JNCC Report
No. 364

Do farm management practices alter below-ground biodiversity and ecosystem function? Implications for sustainable land management

E. A. Stockdale, C.A. Watson, H. I. J. Black, L. Philipps

August 2006

ISSN 0963 8091

For further information please contact:

Patricia Bruneau - Soils Lead Co-ordination Network/JNCC
Scottish Natural Heritage
2 Anderson Place
Edinburgh
EH6 5NP

This report should be cited as:

Stockdale, E.A., Watson, C.A., Black, H.I.J., Philipps L. 2006.

Do farm management practices alter below-ground biodiversity and ecosystem function? Implications for sustainable land management.

JNCC report no. 364

Contents

Executive Summary

Organic farming practices have recently been shown to benefit a wide range of above-ground taxa through increases in abundance and/or species richness. In relation to policy development, there is a parallel need to collate evidence on whether management practices that contribute to this effect above-ground (e.g. prohibition/reduced use of chemical pesticides and inorganic fertilizers) similarly effect below-ground biodiversity. Through a combined process of literature review and stakeholder interaction through expert groups this report addresses this issue but also considers the effects of below-ground biodiversity on the maintenance and enhancement of a range of ecosystem services.

Reported evidence suggests below-ground biodiversity benefits associated with reducing the intensity of use of mechanical and manufactured inputs in lowland systems. Evidence from hill and upland systems is insufficient to allow such conclusions. However, the evidence is not strong enough to draw conclusions about the effects of farming systems *per se* (e.g. organic versus integrated). This in part reflects the limitations of experimental design and the difficulties of transferring the results of reductionist research approaches to practical agriculture. This relates specifically to interactions between individual practices associated with producing a particular crop/crop sequence. Furthermore, it is clear that best practice is likely to be farm and even micro-site specific due to the complexity of interactions between soil organisms, soil type, weather and management factors. There is a critical need to address how current knowledge can be translated into guidance for land managers and policy makers.

Technical Summary

Soils contain a very high diversity of organisms; many of which remain unknown or, at least, little studied. There is also extreme spatial (vertical and horizontal) and temporal heterogeneity in soil which gives rise to a wide range of surface types, pore sizes and microclimates, and a range of resources and resource partitioning in space and time. Soil organisms not only occupy soil; they are a living part of it and as a result of their interacting activities also change it.

The scientific community has come to a broad consensus on many aspects of the relationship between biodiversity and ecosystem functioning, including many points relevant to management of ecosystems. Further progress will require . To strengthen links to policy and management, we also need to integrate our ecological knowledge of the below-ground ecosystem with understanding of the practical, social and economic constraints of potential management practices. Understanding this complexity, while taking strong steps to minimise negative impacts on below-ground functioning, is necessary to support sustainable soil function.

Based on our review of the scientific literature, we are certain of the following conclusions:

1. Species' functional characteristics strongly influence ecosystem properties. Functional characteristics operate in a variety of contexts, including effects of dominant species, keystone species', ecological engineers, and interactions among species (e.g., competition, facilitation, mutualism, disease, and predation). Relative abundance alone is not always a good predictor of the ecosystem-level importance of a species, as even relatively rare species (e.g., a keystone predator) can strongly influence pathways of energy and material flows.
2. Alteration of biota in ecosystems via species invasions and extinctions caused by human activities has altered ecosystem goods and services in many well-documented cases. Many of these changes are difficult, expensive, or impossible to reverse or fix with technological solutions.
3. The effects of species loss or changes in composition, and the mechanisms by which the effects manifest themselves, can differ among ecosystem properties, ecosystem types, and pathways of potential community change.
4. Some ecosystem properties are initially insensitive to species loss because (a) ecosystems may have multiple species that carry out similar functional roles, (b) some species may contribute relatively little to ecosystem properties, or (c) properties may be primarily controlled by abiotic environmental conditions.
5. More species are needed to insure a stable supply of ecosystem goods and services as spatial and temporal variability increases, which typically occurs as longer time periods and larger areas are considered.

We have high confidence in the following conclusions:

1. Certain combinations of species are complementary in their patterns of resource use and can increase average rates of productivity and nutrient retention. At the same time, environmental conditions can influence the importance of complementarity in structuring communities. Identification of which and how many species act in a complementary way in complex communities is just beginning.
2. Susceptibility to invasion by exotic species is strongly influenced by species composition and, under similar environmental conditions, generally decreases with increasing species richness. However, several other factors, such as propagule pressure, disturbance regime, and resource availability also strongly influence invasion success and often override effects of species richness in comparisons across different sites or ecosystems.
3. Having a range of species that respond differently to different environmental perturbations can stabilize ecosystem process rates in response to disturbances and variation in abiotic conditions. Using practices that maintain a diversity of organisms of different functional effect and functional response types will help preserve a range of management options.

Uncertainties remain and further research is necessary in the following areas:

1. Further resolution of the relationships among taxonomic diversity, functional diversity, and community structure is important for identifying mechanisms of biodiversity effects.
2. Multiple trophic levels are common to ecosystems but have been understudied in biodiversity/ecosystem functioning research. The response of ecosystem properties to varying composition and diversity of consumer organisms is much more complex than responses seen in experiments that vary only the diversity of primary producers.
3. Theoretical work on stability has outpaced experimental, work, especially field research. We need long-term experiments to be able to assess temporal stability, as well as experimental perturbations to assess response to and recovery from a variety of disturbances. Design and analysis of such experiments must account for several factors that covary with species diversity.
4. Because biodiversity both responds to and influences ecosystem properties, understanding the feedbacks involved is necessary to integrate results from experimental communities with patterns seen at broader scales. Likely patterns of extinction and invasion need to be linked to different drivers of global change, the forces that structure communities, and controls on ecosystem properties for the development of effective management and conservation strategies.
5. This paper focuses primarily on terrestrial systems, with some coverage of freshwater systems, because that is where most empirical and theoretical study has focused. While the fundamental principles described here should apply to marine systems, further study of that realm is necessary.

Despite some uncertainties about the mechanisms and circumstances under which diversity influences ecosystem properties, incorporating diversity effects into policy and management is essential, especially in making decisions involving large temporal and spatial scales.

Sacrificing those aspects of ecosystems that are difficult or impossible to reconstruct, such as diversity, simply because we are not yet certain about the extent and mechanisms by which they affect ecosystem properties, will restrict future management options even further. It is incumbent upon ecologists to communicate this need, and the values that can derive from such a perspective, to those charged with economic and policy decision-making.

1. Quantifying the diversity of soil organisms is consequently problematic, for many organisms an appropriate taxonomic framework is weak or absent. give possibility for new approaches. It is likely that that application of molecular methods could rapidly increase our understanding of the diversity and ecology. But only if applied in robust..

2. Consideration of inter-organism interactions and their relation to function can only take an understanding of ecological relations below ground so far; it is essential to also integrate spatial habitat factors. Spatial variability has been often treated as distracting "noise" which obscures the key relationships between structure and function of below-ground biodiversity, however, it is likely that understanding the control over ecological systems imposed by spatial variability is the key to improving our ability to manage below-ground ecosystems. Such integration is currently more or less absent and presents a major interdisciplinary challenge for soil science.

3. The development of landscape ecology has re-emphasized the importance of spatial patterns in constraining any ecological processes. However, application of landscape ecology approaches to below-ground ecology will not be easy. The soil landscape is much more temporally variable than landscapes above-ground. Nonetheless this is an area that would repay interdisciplinary research. Small scale landscapes within the soil could provide useful experimental systems which might be used to test hypotheses that are untestable at larger scales.

4. Compared to the work carried out on diversity-function relationships for plant species, relatively few studies have been carried out below ground. Relationships between soil biodiversity and ecosystem functions range from positive to neutral or even negative. The main biotic controls over ecosystem function result from species traits, changes in species composition and changes in the multi-trophic interactions that occur in soil. Consequently it is unlikely that consistent diversity effects *per se* will be observed. Relationships between diversity, resistance and resilience following disturbance are also not well understood below ground. More work is needed to determine whether a minimum number of functional groups and/or species within functional groups are needed to maintain process rates following disturbance or is the presence/absence of certain species decisive alone?

5. Decomposition is the result of the intermeshing vital processes of many soil organisms and is a central process for the delivery of most ecosystem services, along with the formation and stabilisation of soil structure. While the precise role of many below-ground organisms in relation to soil processes is not fully known, functional groups provide a useful frame to describe interactions and make links between below-ground ecology and soil functions. Increased mechanistic understanding of ecological

interactions is needed if the effects of human management (intentional and unintentional) are to be evaluated and remedied.

1. Introduction

This review was initiated to address the need to understand the role of below-ground biodiversity in agricultural systems. It complements the recent review by Hole *et al.* (2005) that highlighted that a wide range of mainly above–ground taxa, including birds and mammals, invertebrates and arable flora, benefit from organic management of land through increases in abundance and/or species richness. Hole *et al.* (2005) identified three broad management practices (prohibition/reduced use of chemical pesticides and inorganic fertilisers; sympathetic management of non-cropped habitats; and preservation of mixed farming) that are typical of organic farming and particularly beneficial for farmland wildlife from their review of comparative studies of conventional and organic farming systems. For below-ground biodiversity, a focus simply on the biodiversity of below-ground species misses the important consideration of the contribution of below-ground biological processes to the maintenance and enhancement of a range of ecosystem services. This review therefore seeks to consider the evidence for the impact on below ground biodiversity of agricultural management practices as well as different farming systems including both conventional and organic farming systems.

1.1. Organic farming systems policy context

1.1.1. Definition of organic farming systems

Increased consumer awareness of food safety issues and environmental concerns has contributed to the growth in development of a number of 'sustainable' farming systems over the last few years. Organic farming has become understood as part of a sustainable farming options and a viable alternative to the more "high input- high output" approaches to agriculture. For the purpose of this report the term 'conventional agriculture' refers to a system of industrialised agriculture that maximizes productivity by the use of synthetic inputs such as fertilisers and pesticides, these farms have become increasingly specialised.

The International Federation of Organic Agriculture Movements (IFOAM) defines organic farming systems in terms of four basic principles:

- principle of Health that should sustain and enhance the health of soil, plant, animal, human and planet as one and indivisible;
- principle of Ecology that should be based upon living ecological systems and cycles work with them, emulate them and help sustain them;
- principle of Fairness that should build upon relationships that ensures fairness with regard to the common environment and life opportunities;
- principle of Care should be based on a precautionary and responsible manner to protect the health and well being of the current and future generations and the environment.

