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*Published in:*  
Food and Energy Security

*DOI:*  
[10.1002/fes3.158](https://doi.org/10.1002/fes3.158)

Print publication: 01/05/2019

*Document Version*  
Publisher's PDF, also known as Version of record

[Link to publication](#)

### *Citation for published version (APA):*

Stockdale, EA., Griffiths, BS., Hargreaves, PR., Bhogal, A., Crotty, FV., & Watson, CA. (2019). Conceptual framework underpinning management of soil health - supporting site-specific delivery of sustainable agro-ecosystems. *Food and Energy Security*, 8(2), Article e00158. <https://doi.org/10.1002/fes3.158>

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# Conceptual framework underpinning management of soil health—supporting site-specific delivery of sustainable agro-ecosystems

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## Funding information

AHDB- and BBRO-funded Soil Biology and Soil Health Research and Knowledge Exchange Partnership; Natural England

## Abstract

The need for sustainable intensification of agricultural production has ushered in a growing awareness of soil health and a requirement to identify with some certainty how changes to land management will affect soil. From an agricultural perspective, the active management of soil health needs to balance the production of a healthy and profitable crop with environmental protection and improvement. However, the extreme spatial and temporal heterogeneity of soils, and the complexity of biological, physical and chemical interactions therein, makes predicting management effects on soil health challenging. Although the general principles underlying effects on soil health are well understood, they still need interpretation in a local context and the inclusion of site-specific details. Approaches from landscape ecology provide a potential framework to integrate consideration of the structural (pools, patterns), dynamic and functional (processes, flows) aspects of the soil system. These approaches allow the crucial transition from a “descriptive and general” understanding toward a “detailed and site-specific” prediction to be made. Using this conceptual framework, we have taken knowledge of the effects of fixed site factors (soil type and climatic zone), cropping systems and farm management practices on a range of soil physical, chemical and biological parameters for UK lowland agricultural systems, and have developed a predictive framework that shows semi-quantitatively the effects of typical management choices on soil health and crop yield.

## KEYWORDS

farming systems, land management, soil ecology, soil quality

## 1 | INTRODUCTION

There is an increased focus on the impacts (both positive and negative) of farm management practices on soil chemical, physical and biological properties and processes within agricultural systems. This responds to the need to maintain (and

ideally enhance) soil function to support sustainable intensification of farming systems to ensure food security, whilst at the same time maintaining or improving overall ecosystem function (greenhouse gas regulation, flood prevention, conservation of biodiversity). In the UK, concerns about the impact of landowners' management practices on soil function

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and their impacts beyond the farm boundary are rising, and hence actions targeted at changing approaches to soil management are becoming a policy priority (Environmental Audit Committee, 2016). In many countries, spending on soil conservation measures makes up a substantial share of total agri-environmental expenditure with a range of approaches used, including investment and loans, to promote adoption of beneficial practices and advice at farm/catchment level (OECD, 2015). In this context, the Food and Agriculture Organisation has defined soil health in relation to key soil functions as: “the capacity of soil to function as a living system, within ecosystem and land use boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and promote plant and animal health. Healthy soils maintain a diverse community of soil organisms that help to control plant disease, insect and weed pests, form beneficial symbiotic associations with plant roots; recycle essential plant nutrients; improve soil structure with positive repercussions for soil water and nutrient holding capacity, and ultimately improve crop production” (FAO, 2008). The term soil health has clear conceptual appeal, but it remains difficult to interpret operationally.

Soil is an opaque medium with a complex physical structure, spatially diverse and with a temporally dynamic chemistry that is home to a wide range of biological taxa. The extreme spatial (vertical and horizontal) and temporal heterogeneity in soil gives rise to very different surface types, pore sizes and microclimates, and a range of resources together with resource partitioning in space and time. Predicting and modeling soil processes and functions is often caught by the “middle number” conundrum, that is, there are too many individual components with too many complex interactions to deal explicitly with the individual; yet the individual details affect the dynamics of the system as a whole, so general statistical properties yield an incomplete picture (Wu & David, 2002). This problem is amplified by spatial and temporal variations and interdependencies, scale dependencies, and thresholds. Govaerts et al. (2009) carried out a meta-analysis of changes in soil carbon stocks under conservation agriculture and showed that, although key driving factors could be identified at a range of scales (including rooting depth, rhizodeposition, soil bulk density, landscape position, climate), there were complex interactions between these factors. Consequently, the outcomes of the same management change for different sites could be positive, negative, or neutral depending on context, and consequently, simple prediction was not possible (Govaerts et al., 2009). Griffiths et al. (2015) designed an elegant study to detect any underlying associations between soil physical or biological stability (i.e., resistance and resilience) and management factors within a geographically restricted set of soils under similar land use. However, they also found that regional patterns driven by soil type or management systems outcomes were masked by

site-specific soil/management factors so that individual farms remained an important grouping factor even in regional-scale analysis. Therefore, whilst farmers seek guidance and monitoring that can be used to drive selection of locally-adapted, site-specific crop/soil management practices (Ingram, 2008), even the most rigorous scientific reviews can only indicate how farmers might optimize soil biological function and soil health at a farming system level in the most general way (e.g., Beauchamp & Hume, 1997; Clapperton, Chan, & Larney, 2003; Doran & Smith, 1987). For example, Clapperton et al. (2003) in a review of the role of soil microbial biomass in controlling nutrient release and plant uptake conclude: “Ideally agroecosystems should be managed to maintain the structural integrity of the [soil] habitat, increase soil organic matter (OM) and optimize the C:N ratios in soil OM using cover crops and/or crop sequence.” Such advice can barely be distinguished from the more poetic injunctions common a century ago, for example, “You must keep the soil free from stagnant water; keep it sweet ...; keep it open and mellow and fine; keep it free and attractive to air and like wholesome influences” (Burkett, 1917, p. 143). This seems to leave farmers without answers to a range of pertinent and practically important questions such as “how many cover crops and which ones, where is the right balance (economic as well as ecological) between minimizing tillage and optimizing weed control ...” (Stockdale, Watson, Black, & Philipps, 2006).

Management decisions that improve soil quality/health need to be taken at the individual field scale (Griffiths et al., 2015) but nonetheless need to be underpinned by robust understanding of the interactions driving soil properties, processes and their interaction with management within this site-specific context. Therefore, by considering approaches used within the discipline of landscape ecology, we have developed an integrating conceptual framework, which explicitly considers structural (pools, patterns), dynamic and functional (processes, flows) aspects within the soil system where site-specific interactions then determine soil processes and overall function. This provides both (a) a clear structure to underpin approaches to the assessment of the impacts of management practices on soil health and (b) support advice to farmers by recognizing the need for site-specific decision-making in management of soil health.

