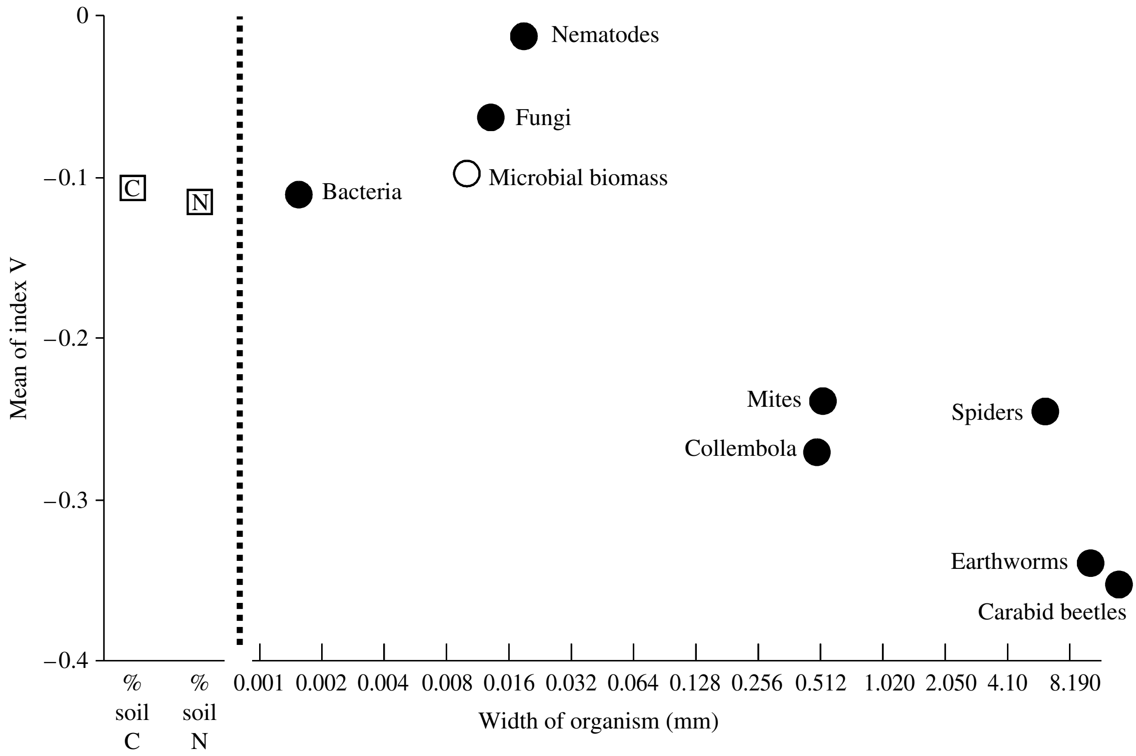

Figure 5. Differences in soil communities (shown in relation to their body size), carbon and nitrogen in mechanized versus no-tillage agriculture averaged for 106 studies across a variety of site conditions (adapted with permission from Wardle 1995). The index V is a measure of the abundance or mass of organisms under mechanized as compared with no-tillage; a value of zero implies equal abundance in each treatment, negative values indicate inhibition by mechanized tillage to a maximum of −1, i.e. the state in which the organisms only occur under no-till (see original paper for further details).

substantial part. The effects of the changes pictured in figure 5 are, however, by no means confined to soil structure modification.