Organic farming has a clear legislative basis and certification schemes for both production and processing. The legal basis of organic food certification in the UK is the EU regulations 2092/91 and 1804/99. This certification is carried out through a number of certification bodies approved by the Department of Environment Food and Rural Affairs (Defra), advised by the Advisory Committee on Organic Standards (ACOS) and UKAS (Defra 2003a). Organic farming systems are not homogeneous and a number of variants can be identified. In particular biodynamic farming is distinguishable. Biodynamic agriculture is a system of organic farming developed by the Austrian scientist and philosopher Rudolf Steiner in the early part of the 20th century. Biodynamic farming takes into consideration both the biological cycles and the metaphysical or spiritual aspects of the farm.

1.1.2. European perspective

The sustainability of both agriculture and the environment is a key policy objective of today's common agricultural policy (the 'CAP'): 'Sustainable development must encompass food production alongside conservation of finite resources and protection of the natural environment so that the needs of people living today can be met without compromising the ability of future generations to meet their own needs.' (European Commission 2004).

Since the 1992 reform of the CAP, the number of organic farms has increased dramatically in all European Member States. In total, just less than 2 % of all agriculture area is devoted to organic farming, on more than 1% of all agriculture holdings. In general, organic farms are larger than average; however the situation varies considerably from one country to another. Production of grass as fodder is by far the most important use of organic land, though horticulture is important in Southern Europe.

This objective requires farmers to consider the effect that their activities will have on the future of agriculture and how the systems they employ shape the environment. As a consequence, farmers, consumers and policy makers have shown a renewed interest in organic farming.

In 2004 the European Commission produced a ‘European Action Plan for Organic Food and Farming’ (European Commission 2004). The action plan recognised organic farming’s contribution to both the development of the market and that organic land management delivers public goods primarily in terms of environmental but also rural development benefits.

1.1.3. UK perspective

At a time when many sections of the agricultural industry are financially depressed and under serious pressure organic farming is expanding. The UK organic market has increased rapidly in recent years, with growth rates of 30% to 50% per annum. Sales in 2000/01 amounted to £802 million, up by 33% on the previous year. The total value of the UK organic retail market is now over £ 1.2 billion with over 4 % of agricultural land in the UK being organic

(Defra 2005a). Despite the recent dramatic growth rates, organic still represents a small proportion of the total food sector, and many factors influence supply and demand. The continued growth of organic farming depends upon the long-term economic viability of organic production, which is dependent on adequate consumer demand for products, on meeting food safety criteria, and importantly upon its ability to meet public expectations of the environmental benefits of such systems. The UK Government believes that financial support for organic farming is justified by the environmental public good which organic farming delivers, which extend to society as a whole and not just to the minority of consumers who choose to purchase organic food.

Organic Action Plan for England

In response to the Policy Commission Report on the Future of Farming and Food (2002), Defra established an Action Plan to Develop Organic Food and Farming in England (Defra 2003b). On the basis of comparing average conventional and organic farms, organic farming is generally accepted to produce the following environmental benefits it results in:

- higher levels of biodiversity;
- lower environmental pollution from pesticides;
- through lower use of energy inputs it contributes to reduction of carbon dioxide emissions;
- because of reduced reliance on external materials it produces smaller quantities of controlled waste and so contributes to waste reduction;.
- organic farming also produces social and economic benefits as organic food is produced to legally enforceable standards and is subject to tight controls on inputs and an official inspection and accreditation system; it therefore meets demands from an increasing number of consumers for high standards of assurance about production methods;
- consumers taking a closer interest in how land is farmed and, in the context of its particular contribution to local food marketing, can help to develop a sense of community between buyer and seller, town and country;
- high standards of animal welfare;
- benefits to rural employment through the particular farming practices used and through it’s tendency to encourage the development of new marketing systems.

Organic Action Plan for Scotland

In 2003 the Scottish Executive launched an Organic Action Plan for Scotland (Scottish Executive 2003) which recognises the importance of the organic sector in Scotland and the opportunities for market development and environmental benefits that organic production can deliver. The recent growth in organic farming is related to both the current financial pressures on agriculture and changing public views of its expectations of the agricultural industries. In Scotland, 359,615 ha of agricultural land is now in-conversion or farmed organically by 632 farmers (Defra 2005). The majority of organic land in Scotland is pasture or rough grazing, reflecting the relative ease of converting extensive systems and greater benefits in the past from area based support payments. The Organic Action Plan recognized the need to develop the arable and horticultural sectors of the market by setting production targets and a differential incentive scheme.

Organic Action Plan for Wales

Under the Organic Action Plan prepared by the Welsh Agri-Food Partnership in 1999 a target was set of 10% of the Welsh agricultural sector to be organic by 2005. Many of the aims of the first Welsh Organic Action Plan have been fully or at least partly achieved, although production issues such as training and supply chain anomalies need further work. However, it is generally recognized that priority should now be given to supporting the hard work of the organic sector stakeholders; this should aim to raise awareness of the benefits to society in general, and to individuals in particular, of growing and eating a greater proportion of food as organic.

1.2. Biodiversity policy context

Policy support for organic farming is based dominantly on evidence relating to above-ground impacts on biodiversity (e.g. Hole *et al.* 2005). However, much of this evidence base is derived from comparisons made in lowland arable or mixed farming systems. Biodiversity in arable weeds in particular has been shown to increase in response to organic management (Bengtsson *et al.* 2005, Fuller *et al.* 2005), positive effects have also been detected for species diversity of butterflies (Feber 1997), soil organisms, predatory invertebrates, bats and birds (Bengtsson *et al.* 2005, Fuller *et al.* 2005). There are several features of organic management that are likely to cause these underlying differences, not least the absence of synthetic pesticides and artificial fertilizers (the latter are likely to be a significant factor in grass-dominated systems). In addition, the greater diversity of land-use (especially through rotations where arable crops are grown) and the generally higher quality non-crop habitat that is typical of organic farming is likely to contribute significantly to farmland biodiversity. For example, hedgerow structure is likely to be a major contributor to significant overall differences between farming systems for several bird species (Chamberlain *et al.* 1999). However, it should be noted that most large-scale studies of biodiversity on organic systems have been biased towards (e.g. Chamberlain *et al.* 1999), or targeted upon, lowland arable systems (e.g. Fuller *et al.* 2005). Bengtsson *et al.* (2005) concluded that diversity of soil organisms, including fungi, tended be higher under organic management.

1.3. Soil policy context

Soil quality has recently moved up the policy agenda, with the introduction of a number of initiatives:

- EU Initiative on Soil Protection to be launched in the spring 2006 (European Commission 2002);
- Soil Action Plan for England, May 2004 (Defra 2004a);
- SEERAD are currently funding a research project to gather evidence on the threats and pressures on soils in Scotland. This report will inform the development of soils policy in Scotland (Scottish Executive 2006);
- The Welsh Assembly Government Environment Strategy and associated action plan, which contains information on future soil management policy, will be published in May 2006.

1.3.1. European Soil Policy

The Council of Europe's European Soil Charter (1972) recognised the importance of the soil resource. Since then European countries have undertaken various activities to better protect their soil. In response to concerns about the degradation of soils in the EU, the European Commission adopted a Communication "Towards a Thematic Strategy for Soil Protection" in April 2002 (European Commission 2002). This identified a number of threats to soil across the EU states:

- erosion;
- decline in organic matter;
- local and diffuse contamination;
- sealing;
- compaction;
- decline in biodiversity;
- salinisation;
- landslides.

1.3.2. UK Soil Policy

The Welsh Strategy for Soils and Soil Protection Strategy for Scotland are not yet as well advanced as the Soil Action Plan for England.

The actions proposed in the First Soil Action Plan for England (Defra 2004a) work towards a common vision that recognises the several vital functions that soils perform for society: "*Our vision is to ensure that England's soils will be protected and managed to optimise the varied functions that soils perform for society (e.g. supporting agriculture and forestry,* *protecting cultural heritage, supporting biodiversity, as a platform for construction), in keeping with the principles of sustainable development and on the basis of sound evidence."*

In order to achieve this vision, our aims are to ensure:

- *soil managers will look after their soils with a view both to their own and society's short-term needs and to the interests of future generations;*
- *the regulatory, legislative and political framework will provide appropriate protection of soil as an irreplaceable natural resource and empower and encourage people with soil to manage it properly;*
- *a better understanding of, and access to, information on the state of our soils and the physical, chemical and biological processes which operate on and within them.*

In addition to the Soil Strategy, the Environment Agency (2004) produced a report on the State of Soils in England and Wales, with the view to promote schemes and implement strategies that bring about better soil management and more sustainable farming practices. To achieve these goals the EA have set targets for land and soil.

Soil functions

Within the Soil Action Plan for England the focus is on soil function. In each case the defined soil function is the result of the interaction and/or integration of a number of soil processes and in many cases the same processes may be linked to a number of functions. The definition of soil functions is anthropocentric, and recognises the importance of soil processes in providing the biophysical necessities for human life and/or making other contributions towards human welfare. Definition of soil functions is closely linked to the allied concept of the role of ecosystems in providing services that are of value to society. Even where they are mainly used for one purpose, soils have the potential to deliver a range of ecosystem services (Box 1.2). Some soil functions can work together so that benefits arising from a particular soil protection measure may extend beyond the original aim. For example, the conservation of peat-lands for their biodiversity and carbon storage interest will also protect their value to the historic environment and vice versa. The Soil Action Plan for England has defined six key soil functions:

- Food and fibre production;
- environmental interaction (between soils, air and water);
- support of ecological habitats and biodiversity;
- protection of cultural heritage;
- providing a platform for construction;
- providing raw materials.

Box 1.2 Ecosystem services of soil used for agricultural production.

After the production of crop and/or animal biomass soil functions can be described as:

- degrading xenobiotics (synthetic/foreign chemicals) used in crop and animal production;
- a sink/source for nutrients;
- a source of food for birds and mammals;
- a sink/source for carbon;
- a sink/source of water (flood defence/ water resources);
- degrading organic wastes and associated contaminants;
- a sink/source for inorganic contaminants, trace elements;
- a sink/source for atmospheric pollution and greenhouse gases;
- a habitat for plants and animals;
- a sink/source for sediments.

Adapted from report of R&D project P5-053 PR/02 (Loveland and Thompson 2002)

For the purpose of this report the three most important functions are considered to be:

Food and fibre production

Food and fibre crops require soils to be maintained in a suitable state that provides good soil structure, water retention and nutrient availability. Inappropriate soil management by land managers can lead to soil erosion, a loss in soil organic matter and/or degradation of soil structure. These changes can result in a decline of productive capacity through loss of water holding capacity, loss of machinery days and loss of nutrients.