## 1.1 | Considering concepts from landscape ecology

The development of landscape ecology as a discipline at the end of the 20<sup>th</sup> century initially focused on the impacts of spatial factors, that is, “the causes and consequences of the spatial composition and configuration of landscape mosaics” (Wiens, 1992). For example, animal populations in landscapes are rarely constant in either time or space and, hence, where the spatio-temporal interactions affecting population

dynamics are not taken into account, errors in understanding and in management can result (Conn et al., 2015). Pattern prediction is complex and multifactorial resulting from interactions between access to resources and refuge from predators (Brown, Mehlman, & Stevens, 1995). Reductionist approaches to study landscape patterns and processes were not well suited for application where the array of possible spatial configuration was great, the range of relevant scales broad and the diversity of responses to these landscape patterns and processes large. The interactions of spatial and temporal variance and the emergence of scaling phenomena are also difficult to handle with the mathematical approaches traditionally used in ecology. Hence, the underpinning theories and models used are often verbal and qualitative rather than quantitatively-based. Similarly, in soils, the complex interactions in the soil food web are confounded by effects of the physico-chemical environment and land management. Thus, below-ground it has also been recognized that consideration of inter-organism interactions and their relation to function (Wardle, 2002; Wardle & Giller, 1996) need to be integrated with a description of spatial habitat factors (Young & Ritz, 1998). This is required to provide a coherent framework linking population dynamics of soil organisms to biodiversity and function in terms of the soil microenvironment (Young & Crawford, 2004). Lindenmayer et al. (2008) provided a synthesis of the concepts and approaches used within the discipline of landscape ecology and its application to conservation practice. They concluded that whilst there could be a common underpinning theoretical framework, each specific landscape problem would require its own specific analysis resulting in a range of possible outcomes and solutions, rather than a single recommendation (Lindenmayer et al., 2008); this is equally applicable to site-specific soil health solutions.

In landscape ecology, the landscape must be identified and described in a meaningful, measurable, systematic and, ideally, universal way. Landscapes are composed of multiple elements (e.g., the patch-corridor-matrix model; Forman, 1995), which characterize the heterogeneity within an area. Traditionally, a landscape is defined by the characteristic ecosystems that constitute it; landscapes then extend laterally until the recurring cluster of ecosystems or site types change significantly (Wilson et al., 2002). Often at national/regional level, major changes in underlying geology or geomorphology (e.g., rolling chalk downland, depositional basins) provide the initial broad segmentation, as these give clear physical boundaries rather than the more diffuse boundaries resulting from climate variation. These structural aspects of the landscape can be captured by maps. However, the dynamic and functional aspects of the landscape (processes, flows) are no less important, but are often less well identified and described (Lindenmayer et al., 2008). Work defining habitats within landscape ecology models above-ground shows the need to use variables that describe characteristics from across a range of scales (landscape context, landscape mosaic, microhabitat, food/refuge; Fernandez, 2005). However, it is the spatial relationships of these elements, as much as their diversity, that are key to affecting the interactions within the mosaic (Table 1).

Landscape ecology also explicitly recognizes the sensitivity of ecological patterns and processes to scale. This is often due to the different temporal (e.g., geological or diurnal) and spatial (e.g., km or mm) responses of organisms and habitats due to differences in size, mobility, and physiology. Scale issues therefore influence all the underlying decisions about identification and description described above (Lindenmayer et al., 2008). Ludwig, Wiens, and Tongway (2000) describe

**TABLE 1** Range of potential measures of landscape structure (adapted from Wiens, 1992 and Lindenmayer et al., 2008)

Patch-scale measures	Landscape-scale measures
Size	Number of patches
Shape	Patch size frequency distribution
Orientation	Patch diversity (richness, evenness, dominance, similarity)
Perimeter length	% of landscape in any patch type
Perimeter: area ratio	Patch dispersion (contagion)
Context (adjacency, contrast)	Edge density
Condition (habitat quality measures)	Fractal dimension (edge, area)
Distance (nearest neighbor, proximity)	Heterogeneity
Corridor characteristics (length, shape, linkage e.g., stream order)	Gaps (lacunarity)
	Spatial correlation (semi-variance, distance decay, anisotropy)
	Connectivity (network, lattice properties)
Local disturbance/disruption	Landscape-scale disturbance/disruption
Resilience	
Local →Landscape	

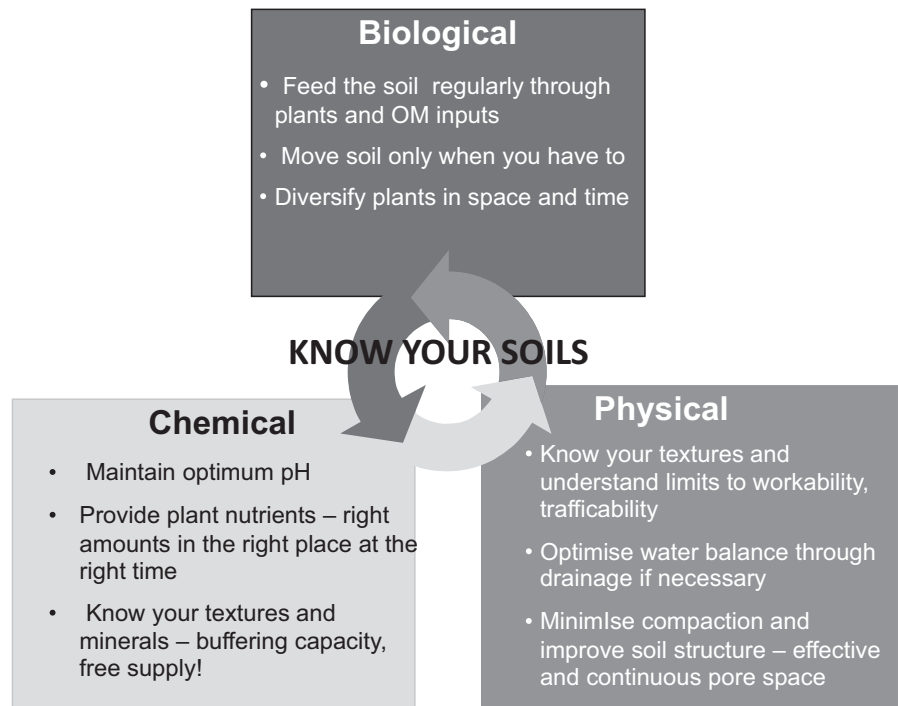
**TABLE 2** Lindenmayer et al. (2008) identified key issues for landscape ecologists to consider when seeking to translate broad principles into specific conservation outcomes at landscape scale. Here, these are considered from the perspective of soil scientists seeking to support the management of good soil health within farming systems