In the food web of Hendrix *et al.* (1986) for a no-tillage system at Horseshoe Bend in Georgia, USA (figure 2), the dominant path of organic matter decomposition was that leading from the fungi to the earthworms, a pattern very similar to that typically found in forest ecosystems. In contrast, under conventional tillage bacteria assumed greater importance as primary decomposers, and in the higher trophic levels smaller fauna with shorter generation times, such as enchytraeid worms, largely replaced the earthworms and other macrofauna. The explanation given by Hendrix *et al.* (1986) for these effects is that during the process of ploughing such crop residues as are retained are both broken up and redistributed through the plough layer. This provides a greater surface area for microbial colonization as well as enhanced aeration and stimulates faster and more complete decomposition. This dispersed pattern of small ‘resource islands’ compared with the layering of residues at the surface in no-till systems also favours bacteria over fungi, which have a greater capacity to temporarily immobilize nitrogen. Mechanical tillage also results in a disruption of the spatial organization of the soil, rendering previously physically protected organic matter available to microbial decomposition, and previously inaccessible prey to predation, with a concomitant acceleration of nutrient cycling. As a consequence, increased N mineralization and higher rates of soil carbon loss are regularly found under intensive as compared with reduced or zero tillage. Fungi also tend to predominate in reduced tillage systems, since there is less frequent physical disruption of their mycelia. Fungi are often overlooked as ecosystem engineers but they contribute to structural genesis via many mechanisms (Ritz & Young 2004), and fungally mediated translocation of C and N between residue layers and soil horizons can be substantial (Frey *et al.* 2000, 2003). Long-term studies of soil organic matter dynamics have shown how progressive decline in soil carbon from the initiation of continuous intensive cropping in the mid-West of the United States over 60 years ago may be halted and even reversed by reduced tillage (Paustian *et al.* 2000). These effects have been shown to be not only due to greater retention of carbon but also due to improved aggregate formation (Six *et al.* 1999).

Thus, with respect to three of our four key environmental services, carbon transformation, nutrient cycling and soil structure modification, intensive mechanical tillage shows a significant negative effect, which can, however, be reversed under many circumstances without loss, and in many cases with gains, of crop production. Reduced tillage is usually associated with increased weed development and the retention of residues can also stimulate diseases that are retained in them (e.g. the root-rot and stem-infecting fungus *Rhizoctonia*) or pests and diseases that respond to the increased moisture (such as slugs, and fungal rots such as *Pythium* species). This has lead to increased use of herbicides in many such systems, sometimes also associated with other pesticide. Wardle’s (1995) review revealed two interesting features in this respect. First, that the impacts of herbicides on the soil food web, and thence on their functions, was usually very slight; and second, that the presence of weeds frequently enhanced both carbon and nutrient dynamics by providing additional organic inputs.

The most significant lesson that the widespread and detailed studies of the effects of tillage contribute to our understanding of soil health is that the single practice of ploughing has multiple, and largely negative, impacts on the soil biota, the processes they mediate and the environmental functions that they contribute. Furthermore, these impacts are highly interactive between the functions, supporting the concept proposed in figure 3. The positive side of this story is that the response of the soil biota to a removal of this disturbance is one of integrated reconfiguration, revealing the resilience of the system, and that the capacity for self-organization has not been lost. These generalizations must clearly be qualified by the differences in detail that occur under the varying circumstance of soil type, climate and management intensity.

The same broad lesson emerges from consideration of the impact of pesticides and fertilizers, so these will be described only briefly. While there have been many studies of the ecological impacts of pesticides on above-ground ecology and soil macro- and mesofauna, rather less information is available on their impacts, or indeed those of other practices, on soil microbial ecology. All pesticides, whether applied directly or targeted at the above-ground parts of the plant or the pests themselves, are liable to end up in the soil and in contact with soil organisms. The impacts of a wide range of pesticides on specific groups of soil organisms, soil food webs and, to a more limited extent, biological processes in soil have been extensively documented (Edwards 1993). Predictably, the effects are highly variable, dependent on the type and amount of pesticide, soil environment and biotic groups studied. Generally speaking, however, the impacts are similar to those of tillage in that the impacts with the most far-reaching effects are often those at the higher trophic levels. Thus, the impact is not restricted to the target but has disruptive effects on the biological regulatory capacity of the soil community with damaging consequences for all soil functions (Edwards & Bohlen 1995; Edwards 2002). The same concerns thus exist with respect to the impact of pesticides on soil health as exist for their use generally, i.e. not only that indiscriminate use can have dangerous consequences for human health and impacts on environmental functions, but also that the whole basis of pesticide use can be economically inefficient if the non-target impacts are weighed against the targeted success.