Environmental interaction (between soils, air and water)

Soil provides the essential link between the components that make up our environment. These components include the atmosphere, surface and ground waters, above-ground habitats, and human activities. Managing and protecting soil is therefore an essential part of protecting the environment as a whole. Soil forms these links through:

- the exchange of gases, such as carbon dioxide, with the atmosphere;
- its role in regulating the flow of water and rainfall in the water cycle;
- its role in the degradation and storage of organic matter;
- the storing, degradation and transforming of solid materials, such as nutrients, organic materials and contaminants that are applied through animal and human activities or deposited by flood waters and aerial deposition;
- protecting the capacity of soils to store, transform and regulate these processes is critical to environmental sustainability.

Support of ecological habitats and biodiversity
Soils contain a very diverse biota and soil biodiversity is vitally important in maintaining soil functions and sustainable systems as many of the key processes underpinning these functions are mediated by the soil biota. Fungi, bacteria and larger organisms, particularly earthworms, play a crucial role in the generation and stabilisation of soil structure that influences rooting, aeration and drainage.

1.4. Aims and objectives

The project reviewed existing literature to:

- detail the direct and indirect functions of below-ground biological activity and link this where possible, to species richness and other measures of below-ground biodiversity;
- draw from a wide base of literature to assess the evidence for the direct and indirect impacts of land management practices on species diversity and function in soil;
- evaluate the implications for below-ground biodiversity and ecosystem function of modifications to land management approaches and farming systems, particularly the implications of organic agriculture.

The scope of the review has included the following agricultural systems:

- livestock based systems: extensive pasture upland, (predominately beef and sheep production); intensive lowland pasture (predominately dairy systems);
- cropping-based systems: ley-arable mixed farms; predominately arable or stockless systems; field -scale vegetable systems.

The review has not covered the possible impacts of orchards or woodland/forestry-based systems.

1.5. Methodology

There were five stages to the review:

- an initial scoping study using the review of Hole *et al.* (2005) as a starting point;
- a literature review that in the first instance concentrated on identifying review papers. From this the direct and indirect functions of below-ground biological activity were identified and linked, where possible, to species richness and overall below-ground biodiversity;
- working with the expert group, key research groups were identified this included researchers who were involved in the Natural Environment Research Council (NERC) thematic programme on Soil Biodiversity;
- a series of one-to-one meetings with key researchers with particular expertise in soil biodiversity and the function of the below-ground biomass were held. Through the use

of structured interview techniques the literature review gained access to relevant information contained within 'grey-literature';

- the draft report was circulated for comment to stakeholders and discussed at a workshop held in March 2006.

2. Below-ground ecology

2.1. Soil organisms

Soil organisms not only occupy soil; they are a living part of it and as a result of their interacting activities also change it (Killham 1994). Soils contain a very high diversity of organisms; many of which remain unknown or, at least, little studied (Brussaard *et al.* 1997). In the following sections the characteristics and "roles" of the main soil organisms in relation to soil processes are described.

2.1.1. Bacteria, archaea and actinomycetes

Bacteria are single-celled prokaryotes. The large majority of bacteria existing in soil (> 95%) are not culturable usig conventional techniques and so for a long time could not be studied. Much of our current understanding of the roles of bacteria in soil (Box 2.1) therefore derives from studies of culturable bacteria and/or approaches which treat bacterial communities in soil as a single unit –often known as the "black box" approach. (Stockdale and Brookes 2006). Archaea, which are a distinct prokaryotic kingdom, are much less numerous than bacteria but account for around 1% of prokaryotic activity in soil (Buckley *et al.* 1998) but may contain important functional groups responsible for methanogenesis and even ammonia oxidation (Nicol *et al.* 2003). Actinomycetes are predominantly decomposers of a range of complex organic compounds in soil and make up a small proportion of the soil community. However, actinomycete populations have been studied relatively intensively as they are widely know as sources of exude antibiotics that they produce as secondary metabolites.

Box 2.1. Roles of Bacteria.

- *Free-living* Decomposition and mineralisation of organic compounds (including agrochemicals and xenobiotics); synthesis of organic compounds (humus, antibiotics, gums); immobilisation of nutrients; mutualistic intestinal interactions; resource for grazing animals; formation of biofilms; pathogens of plants; parasites and pathogens of soil animals; helpers in mycorrhizal associations, food resource for grazing microfauna. Some specialists identified by their particular role in soil processes e.g. methanotrophs, methylotrophs, methanogens, butyrate oxidisers, nitrifiers, denitrifiers, sulphur oxidisers, sulphate reducers, and many more.
- *Symbionts* As associative N_2-fixers with legumes, N fixing shrubs and trees.

Bacterial cells lack a nuclear membrane and reproduce largely by binary fission. Consequently they are able to adapt rapidly. This plasticity and capacity for change are a very important characteristic of bacterial populations. The soil bacterial population is also able to 'slow down' metabolic activity and grow, even under conditions of very low energy and nutrient availability. At any time a high proportion of the population may be dormant.

Quantifying the diversity of soil bacteria is consequently problematic; there is currently no satisfactory conceptual framework for defining taxonomic units in prokaryotes (Fitter 2005). Using non-culture based molecular methods which extract DNA from soil and analysis of the DNA fragments, the vast diversity of bacterial ribotypes (i.e. subtypes of bacterial strains identified by their ribosomal nucleic acid composition) is increasingly being revealed. Molecular approaches quantify diversity based on assumptions about the degree of nucleic acid sequence similarity that can be used to differentiate taxa (Fitter, 2005). In arable ecosystems, proteobacteria have been shown to dominate the ribotypes determined with over 30 percent of the ribotypes attributed to this bacterial group (Sun *et al.* 2004; Smit *et al.* 2001). Acidobacteria are also very common (Sun *et al.* 2004; Smit *et al.* 2001), this is not unexpected as this group have been found in very wide range of environments (Barns *et al.* 1999). As might be expected, different patterns of dominance are beginning to emerge from studies of grassland soils (e.g. Bornemann *et al.* 1996). Nicol *et al.* (2003) have shown that upland grassland archaeal communities are dominated by Crenarchaeota with some evidence that management practices influence the nature of the crenarchaeotal community. However, a considerable proportion of the bacterial community in soils remains unidentified (Sun *et al.* 2004).

2.1.2. Fungi

Fungi are eukaryotic and have a mycelial morphology with a mass of hyphal tubes enclosing multi-nucleated cytoplasm. Fungal hyphae are usually 2-10 µm in diameter, but can extend to m or even km in length. Fungi are involved in a large number of interactions and processes in soil (Box 2.2) and are part of many complex relationships with other soil organisms. Mutualistic relationships between arbuscular mycorrhizal fungi (AM fungi) and crops are widespread with only a very few crop species not forming such associations (Brassicae and Chenopodiaceae). The benefits of the associations to crops are well documented (see reviews of Harrier and Watson 2004; Leake *et al.* 2004; Gosling *et al.* 2006).

Box 2.2. Roles of Fungi

- *Free living* Decomposition and mineralisation of organic compounds (including agrochemicals and xenobiotics); synthesis of organic compounds (humus, antibiotics, gums); immobilisation of nutrients; mutualistic and commensual associations; resource for grazing microfauna; parasites of nematodes and some insects; soil aggregation.
- *Symbiosis* Mycorrhical species mediate the transport of water and ions from soil to plant roots; mediation of plant /plant exchanges of C and nutrients; regulation of water and ion movement through plants; regulation of photosynthetic rate; regulation of C allocation below-ground; protection from root disease and root herbivores; resource for grazing microfauna.

Fungal classification has classically proceeded on the basis of whether hyphae show internal division (septate or aseptate) and by characteristic reproductive structures. The physiology of fungi means there are limitations to the study of diversity (as classically applied) and, computation of ecological indices needs robust taxonomic approaches. AM fungi have a wide variety of genotypes, which are not necessarily separated neatly into functional or morphological units – individual species are almost impossible to define in the hyphal stage, hence they are identified usually by their spores. Fungal biomass in soils has been estimated from the extraction of ergosterol since ergosterol is an important cell wall component of most fungi and is produced almost exclusively by fungi (Stahl and Parkin 1996). The development of molecular methods for the identification and quantification of fungi has lagged somewhat behind that for bacteria (van Elsas *et al.* 2000). However, it is likely that that application of these methods will rapidly increase our understanding of the diversity and ecology of soil fungal communities (Anderson and Cairney 2004).

2.1.3. Protozoa

Protozoa are unicellular organisms are predominantly predators of the microbial (mainly bacterial) populations in soil (Box 2.3). Three classes of protozoa are distinguished in soil: flagellates, amoebae and ciliates. They are all aquatic organisms occupying water-filled pores and water films in soil but are capable of encystment to enable survival in low moisture conditions. Changes in species balance and biomass within protozoan populations have been related to soil conditions and the impact of soil management practices (Foissner 1997).

Box 2.3. Roles of Protozoa

- grazers of bacteria and fungi;
- disperse bacteria and fungi;
- enhance nutrient availability;
- prey for nematodes and meso/micro fauna;
- vector of bacterial pathogens;
- parasites of higher-level organisms.

2.1.4. Nematodes

Nematodes are microscopic roundworms with a diameter of < 50 µm which occupy water-filed pores and water films. Nematodes occupy central and diverse trophic positions within the soil food web (Box 2.4.), with at least three different functional groups identifiable: i) plant feeding/root herbivore species are primary consumers; ii) bacterial and fungal feeding nematodes, which are secondary consumers; iii) predatory and omnivorous species (tertiary consumers) are also common. Species balance can therefore indicate changes in below-ground ecological relationships (Bongers and Bongers 1998; Mulder *et al.*, 2003). In agricultural systems bacterial feeding species often dominate with about 15 times more bacterial feeders in agricultural grasslands in the Netherlands than fungal feeders (Mulder *et al.*, 2005).

Box 2.4. Roles of Nematode

- grazers of bacteria and fungi;
- disperse bacteria and fungi;
- enhance nutrient availability;
- root herbivores;
- plant parasites;
- parasites and predators of micro-organisms, meso-organisms and insects;
- prey for meso- and macro-fauna.

2.1.5. Mites

The smallest arthropods in soil (usually less than 1 mm), mites are generally characterized by the presence of a gnathosome (specialization of the head), absence of obvious body segmentation, and a six-legged developmental stage. Mites are also the most diverse group of arthropods in soil and therefore show a very wide range of feeding habits and life-history strategies (Box 2.5). Prostigmatid and oribatid mites have been relatively well studied in agricultural soils (Crossley *et al.* 1992; Siepel 1995). On average the presence of microarthropods increase decomposition rates across a range of environments (Seastedt 1984).