Landscape ecology framework—issues to be considered (Lindenmayer et al., 2008)	Supporting site-specific delivery of soil health—examples of how these issues might apply below-ground
Develop long-term, shared vision(s), through processes of clear problem definition, priority-setting, and quantification of objectives.	Widely recognized as critical for sustainable land management at landscape-scale (e.g., Schwilch et al., 2012) and for catchment management (Mutekanga, Kessler, Leber, & Visser, 2013); less clearly applied when considering farm/field scale
<b>Spatial issues</b>	
Manage the whole mosaic do not just consider the separate pieces	The spatial location of microbes, the configuration of pores and their interactions is now studied as a driver of soil processes (e.g., Jones, Clode, Kilburn, Stockdale, & Murphy, 2013; Young & Ritz, 2000)
Consider both the amount and configuration of habitat and land cover types	
Identify disproportionately important species, processes, and flows	For example, nitrification is a key process in determining losses of nitrogen from agricultural systems (Subbarao et al., 2007); arbuscular mycorrhizal fungi (AMF) are a keystone species when studying nutrient uptake by crops (e.g., Smith & Smith, 2012)
Integrate aquatic and terrestrial environments	For example, at farm/landscape scale, Haygarth, Condron, Heathwaite, Turner, and Harris (2005) highlight the importance for models of phosphorus cycling to integrate understanding of soil and hydrological processes
Use landscape classification and conceptual models appropriate to the objectives	For example, Lavelle et al. (1993) developed an underpinning general model for decomposition in forest systems and show how this needed to be adapted with respect to underlying soil and climate conditions
<b>Temporal issues</b>	
Maintain the capability of landscapes to recover from disturbance	The impact of disturbance on soil organisms and processes together with the capacity of soil to maintain ecosystem function following disturbance has been a recent focus for research (Griffiths & Philippot, 2013). It is recognized that change may be gradual or sudden and natural (e.g., climate, fire) and/or human-induced (land-use change, organic manure addition)
Manage for change	
Time lags between events and consequences are inevitable	For example, Powlson, Brookes, and Christensen (1987) highlighted the different timescales of measurable response in soil microbial biomass and total soil organic matter content following straw incorporation
<b>Management approaches</b>	
Manage in an experimental framework	Many of the actions put into practice on-farm as a result of the application of the principles (Figure 1) are not answers but investigations; a soil adaptive management cycle can be implemented by farmers and supported through policy/research (Birgé et al., 2016)
Manage both species and ecosystems	For example, success of inoculation of N fixing bacteria/ AMF to soil may depend on the dynamics of “native” soil microbial communities. This also depends on the host crop, crop rotation effect, site/soil variation, and abiotic stresses, for example, weather (Trabelsi & Mhamdi, 2013). Therefore, practical guidance on the use of inoculants highlights the importance of both ensuring the viability of the inoculant at application and management of the soil to ensure appropriate conditions for proliferation
Manage at multiple scales	For example, Groffman, Tiedje, Robertson, and Christensen (1988) highlighted the way in which denitrification was resulted at a range of scales and identified possible research and management options; Turner, Griffis, Mulla, Baker, and Venterea (2016) showed how investigation at a range of scales could lead to targeted management of N <sub>2</sub> O emissions
Recognize that solutions are often landscape-contingent	Soil type (texture) is often identified as a key factor in determining the impact of management practices (Griffiths et al., 2015)

very elegantly the need to consider how the scale at which the patterns and process of landscape co-occur with the scale of likely response so that the most appropriate scale(s) for

observation and management can be identified. Hierarchical concepts have been widely applied in landscape ecology (Allen & Starr, 1982) so that the mechanism(s) underlying an





**FIGURE 1** Summary of key principles for maintaining/improving soil health drawn together to underpin advice to farmers in the UK

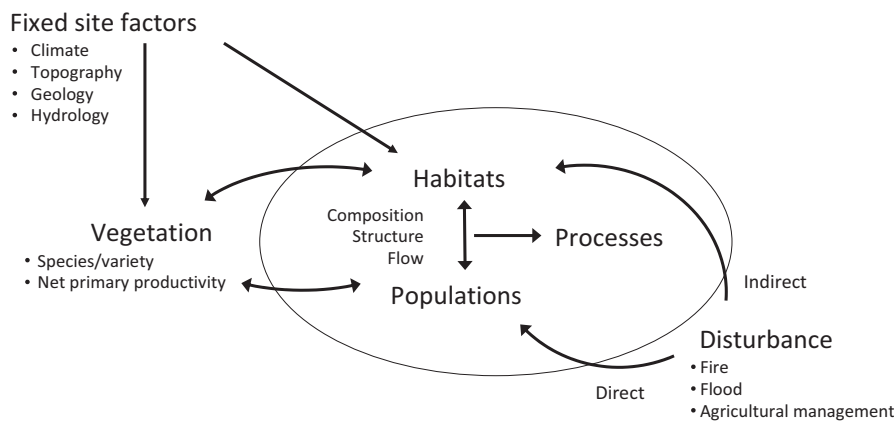
ecological process is best sought at the next lower level in the hierarchy; however, this is defined. Tschardt et al. (2012) also highlight the way in which mechanisms affecting biodiversity are driven by patterns/processes from the level above and hence the importance of considering the whole landscape context. Spatial and temporal hierarchies have similarly been widely used as descriptive tools to identify domains within which process-functions are reduced to graspable proportions (Klijn, 1994). Care needs to be taken in extending models and their conclusions beyond the scale at which they have been developed, as the pattern-process linkages very rarely change linearly with scale changes (Ludwig et al., 2000). In adapting these concepts to developing site-specific soil health solutions, the key step is moving from a general and descriptive understanding of soil processes to an approach that allows consideration of a detailed and specific knowledge of the biological interactions at that site.

Opdam, Foppen, and Vos (2002) noted that in the early phase of landscape ecology, there was a proliferation of empirical studies taking place at a range of scales, studying different organisms and processes enhanced by some modeling studies to extrapolate these studies across space and/or time. General principles can be drawn out of such data through meta-analysis; however, Rossetti, Tschardt, Aguilar, and Batáry (2017) show the importance of taking a structured (hierarchical) approach to data integration so that the nested structure of data can be considered fully during analysis. Opdam et al. (2002) also noted a lack of a structured approach to data integration in landscape ecology, and consequently, few applications to inform spatial planning

in practice. Where the aim of description and modeling of landscape processes is the identification of steps that are practical for local implementation and likely to yield positive (conservation) benefit, Lindenmayer et al. (2008) identified a checklist of issues that can facilitate the process of translation of broad considerations into useful landscape-specific practical management actions (Table 2). The development of robust indices that allow the gap between theory and practice to be bridged requires a good understanding of the underlying mechanisms, as well as the needs of the decision-making processes. For example, Opdam, Verboom, and Pouwels (2003) show how a structured approach to data consideration and the development of indices can result in effective tools to support planning of habitat networks. Studies in landscape ecology are also known to be strongly dependent on knowledge transfer to and from practice because controlled experiments at landscape scales by the research community alone are largely infeasible (Silbernagel, Chen, Normets, & Song, 2006). Thus, translation of broad general conclusions to practical tools will require discussion with and iterative development with the user community; for soil health, this will mean active engagement with farmers themselves.