The impacts of industrially produced fertilizer on the soil health system and ecosystem functions relate firstly to their effect on primary productivity. The effects of excessive quantities are on process rates rather than any direct toxic effects. A very important indirect impact is the fact that high fertilizer input use is commonly associated with reduction in the quantity of organic matter input.

The presence of high concentrations of ammonium inhibits nitrogen fixation and stimulates nitrification. High levels of some nitrogenous fertilizers can lead to acidification in some soils and consequent effects on the soil biota. Excess nitrate may leach from the soil and contaminate sources of drinking water and/or change the nutrient balance in aquatic ecosystems. These excesses also fuel denitrification and the production of nitrous oxide. The combination of these effects has been documented extensively and has led to the conclusion that the global nitrogen cycle is significantly out of balance, and that agriculture is one of the main contributors (Vitousek *et al.* 1997; Wood *et al.* 2000). In terms of soil health at a more local level, the effect is a substrate-driven loss of internal controls and the opening up of cycle function.

These direct effects of inorganic fertilizers on the nutrient cycling function are exacerbated by the reduction in organic matter inputs which often accompanies high rates of fertilizer use. Although fertilizers are highly effective in increasing crop production, integrative practices of combining them with organic inputs are commonly abandoned in the interests of efficiency, and above-ground residues are often removed or burned. Inorganic fertilizers have been shown to increase the rate of decomposition of 'low-quality' organic inputs and soil organic matter (Vanlauwe *et al.* 1994, 2001; Recous *et al.* 1995). This effect is usually attributed to the enhancement of microbial decomposer activity previously limited by low nutrient concentrations in the organic resources. It should be noted, however, that the results of experiments on this effect are equivocal: although a majority of results indicate the above effect, in a significant minority added inorganic nitrogen has either a neutral or even an inhibitory effect on the decomposition of low-N plant materials (Hobbie 2005). This is probably indicative of the interaction with secondary rate-limiting factors, but makes the point that the addition of a single 'simple' source of nitrogen can have complex interactive effects on carbon transformations in the soil. The commonly observed overall effect of continuous inorganic fertilization with diminished input of carbon and energy is continuing decline in soil organic matter content.

Finally, it should be noted that although each of the three substitutive practices have been considered separately, they are commonly used in concert. Comparative studies of soil food webs and functions in such multiple substitution systems and low substitution integrated agriculture confirm the improved soil health in the latter (Brussaard 1994).

6. TOWARDS AGRICULTURAL MANAGEMENT FOR SUSTAINABLE SOIL HEALTH

The conclusions arising from this paper are derived from the premise that soil is the site of a vital range of ecosystem functions which provide humans with a range of essential services. In natural ecosystems, these functions and services are driven by the energy generated by carbon transformations carried out by the soil biological community acting in a highly interactive and integrated fashion.

Conventionally, the practice of agriculture may be seen as providing only a single service, namely arable or livestock food production. Primary and secondary production depends on soil-based ecosystem functions such as nutrient cycling, maintenance of soil structure and biotic population regulation. Society may also require that other services, such as the supply of good quality water, protection of human health and reduction of greenhouse gas emissions, be maintained at acceptable levels. This demand is already being strongly voiced in many developed economies. A major target of sustainable agriculture must be to ensure that the full range of ecosystem services is conserved for future generations: agricultural soils must thus retain a multifunctional capacity. We use soil health as a term to describe the capacity of soil to deliver a range of different ecosystem functions and services.