Box 2.5. Role of Mites

- grazers of bacteria and fungi;
- consumption and comminution of plant litter and animal carcases;
- predators of nematodes and insects;
- root herbivores;
- disperse bacteria and fungi;
- host for range of parasites;
- disperse parasites, especially nematodes;
- parasites and parasitoids of insects and other arthopods;
- prey for macrofauna;
- modify soil structure at micro-scales.

2.1.6. Collembola

Collembola, also known as springtails, are small (less than 6 mm in length) wingless insects in the subclass Apterygota. Different collembola species are specialised for different microhabitats in soil and litter and are quite susceptible to desiccation unless they remain in a moist environment. Collembola are food generalists; their diets are composed of a mix of detritus, algae, bacteria and fungi and vary with season. Hence collembola have a central role in soil food webs and affect decomposition processes (Box 2.6).

Box 2.6. Roles of Collembola.

- grazing of microorganisms and microfauna, especially in the rhizosphere;
- consumption and comminution of plant litter and animal carcases;
- micropredators of nematodes and other insects;
- disperse bacteria and fungi;
- host for range of parasites;
- disperse parasites, especially nematodes;
- prey for macrofauna;
- modify soil structure at micro-scales by production of faecal pellets.

2.1.7. Enchytraeids

Enchytraeids are related to earthworms (class Oligochaeta) and are important members of the mesofauna in many soil ecosystems (Hansson 1990). Morphologically, they look like small, white or transparent earthworms. Functionally, they are detritivores and microbial feeders and are therefore an important component of the decomposition system in soils (Box 2.7). There is some evidence that specific soil types are inhabited by specific enchytraeid communities and that these respond to changes in management (Didden 1993).

Box 2.7. Roles of enchytraeids

- comminution of plant litter;
- grazing and dispersal of micro-organisms;
- create pores for movement;
- mix soil particles and organic matter.

2.1.8. Earthworms

Earthworms show differences between species in both burrowing and feeding activities. Earthworm species are most commonly grouped into;

- epigeic species feed and inhabit the litter layer. Very few species in this group are found in UK agricultural systems;
- anecic species, which feed on fresh organic material pulled down from the litter layer and form deep and permanent burrows. These species play a crucial role in initiating the contact between inorganic and organic components in the soil. Anecic species make up about 70% of species present in UK agricultural systems. *Lumbricus terrestris* can reach population sizes up to a biomass of 2.5 t ha^{-1} in grassland systems (Killham 1994);
- endogeic species live and feed on OM from within the soil.

Earthworms have a very important role in the decomposition of organic matter in soils mainly as a result of the mixing of organic and mineral components and the incorporation of litter into deeper soil layers (Wolters 2000; Lavelle *et al.* 2001; Box 2.8). The structure of the earthworm community, as well as their abundance and biomass, has been suggested as an ecological soil quality indicator and these measures have been shown to indicate the influence of different anthropogenic land uses (Rombke *et al.* 2005).

Box 2.8. Roles of earthworms

- create pores in soil for movement;
- mix soil particles and organic matter;
- enhance microbial growth in gut;
- disperse microorganisms and algae;
- host to protozoan and other parasites.

2.1.9. Insects and other arthropods

Soil-dwelling species are common amongst the 29 insect orders. In many case insect species simply use the soil for the egg or pupal stages of the life cycle. Larvae of beetles, flies and ants are common; in addition woodlice, centipedes and millipedes are found in all life stages in soil. A number of these species are root herbivores and thus affect a range of above ground plant processes (Wardle 2002). It has also been shown that root herbivores have a critical role in facilitating the rapid interchange of fixed N between legumes and associated species e.g. in a mixed species sward (Murray and Hatch 1994). Hence insect species have a number of roles in soil processes (Box 2.9).

Box 2.9. Roles of soil dwelling arthropods

- consumption and comminution of plant and animal matter;
- root herbivory modifying plant performance above and below-ground;
- grazing of microorganisms and microfauna;
- especially in the rhizosphere;
- dispersal of microorganisms;
- predators of other soil organisms.

2.1.10. Functional grouping of soil organisms

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 organisms in functional groups. A common grouping is according to organism size (Figure 2.1). Grouping in this way has been shown to be meaningful as it allows a consideration of soil organisms in relation to the pore space within soils (see Section 2.2).

The wealth of information on the soil biota has also been integrated by grouping species into trophic categories. Using this approach the food web in soils can be described (e.g. Beare *et al.* 1992; Bloem *et al.* 1994; Figure 2.2) and modelled (Hunt *et al.* 1987; de Ruiter *et al.* 1993). The use of such models has allowed the relative importance of the interactions between trophic groups to decomposition and other aspects of nutrient cycling to be investigated. Brussaard *et al.* (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 (Section 2.2); Brussaard (1998) outlined a number of further problems with this type of modelling approach.

Bacteria and archaea, including free-living and symbiotic "species" Fungi including non-mycorrhizal and mycorrhizal species	Microorganisms
Protozoa Nematodes	Microfauna < 200 μm in diameter
Mites Collembola Enchytraeids	Mesofauna 100 μm – 2 mm in diameter
Earthworms Insects and other arthropods	Macrofauna >2mm in diameter

Figure 2.1 Size grouping of soil organisms.

Other criteria such as life-history tactics, microhabitat, feeding mode and other aspects of physiology can also be used to group species into functional groups i.e. a group of species that is similar with respect to their impacts on community or ecosystem process (Wall *et al.*

2004). Functional groups might also be defined with respect to the similarity of response of the species group to a given environmental change. Any functional group defined will include a variable number of species resulting in single species to relatively species-rich functional groups defined in soils (Brussaard 1998). Ekschmitt and Griffiths (1998) highlight the fact that definitions of functional groups can rapidly lead to a "very complex parameter space of high dimensionality"; consequently simpler and more aggregated classifications need to be constructed. The number of functional groups that are designated and the criteria used to establish the groups is therefore a function of the questions asked by a study (Swift *et al.* 2004). While there is unlikely to be a single classification developed that will suit all purposes, some groupings have found wide acceptance e.g. decomposers, ecosystem engineers – organisms that change the structure of soil by burrowing, transport of soil particles and hence create micro-habitats for other soil organisms (Jones *et al.* 1994). The description of functional groups allows the development of hybrid modelling approaches that can allow organism-oriented mechanistic representation of a limited number of interactions within a process-oriented approach (e.g. Schröter *et al.* 2004).

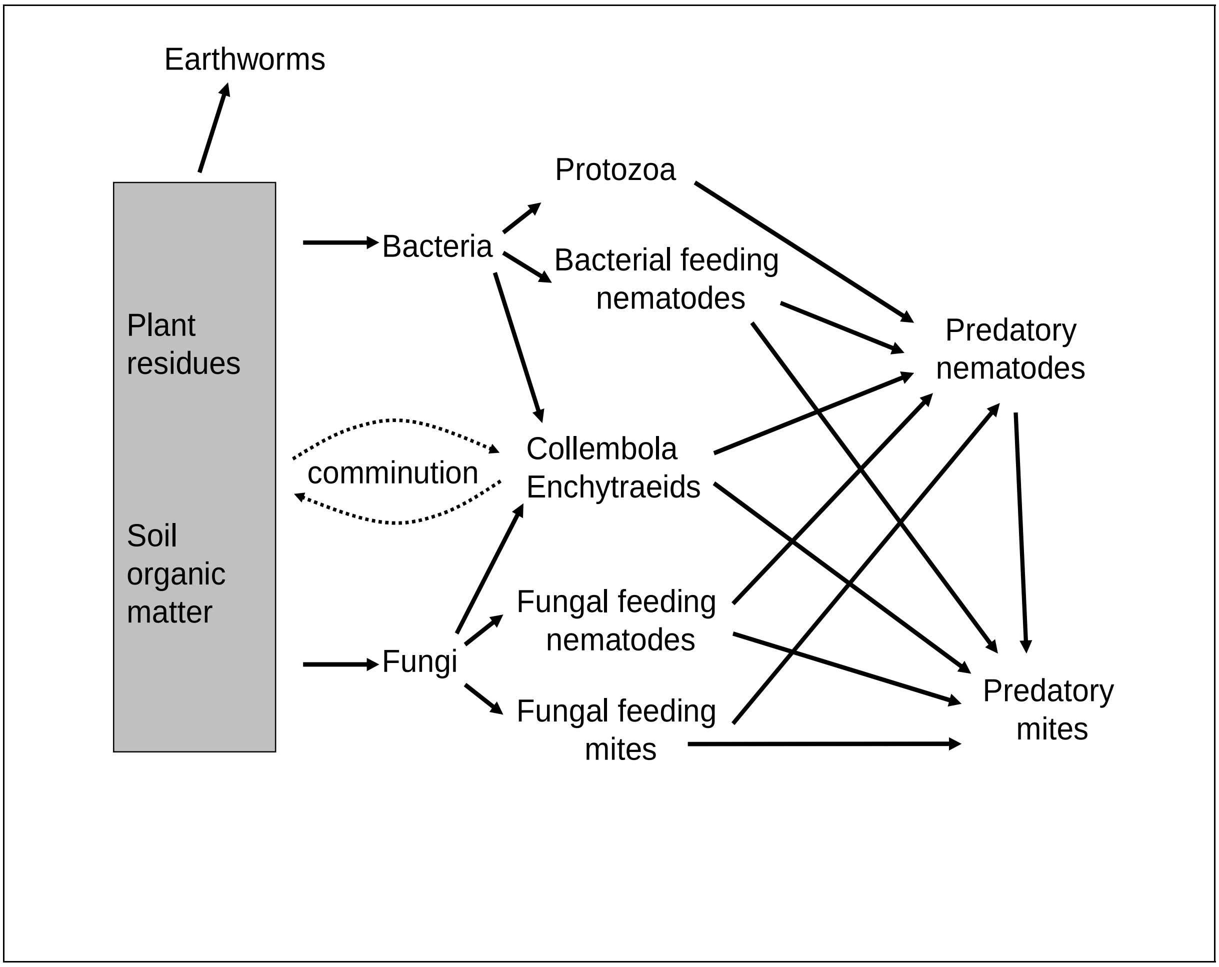


Figure 2.2 Decomposition of organic matter shown in relation to the taxa of the soil food web described in Section 2.1. Taxa are sub-divided into trophic groups where relevant. Returns to the pool of soil organic matter in excreta and/or on the death of organisms are not shown.