## 1.2 | Using landscape concepts to describe soil ecology and processes

Soil conditions are often much more temporally variable than landscapes above-ground; ecosystem engineers (such as earthworms in temperate climates and termites in the tropics)



**FIGURE 2** General model showing the interaction of fixed site and disturbance factors in their effect on vegetation, soil biota and soil processes which are considered to result from the interactions between soil habitats and populations of soil organisms (adapted from the schema presented in Stockdale et al., 2006). The composition and structure of habitat factors describe the soil profile, structure, chemical pools etc and the composition and structure of the soil populations describe the soil biological community

and plant roots are constantly establishing and modifying connectivity and fragmentation in below ground systems. Rapid changes of short- or long-term duration in soil conditions often follow changes in environmental factors (e.g., rainfall events) or management (e.g., topsoil pH change following liming). The array of possible spatial configurations at a range of scales and the diversity of responses to the patterns in space and time often prevent easy description or robust modeling. The soil biological community is not homogeneous in space or time and there are often spatially separated populations of the same species which interact under certain soil conditions (meta-populations), for example, when soil is saturated or after tillage. Consequently, Fitter (2005) suggested from an ecological perspective that “the heterogeneity of soil means that meta-population ideas are necessary or possibly even meta-community or meta-ecosystem approaches.” The meta-population paradigm has been effectively linked with the definitions and approaches of landscape ecology (Opdam, Apeldoorn, Schotman, & Kalkhoven, 1993). The scale at which mechanisms controlling population size and activity are expressed within landscapes, and hence, the scale of the landscape differs for different organisms (Wiens & Milne, 1989); this will certainly be true below-ground. Simply describing the parts of the below-ground ecosystem is not sufficient; information about the connectivity between the components and their temporal and spatial context is also needed (Raes & Bork, 2008). Ettema and Wardle (2002) recognized that whilst spatial variability has been treated often as distracting “noise” which obscures the key relationships between structure and function of below-ground biodiversity, understanding the control over ecological systems imposed by spatial variability may be key to improving our ability to manage below-ground ecosystems.

Lynch et al. (2004) proposed that a hierarchical approach, which explicitly recognized diversity, as had been used to

study traditional habitat diversity above-ground, might also be used to describe soil microbial diversity concepts. Earlier Lavelle et al. (1993) had developed a conceptual model, based on hierarchy theory, to describe decomposition within tropical forest systems, drawing from research that had explored the mechanisms connecting the large and smaller-scale processes. Biological systems of regulation based on mutualistic relationships between macro- and microorganisms ultimately determine the rates and pathways of decomposition. However, these interactions and the rates of the processes that resulted were determined by a set of hierarchically-organized factors which regulated microbial activity at decreasing scales of time and space in the following order: climate > clay mineralogy + nutrient status of soil > quality of decomposing resources > effect of macro-organisms (Lavelle et al., 1993). Studies of boundary dynamics have mostly taken place at landscape scales; however, Belnap, Hawkes, and Firestone (2003) showed how using the conceptual models from large-scale landscapes to describe the interfaces across millimeters between soil and roots and between atmosphere and soil surface as boundaries allowed their function to be assessed more effectively. They demonstrated the range of interactions occurring in three dimensions, with time as a fourth dimension and concluded that models, for example, diffusion-reaction, developed for fine-scale can also be applied at larger scales (Belnap et al., 2003). Patch mosaic and island models initially developed for mammals and birds have also been applied to the study of micro-arthropod communities on and in soil (e.g., Trekels, Driesen, & Vanschoenwinkel, 2017).

The wealth of studies on soil ecology, soil processes and their interactions with management have yielded a large collection of detailed observations many of which are site and system-specific. Systematic review and integration of these studies provide some broad general principles for

maintaining/improving soil health (Figure 1). Our aim was to draw on the approaches of landscape ecology and provide a clear integrating conceptual framework to describe the site-specific interactions driving soil properties, processes and their interaction with management. We have therefore used the checklist of issues developed by Lindenmayer et al. (2008) as a starting point for site-specific delivery of soil health (Table 2). It is also worth noting that, when conceptualized in this way, ecological systems within the soil could provide useful experimental systems to test landscape-scale hypotheses that are untestable at larger scales.

Based on the application of landscape ecology principles, here, we describe a framework in which the below-ground processes, contributing to the delivery of key soil-ecosystem functions, result from the interaction of soil habitats and their associated populations (Figure 2). Fixed site characteristics provide a template within which organisms and ecological systems operate. The structure, composition, and flows between the components (whether physical/chemical or organisms) determine the outcome and rate of the processes observed at the soil scale. Hence, soils with similar nutrient pools and biological communities but with these resources arranged in very different structures may have different crop nutrient supply characteristics and respond very differently to management, for example, tillage. This general conceptual model can then underpin consideration of the soil biological community, soil function (and the impacts of disturbance) within a diversity of natural and agricultural systems. The conceptual framework fits neatly within approaches describing soil in terms of ecosystem services and natural capital (e.g., Baveye, Baveye, & Gowdy, 2016; Dominati, Patterson, & Mackay, 2010; Robinson et al., 2013). The natural capital of soils is considered to emerge from the aggregate of its properties and processes, that is, it is an emergent characteristic of the whole soil landscape. Fischer, Lindenmayer, and Manning (2006) highlight the importance of a proper consideration of resilience alongside ecosystem function in landscape ecology. An appropriate underpinning framework, which describes the connections and interactions within the soil system, is also important to allow the capacity of the soil to resist and then recover from change (resilience) to be related to soil properties and their management.

A contrasting conceptual approach to the description of soil function has been introduced in earth system science approaches, which initially focused on the study of carbon-energy-water cycles and links from atmosphere to surface and sub-surface processes. The critical zone is the heterogeneous zone extending from the top of unweathered bedrock to the top of the vegetation canopy; currently, in this approach, soils are characterized in terms of their formation and lifecycle, which is defined both in terms of natural soil formation processes and the use/degradation of soils under human use (Banwart et al., 2012). Landscapes are not a key feature of the

current critical zone models. However, Chamorro, Giardino, Granados-Aguilar, and Price (2015) and Luo et al. (2018) have recently reviewed the progress, similarities and opportunities for integration of the depth modeling from critical zone science and the spatial considerations of landscape ecology to better inform the sustainable management of multi-functional landscapes.

In the rest of this paper, we seek to show how the conceptual model can be applied in practice by presenting the targeted development of a descriptive model to provide a structured approach to the assessment of the impacts of management practices on soil health for UK lowland agricultural systems (excluding peat soils) and also to support farmers seeking to make site-specific decisions in their management of soil health. We do not believe that it is yet possible to develop a quantitative model; however, we do not believe that farmers should have to wait for this to become possible; instead, we have taken a descriptive qualitative approach that focuses on the direction of change and describes the likely magnitude of the impact.

## 2 | DEFINING SOIL HABITATS

Soil is noted for its extreme spatial (vertical and horizontal) and temporal heterogeneity which gives rise to a wide range of surface types, aggregate and pore sizes and microclimates, and a range of resources and resource partitioning in space and time. This complexity is an obstacle to the use of single measures (e.g., pH, organic matter content) as broadly-applicable indicators of soil health and ecosystem function (Baveye et al. 2016), instead simple indicators need to be emergent or at least well linked to underlying mechanisms. The physical environment can be considered as a template on which organisms and ecological systems operate; for many soil organisms, especially micro-organisms, the architecture of the soil pore network defines the effective habitat space in soil (Young & Ritz, 2000). The amount and nature of the pore space in soil are dependent not only on soil texture but also on the aggregation of mineral particles and soil organic matter (SOM), that is, the formation and stabilization of soil structure. Most soil organisms have limited migration capacity (Fitter et al. 2005) and motility of many soil species is low compared to the scale of resource patchiness (Ettema & Wardle, 2002). Soil organisms also often enter inactive or dormant states in unfavorable conditions, so that diversity is preserved even under extreme conditions; this is analogous to the role of soil seed-banks in preserving plant diversity (Ettema & Wardle, 2002). Hence, organisms' response to the physical environment may exhibit patterns that vary between species and are constrained by the geometry of the environment (Williams, Marsh & Winter, 2002).