Agricultural interventions, such as the use of pesticides, powered tillage and the use of inorganic sources of nutrients, impact upon the biological communities of soil, damage their habitats and disrupt their functions to varying extents. The link between disturbance, targeted biota and effect on function is far from linear owing to the high level of interaction between organisms and functions. The main integrating feature in the soil community is energy flow. The majority of the soil organisms depend directly or indirectly via one or more trophic levels on the processes of organic matter decomposition for their source of energy and carbon. Any disruption of this energy generating system may thus result in changes in the flow of energy and carbon to the different functions. Assessment of the relative energy allocation to different functions remains to be computed but may prove difficult owing to the second integrating feature of the soil health system, that of the probability of participation in more than one function by the same organisms. Although distinct ‘functional assemblages’ of organisms responsible for the different functions have been recognized (figure 1), a significant proportion of the soil biota may contribute to more than one function. For example, a substantial proportion of the organisms participating in functions such as nutrient cycling and soil structure maintenance are also primary or secondary agents of decomposition; while earthworms and termites can clearly be identified as major ‘ecosystem engineers’ with respect to their role in soil structure maintenance, they also contribute significantly to nutrient cycling. At the trophic levels of microbivores and predators, the crossover in function is even more evident as is apparent from food web diagrams (figures 2 and 3). A third major integrative feature is that of the relationship between organism and habitat. The activities of soil organisms are influenced by the condition of their habitat in the soil, but at the same time continuously modify it. Any shift in one function is thus likely to influence others by habitat change.

These generalizations hide a huge amount of the variation possible in the functioning of the soil community and the lack of clear evidence for the patterns of energy flow within the communities and the participation of particular species or taxonomic groups in different functions. Elucidation of these issues remains a major research challenge; nonetheless the overwhelming conclusion from this analysis is that soil health in its functional sense should be seen, and managed, not as a set of individual soil characteristics but as an integral property of the ecosystem.

An integrative approach is also essential for assessment of soil health. It is not feasible to assess soil health directly on the basis of its delivery of different ecosystem services. Furthermore, soil health is related to functional capacity rather than actual service outputs. As argued above, an effective approach appears to be using a set of diagnostic tests for soil system performance, chosen to be indicative of habitat condition, i.e. physical (e.g. bulk density) and chemical (e.g. pH, salinity), of energetic reservoirs (e.g. soil organic matter content) and key organisms and community structure (e.g. earthworms and phenotypic profiling). Nevertheless, we consider that this essentially reductionist method for diagnosing soil health falls short of that required to properly assess the condition of the integrated and complex soil system. While it offers the only current means to attempt diagnosis, the development of more integrative biological methods is a research priority. It is necessary to assess soil health by comparison of diagnostic test data for relevant populations of soils at the landscape scale, covering combinations of soil type and land management classes (e.g. arable and grassland). At present, there are no agreed distinct thresholds above or below which the soil can be said to be healthy or not in a definitive sense.

The integrated nature and high diversity of the soil health system may contribute a significant degree of resilience under conditions of disturbance, particularly at lower (largely microbial) trophic levels. Nonetheless, the conversion of natural vegetation to agricultural land results in major changes in both physical organization and community structure in the soil, including species loss and changes in dominance among the surviving biota. This then becomes the resource with which agriculture must work and any targets should realistically be set in relation to the potential equilibria in agricultural systems rather than the natural systems from which they are derived, as has sometimes been advocated. More importantly, it is clear that subsequent agricultural practices may also impair soil health through significant impacts on the composition and structure of the soil biological community and consequently on soil-based ecosystem functions and services. Damage to ecosystem functions can arise both owing to an inadequate supply of resources (carbon, energy, nutrients or water) and through the impact of intensive substitutive practices such as continuous mechanical tillage, the use of pesticides and excessive amounts of fertilizers. These interventions may also impact on soil functions by destroying or changing the habitat of the soil organisms and their capacity to repair it.

Sustainable management of soil health requires the setting of criteria for acceptable levels of soil-based ecosystem functions and in particular the balance between the food production functions and others supporting soil conservation, water flow and quality, crop, livestock and human health control, and greenhouse gas emissions. The established principles for establishing and maintaining soil fertility are familiar.

These include inputs of organic matter to meet demand for carbon and energy supply to the soil biota, balanced with the nutrient demand of the crops and the development of integrated (i.e. organic plus inorganic) nutrient management systems where inorganic fertilizers are used in precise dosage in combination with equally carefully designed practices of organic matter management that conserve nutrients and levels of soil organic matter. These principles are consistent with those needed to support soil health, and so capacity to deliver a range of ecosystem services, in the context of the integrated description of the soil system that we have proposed. In addition, however, the maintenance of continuous vegetative cover and in particular rooting systems as advocated in some integrated farming practices (Tilman *et al.* 2002; Sanchez *et al.* 2004) will also promote a healthy soil, via an associated continuous feed of carbon substrate below ground. In general, intensive mechanical tillage and pesticide use should be kept to a minimum.