2.2 Soil as habitats

Anderson (1975) put into words the paradox that the diversity of below-ground species poses for ecologists. How is it possible for such a large number of species to apparently co-exist without biotic mechanisms (e.g. competitive exclusion) reducing diversity? The usual explanation given is the extreme spatial (vertical and horizontal) and temporal heterogeneity in soil which gives rise to a wide range of surface types, pore sizes and microclimates, and a range of resources and resource partitioning in space and time. Soil is an opaque medium with a complex physical structure and spatially diverse and temporally dynamic chemistry. 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 and Wardle 2002). Soil organisms are also often entering inactive or dormant states in unfavourable conditions, so that diversity is preserved; this is analogous to the role of soil seed-banks in preserving plant diversity (Ettema and Wardle 2002). The following section will describe soil properties with particular reference to the key characteristics of soil as a habitat for the organisms described in the previous section.

Soils form as a result of the physical and chemical alteration (weathering) of parent materials (solid rocks and drift deposits). Biological activity is a critical component of soil formation; soil is distinguished from weathered rocks as a result of the biological cycles of growth and decay leading to the incorporation of organic matter (OM). The vertical differentiation of soil through time as a result of the interactions of biological activity and water movement leads to the development of characteristic horizons, layers approximately parallel to the surface. Where soils are relatively undisturbed by man, the soil surface is often characterised by a layer of plant litter with organic matter is incorporated into lower mineral horizons through the activity of decomposers and ecosystem engineers; OM content usually declines rapidly down the profile. Soil profiles are the result of a series of complex interactions between climate, geology, topography and biological activity and where these factors are stable over a long period very distinctive horizons can form e.g. podsols. Under agricultural management soils are not in an equilibrium situation, they are regulated by a series of human managed perturbations and consequently represent something closer to a plagio-climax or pulse stabilised system (Odum 1969); soils under agricultural management are most often distinguished from adjacent soils in semi-natural systems by the character of their well mixed surface horizons.

Underlying geology shows a high degree of spatial variability in the UK and geology has a strong influence on topography, as well as directly on soil formation. Soil parent material controls a range of intrinsic soil properties including soil depth, stoniness, mineralogy and texture. Soil texture, that is the relative proportions of sand, silt and clay minerals in soil, plays a large part in determining the physical and chemical properties of soil. Farmers recognise soil textures in relation to their ease of cultivation and management, but these emergent properties also result from clay mineralogy and biologically-mediated interactions between soil minerals and OM. The principal minerals found in the soil also have a large control over inherent soil fertility and buffering capacity to acidity, as they contain varying amounts of basic cations (Ca^{2+}, K^{+}, Mg^{2+}, Na^{+}). Negatively charged sites on the surface of clay minerals and organic matter (cation exchange capacity) in soil also act as a nutrient sink for cations, restricting their loss by leaching from soil.

The architecture of the soil pore network largely describes the habitat space in soil (Young and Ritz 2000). It 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. The amount and nature of the pore space in soil is dependent not only on soil texture but also on the aggregation of mineral particles and soil OM i.e. the formation and stabilisation of soil structure. 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 > 50 µm in diameter and in topsoil are usually filled with air; storage pores 5-50 µm in diameter are the main pores which fluctuate in air/water balance whereas residual pores 0.5-5 µm are commonly full of water, though plant roots can effectively empty pores down to 0.2 µm. Roots use pores of > 100 µm as points of entry, root hairs, protozoa and fungi use pores of > 10 µm, while bacteria can move in water films of only 1 µm depth. Soil structure influences the nature and activity of soil organisms (Young and Ritz 2000), but soil organisms also have a key role in its formation and stabilisation (Figure 2.3; Tisdall and Oades 1982; Beare *et al.* 1995; Wolters 2000; Lavelle 2000). Plant roots also have a central role in structure development (Angers and Caron 1998). Ecosystem engineers change the structure of soil by burrowing, transport of soil particles and hence create micro-habitats for other soil organisms (Jones *et al.* 1994). In temperate agro-ecosystems, earthworms are very dominant within this functional group. For example it has been estimated that 90% of the soil OM in an upland improved grassland soil had been processed by earthworms and enchytraieds (Davidson *et al.* 2002).

The physical environment can be considered as a template on which organisms and ecological systems operate; but it is clear that organisms' response to the physical environment may exhibit patterns that vary between species and are constrained by the geometry of the environment (Williams *et al.* 2002). Ettema and Wardle (2002) review a range of evidence showing spatial patterning in the distributions of soil organisms over sales of mm to hundreds of metres, and show the importance of both environmental controls and interactions between organisms. Franklin and Mills (2003) demonstrated that bacterial distributions can be highly structured, even within a habitat that appears relatively homogeneous at the plot and field scale. Different subsets of the microbial community were distributed differently, and this is thought to be due to the variable response of individual populations to spatial heterogeneity associated with soil properties. Rossi *et al.* (1997) found that the patchy distribution of earthworms in pasture was linked to the difference in dispersal between juveniles and adults. Whalen (2004) found that earthworms show temporally persistent spatial patterns in forest soils, but not under agricultural management, and linked this to patterns of resource availability as well as the impact of disturbance.

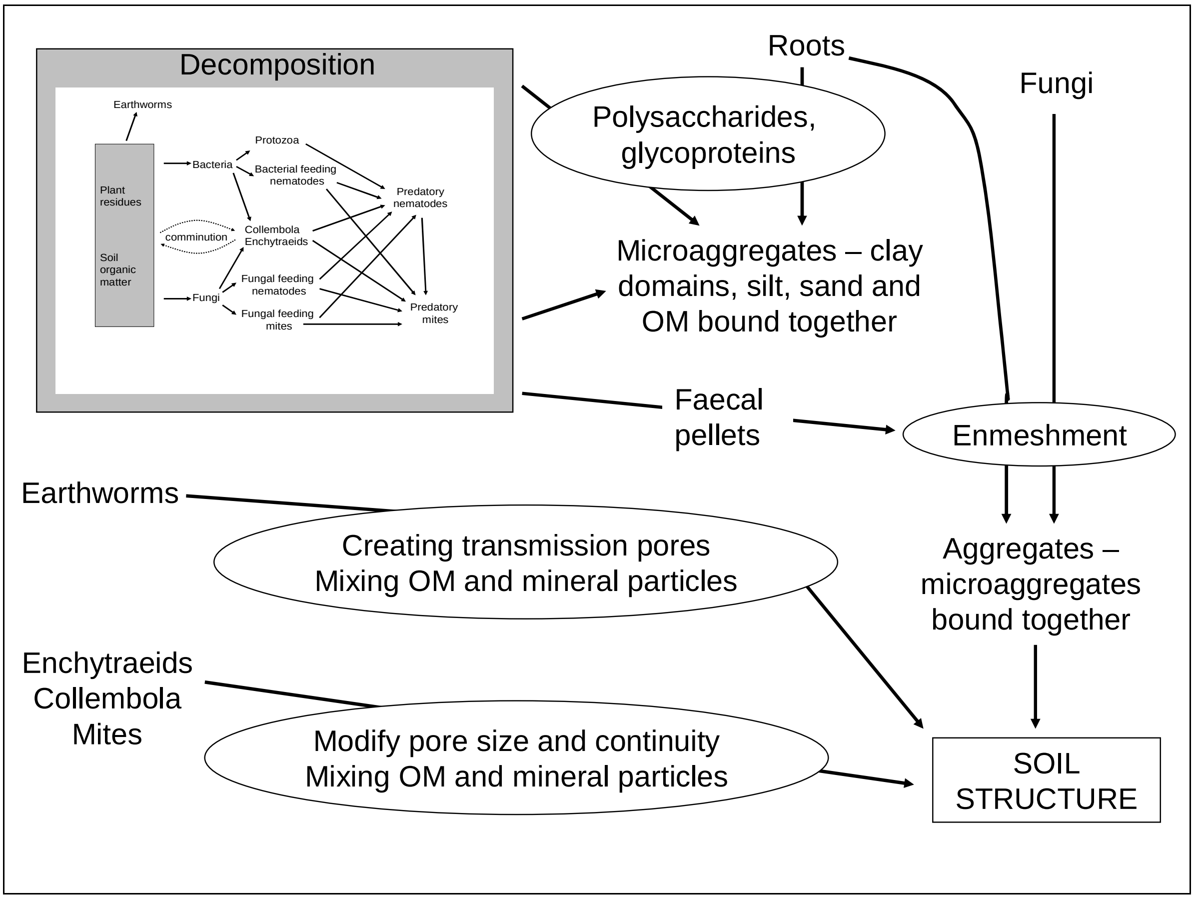


Figure 2.3 Soil structure (development and stabilisation processes) shown in relation to the roles of soil organisms.

Soil OM consisting of plant, animal and microbial residues in various stages of decay is the main food resource for soil organisms. Van Nordwijk *et al.* (1993) observed increases in protozoa concentration associated with buried stubble and other old organic matter. Buried litter and other concentrations of organic matter in soil have therefore been described as resource islands (Wardle 1995); in contrast the bulk soil is a relatively resource poor habitat. Across a range of climates and systems Wardle (1992) showed a strong correlation between total OM contents in soil and the size of the soil microbial biomass population. Lynch *et al.* (2004) cite two studies which suggest that there is a critical level of SOM for microbial functional diversity in soil (1.7% OM).

Cell numbers and the amount of bacterial biomass in soil have been shown to be dominantly associated with the smaller particle size-fractions in soil (clay and silt) and hence pores of 5-30 µm (Amato and Ladd 1992; Hassink *et al.* 1993; Kirchmann and Gerzabek 1996). It has also been shown that denitrifier activity differs amongst aggregates of different sizes (Seech and Beauchamp 1988); numbers of Rhizobium also were affected by aggregate size (Mendes and Bottomley 1998). Sessitsch *et al.* (2001) showed that the bacterial community associated with clay and silt fractions was larger and more diverse than that associated with the sand-fraction; it was suggested that bacteria found in larger pores operate in a relatively nutrient poor environment and are more strongly affected by predation. Tiedje *et al.* (2001) suggested that spatial isolation limits competition between microbial species and hence leads to more diverse microbial communities; this is related directly to soil moisture content and pore connectivity. In a study in cultivated silt loam soils Schutter and Dick (2002) found no strong influence of aggregate size on the microbial community structure. However, there was a weak association between increased fungal biomass and larger aggregates (Schutter and Dick 2002), consistent with the roles of hyphae in binding aggregates (Figure 2.3).