The plant or plant community integrates across the diversity of soil functions; in some ways, this role can be compared



to that of the top predator in above-ground systems. However, plants also have a series of roles in the below-ground ecosystem; they are affected by the interactions of organisms and their habitats below ground, but also affect them (Brussaard, 1998). Hence, plant root systems are now recognized as a dynamic and varied component of below-ground ecology and form a key habitat for a number of soil organisms including symbiotic bacteria, mycorrhizal fungi and root pathogens and herbivores (Brussaard, 1998).

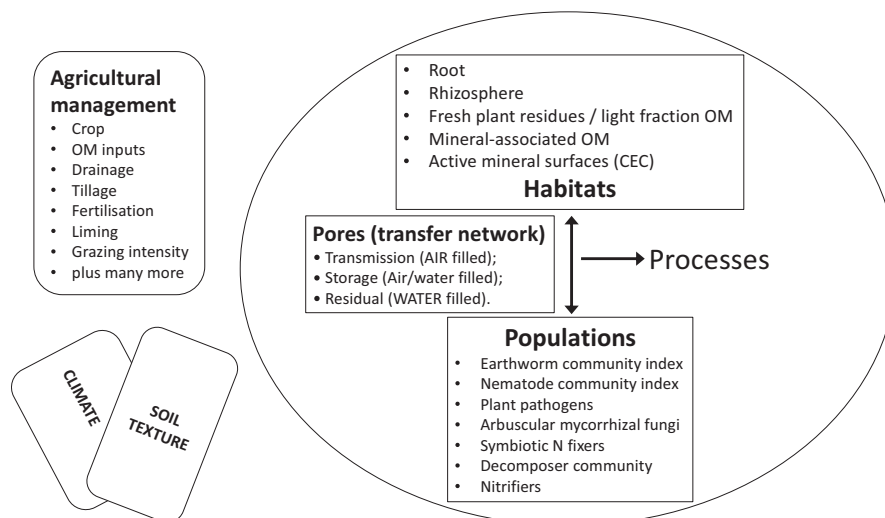
Conceptualizing soil as a series of linked habitats, rather than a single habitat for soil organisms has been shown to provide a useful representation of soil faunal populations. Following a seasonal study of nematode populations in a grassland soil, Yeates (1982) concluded that the nematode fauna observed represented the sum of numerous populations; their dynamics could not be adequately represented by either considering them as a community of interacting species nor a guild of species exploiting a single resource base. Whatever approach is used to define soil habitats, it is important that each has distinctive physical and chemical characteristics together with distinguishable communities of soil organisms. A number of conceptual approaches have been developed previously to define/divide soil into a series of distinct habitats/ecosystems. In tropical forest soils, Lavelle et al. (1993) identified four distinct below-ground ecosystems through whose activity and interaction decomposition in soil largely occurred:

- The system at the soil surface—litter and surface roots, regulated by litter arthropods and activity of surface roots;
- The rhizosphere, regulated and defined by the presence of live subterranean roots;
- The drilosphere regulated and defined by the activity of endogeic earthworms;
- The termitosphere in which the regulating macroorganisms are termites.

Beare, Coleman, Crossley, Hendrix, and Odum (1995) also considered soils to be composed of a number of distinct biologically relevant spheres of influence that defined much of the spatial and temporal heterogeneity within the soil. Building on the work of Lavelle et al. (1993), they proposed a hierarchical model, where the surface soil (drilosphere) contained hot spots of activity within the detritosphere and the bulk soil pore space (porosphere) contained distinct zones within it associated with living roots (rhizosphere) and within soil aggregates (aggregatusphere). Kuzyakov and Blagodatskaya (2015) have also recently proposed a conceptual model where the key factors driving soil processes were the amount, location and connectivity of hotspot habitats, such as the rhizosphere, detritosphere and in biopores, and the way in which the biological activity in these locations is driven by inputs of labile carbon (of varying duration). We note the challenge of Prosser (1989) based on his studies of the nitrification process in soil that it is much easier to postulate the existence of micro-environments with any required property but much more difficult to actually determine their existence and significance. Therefore, we have drawn from these previous studies to establish a descriptive framework that captures the diversity in surface types, pore sizes, and microclimates, and the range of resources and resource partitioning (Figure 3) and which can also be described semi-quantitatively, even where the amount and composition of the habitat factors cannot be measured directly.

### 3 | IDENTIFYING SOIL POPULATIONS

Given the wide range of biological taxa occurring in soil and the limited knowledge about the ecophysiology of individual species in many cases, it is often convenient to consider soil



**FIGURE 3** Descriptive model developed to provide a structured approach to the assessment of the impacts of management practices on soil health for UK lowland agricultural systems (excluding peat soils) and also to support farmers aiming to make site-specific decisions in their management of soil health. This was developed from the general model presented in Figure 2

organisms in groups. The most common grouping used from the earliest days of soil biology was according to organism size (as summarized by Swift, Heal, & Anderson, 1979). Grouping in this way has been shown to allow a consideration of soil organisms and soil processes in relation to the accessibility of pore space within soils (e.g., Killham, Amato, & Ladd, 1993). However, there is no clear correlation between the size of an animal and its trophic position or link to any other soil function, hence this classification, whilst useful, was somewhat arbitrary. The wealth of information on the soil biota has also been integrated by grouping species into trophic categories. Hunt et al. (1987) pioneered the description and modeling of the whole detrital food web using trophic groups (e.g., Beare et al. 1992; Bloem et al., 1994; De Ruiter et al., 1993). Such models have been used to demonstrate the relative importance of the fauna in carbon and nutrient cycling; for example, Brussaard, Bakker, and Olf (1996) found that soil fauna can account overall for 30%–40% of net N released into plant available forms. However, such models do not take account of non-trophic interactions—such as impacts on soil structure; Brussaard (1998) outlined a number of further problems with this type of modeling approach. More recent approaches have taken a broader approach to the definition of functional groups in both above and below-ground ecology, that is, by grouping organisms that have similar suites of functional attributes, impacts on overall ecosystem processes and where the effects of external drivers operate similarly (De Bello et al., 2010). There are an emerging number of possible ecological indices for taxa (e.g., nematodes; Ferris, Bongers, & De Goede, 2001) or integrating across taxa (e.g., microarthropods; Menta, Bonati, Staffilani, & Conti, 2017). The complexity of soil community interactions together with the limited experimental studies currently available to link soil organisms to soil functions means that an exhaustive approach to the description of soil functional groups is neither practical nor effective (Barrios, 2007). Thus, we have adapted the selective functional group approach outlined by Barrios (2007), and focus on potentially manageable soil biota and tangible functions that underpin “soil based” ecosystem services that are linked to agricultural productivity and sustainability (Figure 3).