Sustainable management of soil will nonetheless always be related to particular circumstances. The priorities in industrial agriculture, to reduce and refine the input management system, are clearly different from those of subsistence farmers where the key to sustainable management is to increase inputs. In Africa generally, and more sporadically through much of the tropical regions, production is inadequate, resources limited and food sufficiency and agricultural profitability are lacking. The key issue here is to find management practices that will 'lift' the systems and can be implemented within the limited resource base (including cash) that is available, and are also sustainable in the long term. In sharp contrast is the case of industrial agriculture where productivity and returns are high (although the latter is often distorted by subsidies) and where the realization of unacceptable impacts on the environment and human health has led to the search for more sustainable practices that nonetheless do not compromise productivity or profitability. In both cases, a healthy soil is central to the sustainable solution.

Sustainable solutions with regard to soil health will depend on the willingness of society to pay for its maintenance, which in its turn depends on the value accorded to the various functions and services it supports. To date, it appears that both the measurement and economic evaluation of soil-based ecosystem functions have not been made. Industrial societies, through the agency of governmental policies, have increasingly shown themselves willing to pay the costs for establishing limits to polluting effects (e.g. on nitrate levels in groundwater) or in encouraging actions to enhance ecosystem functions (e.g. for carbon sequestration). Few would now disagree with the assertion that a practice which results in substantial accumulations of heavy metals, pesticides or nitrates is undesirable, and be prepared to pay for it to be avoided or alleviated, even when the effects of these accumulations on human health or agricultural production are unclear. Legislation has placed limits on such effects in many countries. The same widespread consensus in relation to soil degradation by erosion, organic matter loss and physical damage is emerging only now. The effect is to alter the mix and levels of ecosystem services required from soil and the definition of 'healthy soil'. An example is temperate arable soil systems. Where these have been managed with intensive substitution over extended periods, their organic carbon levels have declined. High levels of agricultural productivity can, however, be maintained in most soil types through appropriate substitutive practices. On the other hand, this loss of organic carbon has reduced their capacity to absorb and retain pollutants, representing a loss of ecosystem service capacity. Thus, soil that has been assumed to remain healthy from a crop production perspective is increasingly recognized to be unhealthy in the context of greater valuation of environmental service provision.

For farmers in the developing regions of the world who are starting from a very low resource base, access to inorganic fertilizers is essential to 'kick-starting' their degraded systems, as are pesticides under frequent conditions of acute pest or disease problems. The economic circumstances of these farmers, however, render such inputs unobtainable and a wide variety of alternative practices have been developed using variations in cropping and farming system design as an alternative to industrially produced inputs. These 'organic' practices, enforced by necessity, may contribute to improved soil health and sustainable practice but are generally insufficient in terms of production. Wherever inputs are affordable, they must clearly be used to enhance production, but the risk to other ecosystem services and soil health can be minimized by maintaining the integrated nature of their farming systems.

Despite the great variety of biophysical and socio-economic circumstances that need to be accommodated, a working hypothesis for sustainable agriculture may be advanced that 'agriculture can be productively and profitably practised without impairment of soil health'. A more cautious assertion that recognizes the reality behind such a target is that 'some degree of trade-off between the optimization of one ecosystem function (in this case food or fibre production) and others (e.g. water quality, carbon sequestration) is acceptable and indeed inevitable in any managed landscape'. Irrespective of which of these approaches becomes dominant, the emergence of a globally acceptable concept of sustainable agriculture will require the convergence of the excess-resource and inadequate-resource trajectories of change on a diversity of practices rather than any single homogenized approach such as has characterized agricultural development over the past 50 years.

As in any broad review, our concepts and ideas are those formulated over a long time and in dialogue with many valued colleagues and peers. We especially acknowledge John Crawford, Jim Harris, Iain Young and numerous colleagues in TSBF and NSRI (*sensu lato* in both cases) for collaboration and stimulating discussions.

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