Many predators of bacteria (e.g. protozoa and nematodes) are sensitive to changes in soil water content as they are dependent on the presence and continuity of water films. As water content decreases the frequency of encounter of predators and bacterial prey also reduces; pore size will also play a role. Pores with neck sizes below a certain diameter may restrict entry by some organisms and hence protect smaller organisms from predation (Young *et al.* 1994); substrate may also be protected from decomposition by similar mechanisms (Powlson 1980). Nevertheless extracellular enzymes can lead to apparent biological activity in smaller pores than organisms are able to inhabit (Young and Ritz 2000). In a field soils, nematodes were shown to be dominantly associated with the pore class size of 30-90 μm, while bacterial biomass was correlated with pores 0.2-1.2 μm (Hassink *et al.* 1993). Habitable pore space has provided an important framework for consideration of organismal interaction in soil; however, it this is not a model capable of robust application since pore size distributions are usually calculated from moisture release curves which assuming simple cylindrical pores and rigid body sizes of soil organisms (Young and Ritz 2000). Fractals have been shown to be a potentially powerful tool for the quantification of the relationship between soil structure and the dynamics of soil organisms. Fractal geometry allows the quantification of scale dependence in ways that allow predictions in the face of ambiguity. (Williams *et al.* 2002). Young and Crawford (2001) showed that fractal geometry was a potentially powerful method to relate functional characteristics of the soil habitat and protozoan dynamics.

Plant root systems are a dynamic and varied component of below-ground ecology. Plant roots are themselves a key habitat component for a number of soil organisms including symbiotic bacteria, mycorhizal fungi and root pathogens and herbivores (Brussaard 1998). Lupwayi *et al.* (2004) showed that distinct populations of culturable endophytic bacteria were associated within the plant root for a range of crop plants (barley, wheat, oilseed rape); Normander & Prosser (2000) also showed discrete endophytic bacterial community compositions using carefully controlled molecular methods. The composition of the community of root (endophytic) bacteria is determined by a combination of both plant and soil specific factors – endophytic bacterial communities show similarity to the soil community (Germida *et al.* 1998; Seghers *et al.* 2004) but root communities had much lower bacterial species richness than those associated with the root surface (rhizoplane) or soil associated with roots (rhizosphere).

The rhizosphere is now generally defined as the zone of soil surrounding the root within which the soil is directly influenced by the presence of the root (Killham 1994). The rhizosphere is a dynamic zone of soil and several stages of rhizosphere development have been recognised (Jones *et al.* 2004):

- arrival of root usually entering large macro pores. Exudation of soluble and insoluble mucilage is large compared to uptake rates of C by root;
- developing rhizosphere Root hairs fully expanded. Mycorrhizal infection complete. Rate of C exudation low, larger influx of C to rot;

- mature rhizosphere – root hair, epidermal and cortical cell death occurring. Fully developed mycorrhizal network. Exudation large where cell lysis is occurring. Some influx of C;
- dying rhizosphere. Mycorrhizas lost. Cortical loss. Root becomes OM input to soil. Eventually becomes relict with leaving a macropore in the place of the root.

Very rapid and dynamic transfers of C below ground are mediated by the root into the rhizosphere. Studies in upland grassland in the UK showed very rapid transfer of pulse-labelled photosynthetically fixed C by plants to below-ground organisms (within hours). Labelled C was recovered from only 10% of roots (Bruneau *et al*. 2002), indicating a range of root demand for C consistent with the rhizosphere development model presented above. About 5-8% of C fixed by plants rapidly appeared in AM fungi and c. 2% in bacteria (Ostle *et al.* 2003).

Around 4-10% of the root surface has been shown to be covered by soil microbes. Increased bacterial populations for both the rhizosphere and rhizoplane compared to the total soil is well known; fungal populations tend to be reduced but with a relative increase in plant pathogens (Killham 1994). Distinct populations of bacteria have been observed in the rhizosphere and rhizoplane for a range of plants (Normander and Prosser 2000; Lupwayi *et al.* 2004). The community composition in the rhizoplane was more closely related to that of the soil than to root communities or those determined on the seed coat suggesting that the population on the root surface was dominantly derived from a pre-existing soil population (Normander and Prosser 2000). Marschner *et al.* (2001) showed that the rhizosphere community which developed was the result of complex interactions between plant and soil factors together with a consideration of the position along the root; a lower number of bacteria were observed at the root tip than associated with the mature root zone. Increases in bacterial grazers (protozoa, nematodes) are also seen in the rhizosphere; the increase is higher near dead than living roots (Griffiths 1994). Lussenhop (1992) observed microarthropod densities at least twice as high in rhizosphere as bulk soil.

It is clearly possible to conceptualise soil as a series of linked habitats, rather than a single habitat for soil organisms (Box 2.10). Such a conceptualisation has been shown to provide a useful representation of soil faunal populations. Following a comprehensive 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. A number of conceptual approaches have been taken to define/divide soil into a series of distinct habitats.

- Lavelle *et al.* (1993) presented a general model in which the dynamics of decomposition in terrestrial ecosystems are determined by a set of hierarchically organized factors which regulate 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 macroorganisms. Biological systems of regulation based on mutualistic relationships between macro- and microorganisms ultimately determine the rates and pathways of decomposition. For tropical environments four such systems were defined:

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

- Beare *et al.* (1995) considered soils to be composed of a number of biologically relevant spheres of influence that define much of their spatial and temporal heterogeneity. Although not mutually exclusive, each sphere has fairly distinct properties that regulate the interactions among organisms and processes that they mediate. In addition to the systems defined by Lavelle *et al.* (1993), Beare *et al.* (1995) added the porosphere and aggregatusphere which focus on soil organisms and their interaction within soil pore space and aggregates respectively.
- Brussaard (1998) identified three “guilds” of organisms in soil (linked in particular to decomposition) which were characterised based on characteristics of the physical environment, resource quality and the roles taken by soil organisms:

  - organisms living in association with the living plant (N fixer, mycorrhiza, diseases and pests of root);
  - decomposers including comminuting species, primary decomposers and microbial feeders which regulate numbers and activity of microbes found in the rhizosphere, litter and in bulk soil ;
  - ecosystem engineers - Meso- macro fun that create habitats for other organisms by affecting physical structure.

However soil habitats are defined, each is likely to have distinctive physical and chemical characteristics together with distinguishable communities of soil organisms. Soil habitats and the links between them will be perceived differently by different species due to differences in size and mobility (Giller *et al.* 1997).

Box 2.10 Habitat types in soil

Resources (places)

- Root;
- Root surface (rhizoplane);
- Rhizosphere;
- Organic mater (litter to old humus);
- Mineral surfaces.

Pores (spaces)

- Transmission (AIR filled);
- Storage (Air/water filled);
- Residual (WATER filled).

Soil temperature is strongly dependent on air temperature and shows a similar seasonal pattern. Temperature has a direct effect on the rates of biological reactions; plant root growth and the activity of soil organisms increase with increasing temperature. Soil animals are also sensitive to overheating. Temperature may also affect the proportion of the organic matter that is decomposable (Dalias *et al.*, 2003) and/or microbial efficiency (Henriksen and Breland 1999). In the UK there is also marked seasonal variation in topsoil moisture content due to the interaction between evapotranspiration and rainfall events. Soil water not only affects the growth and activity of soil organisms but also affects flows between habitats and organisms. During periods of plant growth, water is drawn to the root from relatively large distances in soil as a result of the gradient established by root demand and uptake. In temperate climates it is the interaction of temperature and moisture that largely control the rate of biological processes and hence N cycling (Nishio and Fujimoto 1989; Recous *et al.* 1999). Seasonal variation in plant growth also leads to temporal variability in organic matter inputs to soil. Spatial patterns and activity of soil organisms often show greater fluctuations than underlying patterns of soil resources (Ettema and Wardle 2002), though e.g. mineral N variability is highly dependent on interactions between microbial activity and water movement through soil. Temporal variability in the activity and biomass of below-ground organisms can therefore be as significant as spatial variability.

2.3. Below-ground ecology

In above ground ecology the factors affecting diversity (in theory) have been identified and ranked as trophic interactions between species, spatial habitat heterogeneity, temporal habitat heterogeneity, disturbance and nutrient resource availability (Torsvik *et al.* 2002). However, “very little supports the notion that these relationships above-ground can be simply transferred below ground” (Bardgett 2002). The extensive and critical review of Wardle (2002) highlights that competition is not the main regulator of trophic relationships below-ground. This is not to say that competition plays no role, for macro-fauna and some fungi there is evidence of some competitive regulation (Wardle 2002). Wardle (2002) concluded that within the decomposition interaction web (Figure 2.2) the fungi-based channel has been shown to be resource driven (bottom-up regulation) while the bacteria-based channel has been shown to be predator controlled (top-down regulation). Consideration of inter-organism interactions and their relation to function (Wardle and Giller 1996; Wardle 2002) can only take an understanding of ecological relations below ground so far; it is essential to also integrate spatial habitat factors (Young and Ritz 1998). The high degree of specialisation amongst soil animals provides evidence that increasing spatial heterogeneity increases soil animal diversity (Bardgett 2002). Spatial variability has been often treated as distracting "noise" which obscures the key relationships between structure and function of below-ground biodiversity, however, it is likely that understanding the control over ecological systems imposed by spatial variability is the key to improving our ability to manage below-ground ecosystems (Ettema and Wardle 2002). This may provide the "theory linking microbial population dynamic to biodiversity and function in terms of the soil microenvironment" which Young and Crawford (2004) conclude is more or less absent and presents a major interdisciplinary challenge for soil science.

Describing and modelling the interaction of below-ground ecology is often caught in the "middle number" conundrum i.e. 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 incomplete picture of what is going in (Weinberg 1975; Wu and David 2002). This problem is amplified by spatial and temporal variations and interdependencies, scale dependencies and thresholds. Fitter (2005) suggests that "the heterogeneity of soil means that meta-population ideas are necessary or possibly even meta-community or meta-ecosystem approaches". Wilson (1992) showed that large-scale ecological systems which are fragmented into a mosaic of patches, i.e. a meta-community can be successfully modelled. It is clear that all ecological processes occur in a spatial context.