#### 4 | DEFINING THE LANDSCAPE (COMPOSITION, STRUCTURE, AND FLOWS)

Although the definition of the landscape elements is important, the context and connectivity of these landscape elements are key to the outcome and rate of the processes observed at the soil scale. Soil structure, in particular soil pore size distribution and connectivity, controls the balance of oxygen and water available to organisms at any given soil moisture

potential, as well as regulating access of soil organisms to one another and to their resources. Greenland (1977) grouped pores in soil by size and in relation to their function in the mediation of the balance of air and water. Transmission pores are  $>30\ \mu\text{m}$  in diameter and in topsoil are usually filled with air; storage pores  $5\text{--}30\ \mu\text{m}$  in diameter are the main pores which fluctuate in air/water balance whereas residual pores  $<0.6\ \mu\text{m}$  are commonly full of water, though plant roots can effectively empty pores down to  $0.2\ \mu\text{m}$ . Elliott, Anderson, Coleman, and Cole (1980) introduced the concept of habitable pore space in facilitating trophic interactions in soil, whereby protists such as amoebae can graze bacteria in pores too small to allow nematodes in, but when the amoebae enter larger pores they can be consumed by nematodes so ensuring the transfer of food and energy up through the food web. This was later developed by Postma and van Veen (1990) to describe the survival of bacteria in water-filled pores, such that pores  $<0.8\ \mu\text{m}$  diameter are inaccessible (too small for bacteria to enter),  $0.8\text{--}3\ \mu\text{m}$  are protective (bacteria can enter but not bacterial-feeding fauna) and pores  $>3\ \mu\text{m}$  are variously accessible (so  $3\text{--}6\ \mu\text{m}$  would be available to small protists,  $6\text{--}30\ \mu\text{m}$  to slightly larger protists and  $>30\ \mu\text{m}$  to nematodes). Killham et al. (1993) and Carson et al. (2010) have demonstrated how, as a consequence of the interactions resulting from accessibility and flows of water/nutrients, pore connectivity could explain the high diversity of bacteria in soil. More recently, Kravchenko and Guber (2017) have shown the importance of transmission pores in controlling decomposition rates. Hence, we consider that, whilst not complete, including a simple description of pore size distribution in the descriptive framework allows the transport network in soils to be considered explicitly and the impacts of management practices on connectivity of populations and habitats to be assessed (Figure 3).

#### 5 | FIXED SITE FACTORS SETTING CONSTRAINTS

The potential use of land in any location, whether for agriculture or another use, is rarely unconstrained. Assessment of land use quality usually uses a number of relatively fixed site characteristics to define the quality of land (e.g., climate, slope, some soil factors; FAO 1972). These site factors are largely unmanageable, and consequently, they set the boundary for the range of agricultural practices that are possible. The fixed site factors also constrain the range of plant species likely to be present and determine the potential net primary production of that plant community. A similar range of fixed factors (climate, depth, stoniness, mineralogy, texture) has also been identified as controlling the maximum potential SOM (Dick & Gregorich, 2004; Ingram & Fernandes, 2001) and soil microbial biomass (Gonzalez-Quiñones et al., 2011).

Texture group	Sandy and light silty	Medium	Heavy
Range of clay content	<18%	18%–35%	>35%
Texture class (from Soil Survey of England and Wales classification)	Sand, loamy sand, sandy loam, sandy silt loam, silt loam	Sandy clay loam, clay loam, silty clay loam	Sandy clay, clay, silty clay
Root Rhizosphere Fresh OM inputs	These are plant and input-driven properties dominantly controlled by crop choice, residue management and inputs in agricultural systems		
Mineral-associated SOM	Low	Moderate	High
Active mineral surfaces (CEC)	Low	Moderate	High
Transmission pores	High	Moderate	Low
Storage pores	Low	High	Moderate
Residual pores	Low	Moderate	High

**TABLE 3** Texture groups used in the descriptive model with notes on the implications for soil habitat characteristics

The previously limited number of biogeographical studies of below-ground organisms (largely completed only for mites, e.g., Luxton 1996; collembola, e.g., Christiansen & Bellinger, 1995 and earthworms, e.g., Reynolds, 1994) is rapidly being updated with the revolution in DNA sequencing technology (e.g., for: bacteria, Mailik, Thomson, Whiteley, Bailey, & Griffiths, 2017; fungi, Tedersoo et al., 2014; protists, Grossmann et al., 2016; and; nematodes, Song et al., 2017). These show that site factors are often a major determinant in the development of below-ground communities. Hence, there is potential for some sites always to have higher size, activity, and diversity of below-ground communities than others as a result of its combination of fixed site factors. Here, we have used simple groupings of UK regional climates (Met Office, 2016) and topsoil texture classes developed for agricultural purposes (Natural England, 2008) as input factors for the descriptive framework (Figure 3). The impact of the texture groups on baseline soil habitat characteristics is substantial, in particular with regard to active surface area (CEC) and pore size distribution; this has significant implications for water balance, buffering and hence also for the size and activity of soil populations (Table 3).

## 6 | USING THE CONCEPTUAL FRAMEWORK TO PREDICT THE IMPACTS OF AGRICULTURAL MANAGEMENT

Baseline states for the typical range of farming systems in the UK lowlands were developed using data taken from the literature summarized in Stockdale et al. (2006) and the understanding embedded in the conceptual framework

(Table 4). Expert knowledge of the authors drawing from an understanding of the underlying principles of soil/plant/organism/environment interactions was needed to interpolate and extend the results from the literature review to give a full coverage for UK agricultural systems. The habitat and population baselines have been compared to data from empirical studies. However, most of the available data does not cover the full gradient of agricultural management and only compares two extreme agricultural land management systems, for example, 56 earthworms per m<sup>2</sup> in intensive cropping systems compared with 229/m<sup>2</sup> in grassland (Spurgeon, Keith, Schmidt, Lammertsma, & Faber, 2013). Griffiths et al. (2010) showed that there were significant differences between earthworm populations in ley-arable (0.8 g earthworm biomass/kg soil) and grassland systems (3.5 g earthworm biomass/kg soil). For the decomposer community, Merino, Pérez-Batallón, and Macías (2004) found only small differences between soil microbial biomass in intensive arable and regularly reseeded intensive grassland systems, 224 mg C/kg compared with 276 mg C/kg respectively; Gosling, van der Gast, and Bending (2017) using PLFA methods found a larger difference between intensive arable and grassland systems (29 compared with 64 nmol/g PLFA, respectively).

The underpinning literature review drawing together the impacts of agricultural management on soil habitats and populations is also taken from Stockdale et al. (2006) and the full detail of that review is not repeated here. The summary of findings considering the impacts of a range of management practices typical of UK farming systems on soil populations and habitats are given in Table 5. Thus, we can confidently say what the typical effects of management changes are in descriptive terms, such as “in general,

**TABLE 4** Typical farming systems of the UK lowlands described in terms of the range of management practices and notes on the implications for soil habitats and soil populations of earthworms and decomposers (as an example) on a medium texture soil

Farming system type	Cropping field vegetable	Cropping combinable crops	Cropping ley-arable	Grassland lowland
Typical management practices				
Crops	Onions, carrots, potatoes, brassica, in rotation with cereals	Cereals (winter and spring sown), oilseed rape, beans Some use of cover crops	Cereals (winter and spring sown), oilseed rape, beans but with 2–3 year grass or grass-clover leys in rotation	Dominantly rye-grass swards; some grass-clover
Tillage	Intensive and often repeated in-season	Rotational ploughing and non-inversion approaches Some direct-drill	Rotational ploughing and non-inversion approaches	Occasional for grassland reseed
Livestock	None	Rare	Leys grazed by sheep/cattle	Regular grazing often intensive by cattle/sheep
Residue management	Incorporation	Incorporation; cereal straw may be baled and sold	Cereal straw baled for bedding	n/a
Soil habitats and populations				
Roots	Annual cropping; some fallow periods	Annual cropping; some active growth all year	Rotation of annual and perennial; often active growth all year. Some years with high root density.	Perennial; high root density
Rhizosphere	Very simple rhizosphere development; some short duration and incomplete.	Simple rhizosphere succession but including crop senescence. Main rhizosphere development phase in spring	Marked differences in annual vs. perennial rotational phases	All rhizosphere ages present simultaneously; repeated biopore systems develop.
Fresh OM inputs	Diverse crop residues OM inputs often restricted by customer protocol.	Mainly senesced crop residues (straw/haulm) Often limited OM inputs available	Marked differences in annual vs. perennial rotational phases Inputs of dung during grazing; manures/slurries	Continual input from leaf fall and root senescence Inputs of dung during grazing; manures/slurries
Earthworms	Few	Some, but mainly topsoil-dwelling Deep-burrowing species increase as tillage intensity decreases	Moderate, differences between rotational phases, some deep-burrowing	Many, diverse community with all functional groups present
Decomposers	Low/moderate Residues may persist for one season	Moderate Residues sometimes persist for one season	Good Residues rarely persist	Very good Rapid cycling of added organic matter

inversion tillage will reduce earthworm abundance”; “the addition of organic matter will increase the stability of transmission and storage pores.” These general predictions vary according to landscape factors and so the expert opinion of the project partners was used to judge how the effect of each management practice is moderated or exaggerated by:

- Topsoil texture group.
- UK regional climate (simplified to cold & wet; cold & dry; warm & wet; warm & dry).
- Main agricultural systems.

As an example, effects of management options on earthworms are likely to be more pronounced in heavy soils (+1) and

**TABLE 5** Summary of impacts of some key agricultural management practices on soil populations, arising both directly and indirectly as a result of impacts via the soil landscape

Indirect effects on structure, composition and flows within the habitat mosaic									
Habitats									
Practice	Direct effects on populations	Mineral-associated OM			Pore network (connectivity, flows)				
		Roots	Rhizosphere	Fresh OM inputs	Transmission pores	Storage pores	Residual pores	Chemical environment	
Tillage	Kills soil macrofauna, earthworms, and beetles	Destroys/damages root systems	Stimulates mineralization	Mixes and blends materials into soil but can slow decomposition rate	New surfaces may be exposed to decomposers	Increase	Reduces connectivity to depth; may decrease	Increases	Aerates and allows oxidation
Rotation of a variety of crops		Diversity of structure and depth	Diversity of inputs in space/time	Variety of crop residues, could give allelopathic effects		Changes will vary through the rotation. Inclusion of deep rooting crops will increase these pore sizes at depth.			Variety of nutrient uptake demand patterns in space/time
Incorporation of crop residues	Rapid decomposition can control some pathogens		Stimulate or reduce mineralization depending on quality	Increase Location within the soil depends on method of incorporation		Increase if aggregation; increased stability			Rapid decomposition can lead to development of anaerobic microsites
Herbicides		Kills roots	Increase dead root materials	Increase		May increase formation due to higher turnover of roots			
Grazing		Fertilizer effect of excreta stimulates growth	Defoliation stimulates root exudation	Dung inputs		Reduce if compaction occurs	Distribution may change if compaction occurs	Increase if compaction occurs	Hotspots of N, P, K associated with excreta
Lime		May increase		May increase if yield increases					Increase in pH; change in element availability
Drainage	Big positive impact on nitrifiers; negative impact on anaerobic metabolism	Increase	Increase	More rapidly decomposed		Drainage may modify pore size distribution and connectivity			

(Continues)



TABLE 5 (Continued)

Indirect effects on structure, composition and flows within the habitat mosaic						
Habitats			Pore network (connectivity, flows)			
Practice	Direct effects on populations	Roots	Rhizosphere	Fresh OM inputs	Mineral-associated OM	Chemical environment
Fertilizer	High soluble P restricts AM fungi	Increase	Increase root exudation	Increase as yield increases		May decrease pH (particularly NH <sub>4</sub> and S-based) High salt content of pore water short-term
Slurry	High NH <sub>4</sub> levels can control some pathogens	Increase		Increase as yield increases, small direct effect		Can give high salt content of pore water short-term
Composted materials		Improved rooting distribution		Directly increase but usually slow to decompose further	May increase as compost rapidly stabilized	Usually have liming effect

less pronounced in light soils (−1). For earthworms, there have been a number of recent meta-analyses looking at the effects of tillage on earthworms and also on the beneficial effects of earthworms on crop yield. Hence, the impacts of management practices, especially but not only tillage, predicted in the model for earthworms (by ecological group) can be compared with the empirical findings summarized by van Capelle, Schrader, and Brunotte (2012) and broken down according to soil type (sand, silt, clay, loam), extent of tillage (conventional plough, minimum tillage, no tillage) and by the different ecological groups of earthworms. The interacting populations, resources and habitats detailed in the descriptive model provides transparency for the derivation of a more specific and detailed qualitative description of the biological interactions at any site (defined by its combination of climate, soil, and cropping system type) from the understanding of soil processes that captures the average/most likely result.

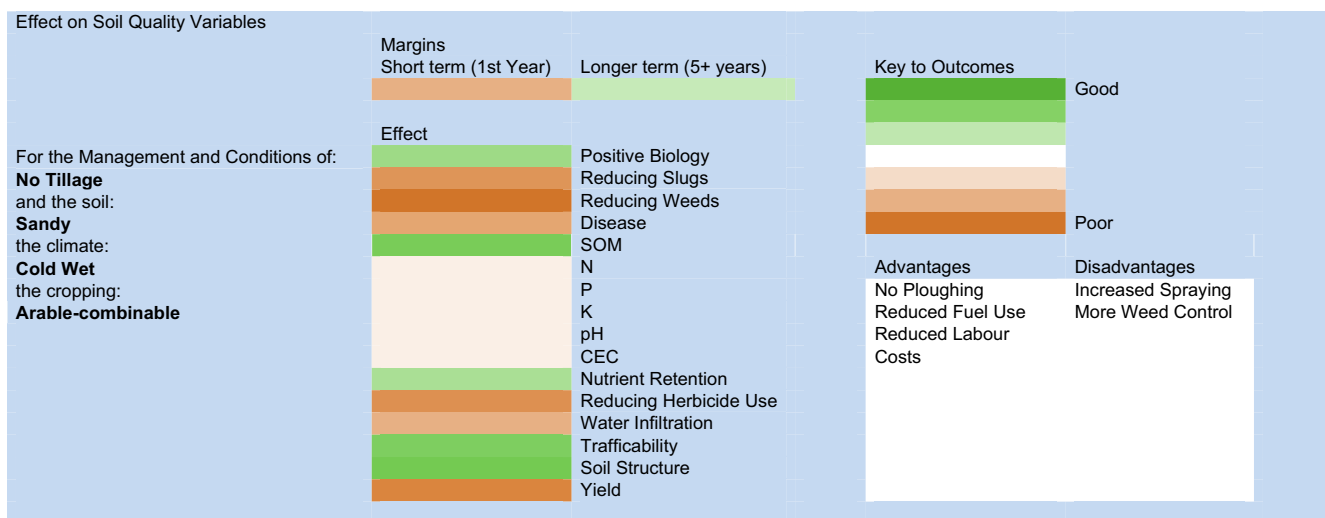
## 7 | SUPPORTING SITE-SPECIFIC MANAGEMENT ON-FARM