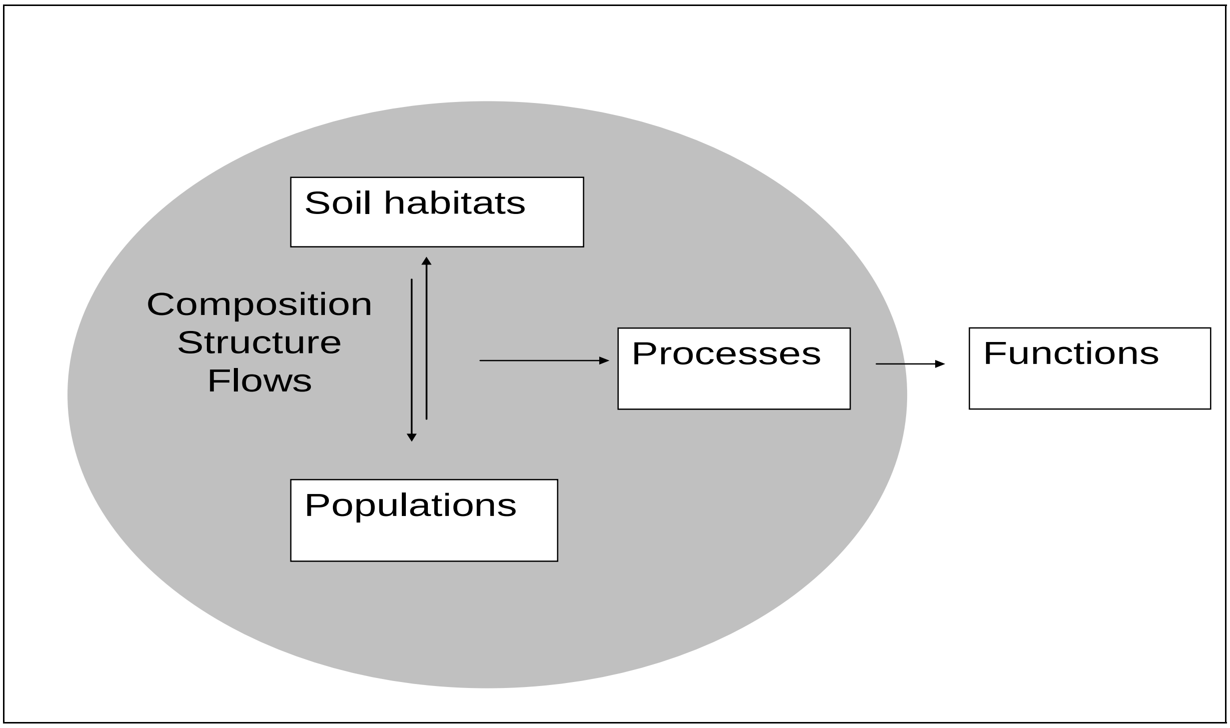
The development of landscape ecology has re-emphasized the importance of spatial patterns in constraining any ecological processes (Williams *et al.* 2002). On a landscape level, diversity may be viewed at different levels of resolution. Whittaker (1972) proposed to distinguish between diversity of species within a community of a habitat (α-diversity), rate and extent in change of species along a gradient of habitats (β-diversity) and richness of species over a range of habitats (γ-diversity). Landscapes are composed of multiple elements which create heterogeneity within an area. To define a landscape it must be possible to identify 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). 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). Pattern prediction is complex and multifactorial – interaction between access to resources and refuge from predators (Brown *et al.* 1995). It is the spatial relationships of these elements, as much as their diversity, that are key to affecting the interactions within the mosaic (Table 2.1). "Landscape ecology deals with the causes and consequences of the spatial composition and configuration of landscape mosaics" (Wiens 1992).

Table 2.1. Range of potential measures of landscape structure (adapted from Wiens 1992)

Patch measures	Mosaic 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
Distance (nearest neighbour, proximity)	Fractal dimension (edge, area)
Corridor characteristics (length, shape, linkage e.g. stream order)	Heterogeneity
	Gaps (lacunarity)
	Spatial correlation (semi-variance, distance decay, anisotropy)
	Connectedness (network, lattice properties)

The hierarchy of diversity, which is clearly plausible for traditional habitat diversity, might also be used to describe soil microbial diversity concepts (Lynch et al. 2004). It is already understood above ground that the scale at which these mechanisms are expressed and hence the scale of the landscape differs for different organisms (Wiens and Milne 1989); this will certainly be true below ground. While studies of boundaries have mostly taken place at landscape scales, Belnap *et al.* (2003) considered the interfaces across millimetres between soil and roots and between atmosphere and soil surface as boundaries through which a range of interactions occur in three dimensions, with time as a fourth dimension. They concluded that there are no fundamental differences between fine (mm) and coarser (km) scale boundaries other than units of measure and methods of study; small scale boundaries with the soil therefore could provide useful experimental systems which might be use to test hypotheses that are untestable at larger scales. However, application of landscape ecology approaches to below-ground ecology will not be easy. As already described the soil landscape is much more temporally variable than landscapes above-ground; ecosystem engineers and roots are constantly establishing and modifying connectivity and fragmentation in below ground systems. Plants have a series of roles in the below ground ecosystem; they are affected by below ground "guilds" but also affect them (Brussaard 1998). The plant or plant community integrates across the diversity of below-ground ecosystem functioning; in someway this role can be compared to that of the top predator in above ground systems.

This report will not attempt the application of a model based on landscape ecology concepts to the study of below-ground ecosystems. However, this is a research topic worthy of further exploration. Initially it is critical that the landscape elements below-ground can be defined and that they clearly differ in quality. The context and connectivity of these landscape elements are then key. Above-ground landscape ecology is moving away from a simple patch-matrix view of landscape and consequently connectivity should be considered as an aggregate property of the structural configuration of the landscape elements. We can consider that below-ground processes, which contribute to the delivery of the anthropocentrically defined functions, result from the interaction of soil habitats and their associated populations (Figure 2.4), where the structure, composition and flows between these components are critical in defining the outcome and rate of the processes observed at the soil scale.



Where:
Composition
- Which habitats, amount, quality, stability (characterised by patch measures).
- Structure
  - How habitats are arranged in space, boundaries, permeability, stability of arrangement (characterised by mosaic measures).
- Flows (= processes, as defined by Wiens, 1992)
  - How habitats are linked through time, movements of individuals, energy, water, nutrients.

Figure 2.4 Soil function presented as the outcome of processes in soil which result from the interaction of soil habitats and populations, forming metacommunities that are strongly influenced by the spatial context (described in terms of composition structure and flows as defined by Wiens (1992).

2.4. Below-ground biodiversity

The Convention of Biological Diversity defines its area of concern as:

"*the variability among living organisms from all sources, including, inter alia, terrestrial, marine and other aquatic systems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems*" (Heywood 1995).

The term "biodiversity", which is the widely used shortened form of biological diversity, is therefore used to refer to diversity at various levels:

- genetic. Within and between species diversity, identification of individual organisms from some unique part of their DNA or RNA.
- taxonomic. Diversity, density and occurrence of species groups, most commonly referred to as species richness. Taxonomic diversity can also be defined at higher taxonomic levels e.g. phyla, orders or families.
- ecosystem. Diversity of species assemblages and their environments.
- ecological/functional. Density and occurrence of ecological/functional groups. Differences between groups are expressed in terms of differences in body size behaviour, resource and habitat preferences etc. rather than taxa (Section 2.1.10). Several species might carry out the same processes leading to apparent functional redundancy; species might also interact leading to functions which are not performed by any individual species.

A key problem in many of the debates on the value of biodiversity is that the broad term "diversity" is not clearly distinguished from the specific attributes of the community of organisms under study. Often biodiversity is simply used to refer to the totality of species and within species variability in a particular system (Swift *et al.* 2004) – a definition that adds little to the use of the term "biomass". In any assessment of biodiversity, total abundance, species richness and dominance pattern are key components in an assessment of diversity; however, many diversity indices aggregate these three components into a single figure (Ekschmitt and Griffiths 1998). It is also clear from discussion in the preceding sections that the diversity of any system is not adequately captured simply by the number of species present, but the relationships between these species in space and time also need to be taken into account. Therefore, it is unlikely that any precise relationships between biodiversity and functioning can be drawn out for testing by experimental and/or modelling approaches. Hence studies of biodiversity in relation to ecosystem function often only examine aspects of the overall concept of biodiversity and limited numbers of functions.

In ecological theory developed from a consideration of above-ground species, four main hypotheses describe how biodiversity and ecosystem functioning relate:

- null hypothesis i.e. no effect of species richness on ecosystem function;

- optimum ecosystem function achieved at intermediate levels of diversity. Humpbacked relationships are often seen between stress / disturbance and species diversity for above ground relationships (Hooper *et al.* 2005);
- optimum ecosystem function maintained until low levels of diversity when ecosystem function lost rapidly (rivet hypothesis). Vitousek and Hooper (1993) proposed that in most cases that there was an asymptotic relationship between increasing no of species and rate/amount of any function, but that this effect is saturated at a relatively low number of species;
- ecosystem function changes with changing species richness but there is no predictable response.

The BIODEPTH project investigated manipulation of species richness through controlled combination of a varied combination of plant function groups (grasses, forbs and legumes). When the significant effects of geographic location were removed, increases in plant diversity (considered as both the inclusion of more functional groups and/or species) had generally positive effects on above-ground ecosystem processes (Spehn *et al.* 2005). As discussed above, this is an area of study where it is very difficult to design experiments in which sampling effects do not affect the results and from which broader scale conclusions can be drawn (Swift *et al.* 2004). There is no space in this report to review in detail the studies reporting on links between diversity and specific functions above ground. We would direct interested readers to the major literature review of Hooper *et al.* (2005) which highlights areas where scientific consensus was achieved in the complex area of effects of biodiversity on ecosystem functioning and also a number of areas where uncertainty remains.

Compared to the work carried out on diversity-function relationships for plant species, relatively few studies have been carried out below ground. There are a number of technical constraints in determining species, abundance and interactions below ground particularly for micro-scale organisms; consequently there are difficulties in applying the concept of species richness formally below ground (Ekschmitt and Griffiths 1998; Bengtsson 1998). It is often assumed that there is a high degree of redundancy of below-ground species, i.e. species are replaceable with other species without an influence on soil function (Groffman and Bohlen 1999). This implies that a range of organisms can perform the same function; but it is unclear to what extent differences in environmental tolerances, physiological requirements, microhabitat preferences exist between apparently functionally similar organisms. In their comprehensive review of empirical evidence to date, Mikola *et al.* (2002) showed that the relationships between soil biodiversity and ecosystem functions range from positive to neutral or even negative. The main biotic controls over ecosystem function result from species traits, changes in species composition and changes in the multi-trophic interactions that occur in soil (Bardgett 2002). Consequently it is unlikely that consistent diversity effects will be observed. Links between above and below ground ecosystems are numerous, complex and often determinant of ecosystem processes (Wardle 2002). In addition it is important that the variability of the process and species dynamics with space and time (Ekschmitt and Griffiths 1998) are determined alongside the mean rates of ecological processes and mean below-ground species abundance, dominance and species richness.