Links between soil populations and crop growth and yield are becoming increasingly well-established. van Groenigen et al. (2014) has quantified the increase in crop yield due to the earthworm effect for different crop types and environmental conditions. Similar quantifiable beneficial effects on crop yield have also been shown for the nematode community (Gebremikael, Steel, Buchan, Bert, & De Neve, 2016), which includes bacterial-feeders, fungal-feeders, omnivores and predators and not simply the plant-parasites that are the focus of many soil tests. Overall, there is now a sound evidence base for the link between soil biology, soil health, and ecosystem services. Consequently, there is an opportunity to communicate more effectively with farmers with regard to the impacts of changes in management. Use of the conceptual framework and the descriptive model developed from it allows us to take account of the differences between sites and systems more effectively and provide information about the underlying drivers and mechanisms (Table 6); this adds detail to the more common approach of providing general descriptions supported by site-specific case studies.

Here, we have also begun to develop the descriptive model to give a visual presentation of the impacts of management change (Figure 4); this provides information on the impacts on physical, chemical and biological aspects of soil for a location/system/soil in an integrated way together with their relationships to sustainable agricultural production. However, these effects still need user interpretation to be truly site-specific. The example given in Figure 4 shows a likely increase in slugs and weeds following a conversion to no-tillage in a combinable arable system under a cold-wet climate, but user knowledge of the inherent slug and weed burden in that

**TABLE 6** Summary of the model described impacts of changes in tillage practice in cropping systems (combinable crops) on soil organisms and soil properties that lead to impacts on crop production and environmental impacts

Land management practice	Direct impacts on soil organisms	Other impacts on soil properties with indirect impacts on soil organisms	Factors that might modify expected outcomes
Minimum intensity tillage	All tillage operations kill soil macrofauna—largest impacts on earthworms and beetles; reduced numbers of tillage operations lead to significant increases in earthworm populations	All tillage operations that mix soil reduce connectivity of transmission pores to depth Changes pore size distribution, disrupts pore connectivity changing water movement in soil Mixes organic matter inputs throughout tilled soil Improve soil structure—reducing sediment loss	If operations are carried out in less than optimal conditions through the whole soil profile, structural re-arrangement below the cultivation layer can lead to compaction, reduced rooting and through flow and ponding within the surface soil horizons even in sandy soils Sandy soils have naturally high macroporosity and longer workability windows; hence, changes in tillage may not give all the expected benefits
No-till compared with minimum tillage	Allows development of deep-burrowing (anecic) earthworm populations toward site carry capacity	Increases connectivity of transmission macropores from surface to depth increasing rates of infiltration Increased profile stratification; higher organic matter contents in surface soils Surface mulch of residues provides more suitable end of season habitat for surface dwelling arthropods Improves soil structure and its stability—reducing sediment loss	Unless no-till is adopted as part of a full package of conservation agriculture measures including increased inputs of organic matter (cover crops), then the described impacts may not be realized



**FIGURE 4** An example of the output from the predictive framework showing soil health outcomes resulting from a change from conventional to no-tillage in a combinable arable system under a cold-wet climate. Biological parameters were grouped into those considered to be positive for crop production: earthworms, microbial biomass, microbial activity, mycorrhizae, soil biota and also reported separately on biological interactions that would be detrimental to agricultural production (i.e., slugs, weeds, and disease). Chemical parameters were also divided into positive (N, P, K, pH, and CEC) and negative (nutrient loss and leaching, herbicide use) attributes for agriculture. Physical parameters related specifically to soil structure, infiltration, and trafficability, which have important practical consequences. Both crop yield and gross margin are included following preliminary interactions with growers (as an example, when compared with conventional inversion tillage, reducing tillage can often lead to a reduced absolute yield, however, the gross margin can improve because of lower establishment costs). Outcomes are shown in a traffic-light format and range semi-quantitatively from an improvement (green) through to a degradation (red)

field would further improve the predictive outcome of the framework. The descriptive model is under development as part of a research and knowledge exchange program which will now evaluate the qualitative predictions made for the impacts of site-specific in-field management systems against measures of physical, chemical, and biological indicators of soil health together with crop yield and quality. We consider that the active engagement of farmers will be critical to help shape this descriptive model into a useful decision-support tool.

On-farm many practices that affect soil health are adopted for several reasons, including economic or management drivers as well as reflecting a concern for soil biological function; consequently, there is a need to provide this breadth of information that can be used to guide uptake and assess cost-effectiveness in the local context. This approach will be strengthened as tools become increasingly available on-farm to measure soil biological indicators together with chemical and physical indicators as an assessment of soil health (Bünemann et al., 2018). Deugd, Röling, and Smaling (1998) stress the most effective approach to improve site-specific management of soil/nutrient use on-farm is to support innovation by increasing farmers' control over the processes of research and emphasizing the process of learning rather than the teaching of content. Such an approach works best where the main blockage is not access to information, but rather farmers' adoption, understanding and integration of that knowledge into practice. Knowledge exchange work on-farm regularly highlights that there are a significant minority of farmers already investing in the development of techniques to improve soil health by taking up practices which require more of their time and which may also have required significant capital investment. However, even these farmers asked for more input, particularly more information and tools that could support them to make more effective decisions for their farming system and evaluate the impact of practices in place. The value of a clear systematized conceptual framework is to provide an underpinning descriptive model that will allow innovations developed and tested on-farm to be evaluated robustly and will allow farmers to become partners in an adaptive research and knowledge exchange process (Sherwood & Uphoff, 2000).

## ACKNOWLEDGMENTS

We acknowledge funding support from Natural England (2010–2011), which allowed the first steps in the development of the descriptive model. Current developments are funded as part of the AHDB- and BBRO-funded Soil Biology and Soil Health Research and Knowledge Exchange Partnership (2017–2021). We would like to thank all those farmers, agronomists, and fellow researchers who have challenged our thinking and contributed to the development of our understanding.

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**How to cite this article:** Stockdale EA, Griffiths BS, Hargreaves PR, Bhogal A, Crotty FV, Watson CA. Conceptual framework underpinning management of soil health—supporting site-specific delivery of sustainable agro-ecosystems. *Food Energy Secur.* 2018;e00158. <https://doi.org/10.1002/fes3.158>