In agricultural systems, which are typically in dynamic non-equilibrium states, the question of whether diversity affects how process rates respond to disturbances is as important as whether there is a link between biodiversity and ecosystem function. A major below-ground ecosystem reconstruction experiment, summarised in Fitter *et al.* (2005) where soil-plant ecosystems were constructed with an increasing gradient of below-ground ecological complexity, showed differences in root growth, AM fungi colonisation and litter decomposition. However, no differences in total plant biomass or ecosystem CO_2 exchange were seen suggesting a high resilience of the below-ground ecosystem, where these broadly-based functions were maintained even where the biological structure had been altered very significantly. In contrast it would seem logical that where a function is dependent on only a few key species then stress and/or disturbance is more likely to affect this function than a more ubiquitous process such as decomposition (Bengtsson 1998). In such an instance it is possible that individual taxonomic species could be a more sensitive indicator of ecosystem disturbance than entire functional groups – however, except for certain macrofauna, the current state of systematic biology for most soil organisms makes definition at this resolution extremely difficult. In general there is agreement that individual species are probably not good indicators except where they have a “keystone”/fundamental functional role – this is particularly true for microbes and smaller invertebrates.

There is a consensus that “some minimum number of species is essential for ecosystem functioning under constant conditions and that a larger number of species is probably essential for maintaining the stability of ecosystem processes in changing environments” (Loreau *et al.* 2001). The response of below-ground ecosystems to disturbance is characterised by both resistance and resilience where resistance is described as the ability of the soil to withstand the immediate effects of perturbation, and resilience the ability of the soil to recover from perturbation (Griffiths *et al.* 2001). Some studies have shown that as biodiversity declined, decomposition becomes less stable to experimental perturbations i.e. it becomes both less resistant and less resilient (Griffiths *et al.* 2000). In contrast, Griffiths *et al.* (2004) concluded that stability of decomposition is related to specific components of the microbial community rather than diversity *per se*. The question set by Brussaard *et al.* (1997) is still valid: are a minimum number of functional groups and/or species within functional groups needed to maintain process rates following disturbance or is the presence/absence of certain species decisive alone?

2.5. Relationships between below-ground ecology and soil functions

Abiotic factors, such as climate and soil texture, are major determinants of ecosystem function – however, the relative importance of these factors together with biological interactions in driving soil processes at a range of scales is not well understood (Bardgett 2002). There is therefore a need to identify the critical biological feedbacks to the abiotic controls, particularly to inform ecosystem models (Andrén *et al.* 1999). Mechanistic understanding of ecological interactions is needed if the effects of human management (intentional and unintentional) are to be evaluated and remedied (Brussaard 1998). While the precise role of many below-ground organisms in relation to soil processes is not fully known, functional groups (Section 2.1.10) provide a useful frame to describe interactions and make links between below-ground ecology and soil functions (Table 2.2; Wall *et al.* 2004). Decomposition is a central process for the delivery of most ecosystem services (Table 2.2); as described above, decomposition is the result of the intermeshing vital processes of many soil organisms (Figure 2.2).

Table 2.2 Functional groups involved in the provision of ecosystem services Adapted from Wall *et al.* 2004

SERVICE	FUNCTIONAL GROUP
Provisioning	
Animal food production	None
Plant food production	Primary producers, decomposers
Biochemicals/medicines	Decomposers, primary producers
Fresh water regulation	Decomposers, macroengineers
Non-living materials	Decomposers, N_2 fixers, primary producers
Supporting	
C sequestration	Decomposers, microengineers, primary producers, macroengineers
Trace gases/ atmospheric composition	Trace gas producers/removers; decomposers, macroengineers
Soil formation and habitat provision (structure)	Decomposers, microengineers, primary producers, bioturbators, macroengineers
Nutrient cycling	Decomposers, mutualists/symbionts, N_2-fixers, S transformers, trace gas producers/removers, primary producers, detritivores – litter transformers, predators, bioturbators, macroengineers
Biocontrol	Mutualists/symbionts, predators (particular species within these groups including insects, nematodes, fungi, bacteria, viruses)
Detoxification/ waste treatment	Decomposers, trace gas producers/removers, primary producers
Flood/erosion control	Microengineers, primary producers, decomposers, bioturbators, macroengineers
Climate regulation	Decomposers, mutualists/symbionts, N_2-fixers, S transformers, trace gas producers/removers, bioturbators, macroengineers
Cultural	
Aesthetic	Decomposers, bioturbators, macroengineers.

2.5.1. Food and fibre production

Water balance

Water is a key component of the soil matrix; many soil organisms are highly dependent on the presence of water films (bacteria, protozoa, nematodes) and most require a moist environment for optimum activity. High water contents in soil facilitate water and solute transport within the soil matrix, but also restrict the exchange of gases; diffusion of gases through air is 10^4 times faster than through water (Young and Ritz 2000). Plant demand for water is one of the main driving forces of soil water balance mediated by the interaction of climate and soil structure (Figure 2.5); for some soils, supply of water from groundwater or irrigation plays a significant role. Modification of soil water balance as a result of agricultural management may have an impact on the recharge of aquifers and the maintenance of proximate wetlands.

The geometry of the soil pore network defines the soil habitat space (Young and Ritz 2000) and also controls the movement of water. Water can move freely only in large pores (> 300 μm diameter). Interactions between water and soil surfaces hold water in soils and prevent rapid movement in smaller pores. Plants can exert large forces to extract water from fine pores within the soil; but except in the case of collapsing pores within clay domains very small pores in soil (< 0.2 μm diameter) will always be water-filled. The drier the soil the more direct routes for gas exchange will be in place; the converse is also true, the wetter the soil the more direct routes for transfer of water, solutes and many soil organisms. Because of the complex 3-D framework that soil structure provides a wet soil which is well structured and has a good mix of pore sizes will contain mosaic of anaerobic volumes embedded in an aerobic matrix.

Pore size distribution and connectivity are key factors controlling water movement; soil organisms affect both (Figure 2.5). Biological interactions in soil also usually have positive impacts on soil structural stability (Figure 2.3). Therefore, biological modification of soil structure has significant effects on infiltration, water retention and drainage. The interaction between rainfall (amount, intensity) and soil surface structure (stability of a network of large transmission pores) determines partitioning between surface runoff and infiltration. The extent of ground cover by plants modifies rainfall intensity and tends to increase infiltration; the use of mulches has the same effect.

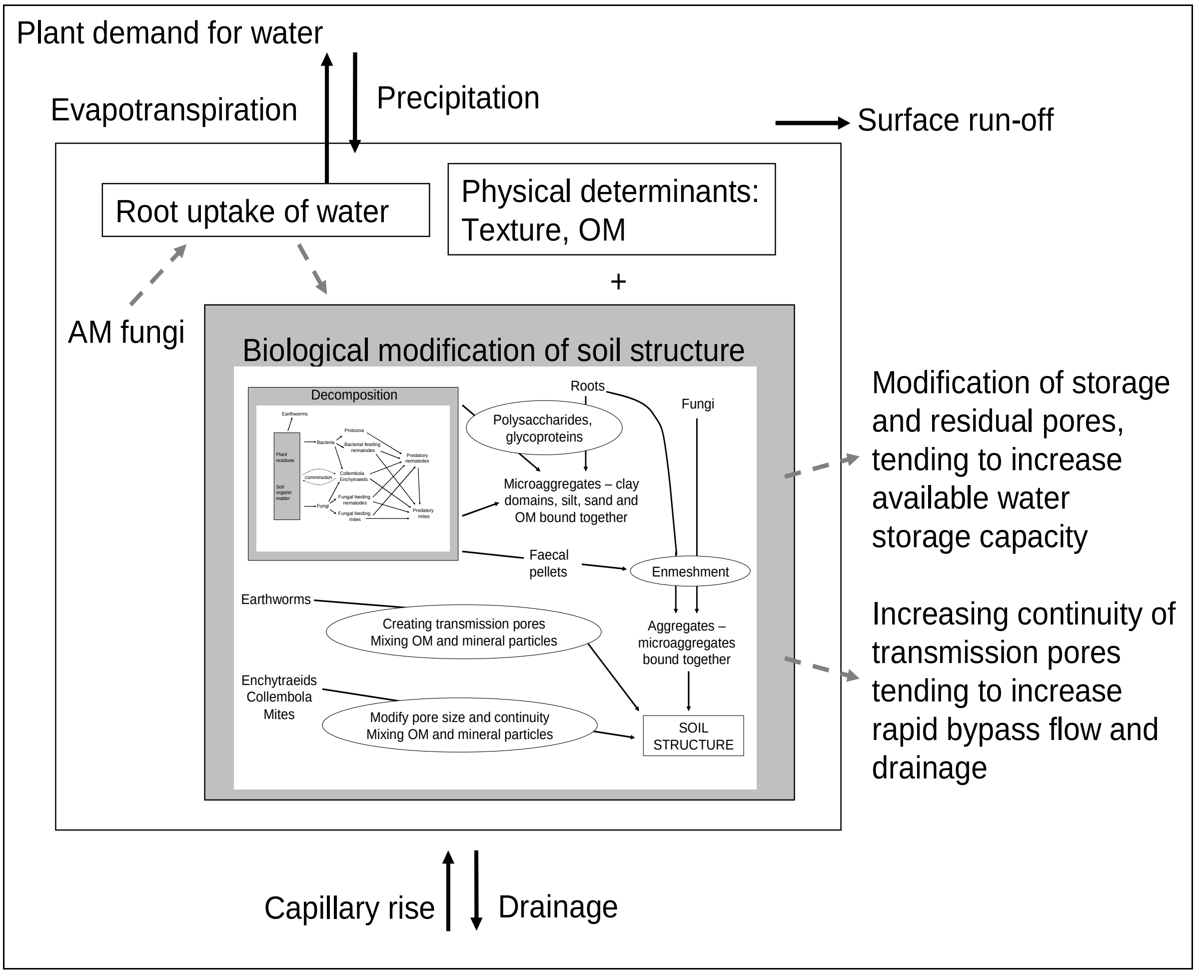


Figure 2.5 Impact of biological modification of soil structure (Figure 2.3) on soil water balance.

Nitrogen (N) cycling

Plants take up N from the soil solution as ammonium (NH_4) and nitrate (NO_3); in some cases simple organic N compounds are also taken up by plants (Nashölm *et al.* 2000; Jones *et al.* 2004). More than 90% of the total nitrogen found in soils occurs in high molecular weight organic polymers; N is an essential component of amino acids, cell peptides and proteins and incorporated into a wide range of other biologically essential compounds such as nucleic acids and chitin. Consequently the dynamics of N in soils (Figure 2.6) are intimately connected with the decomposition of OM (Figure 2.2).

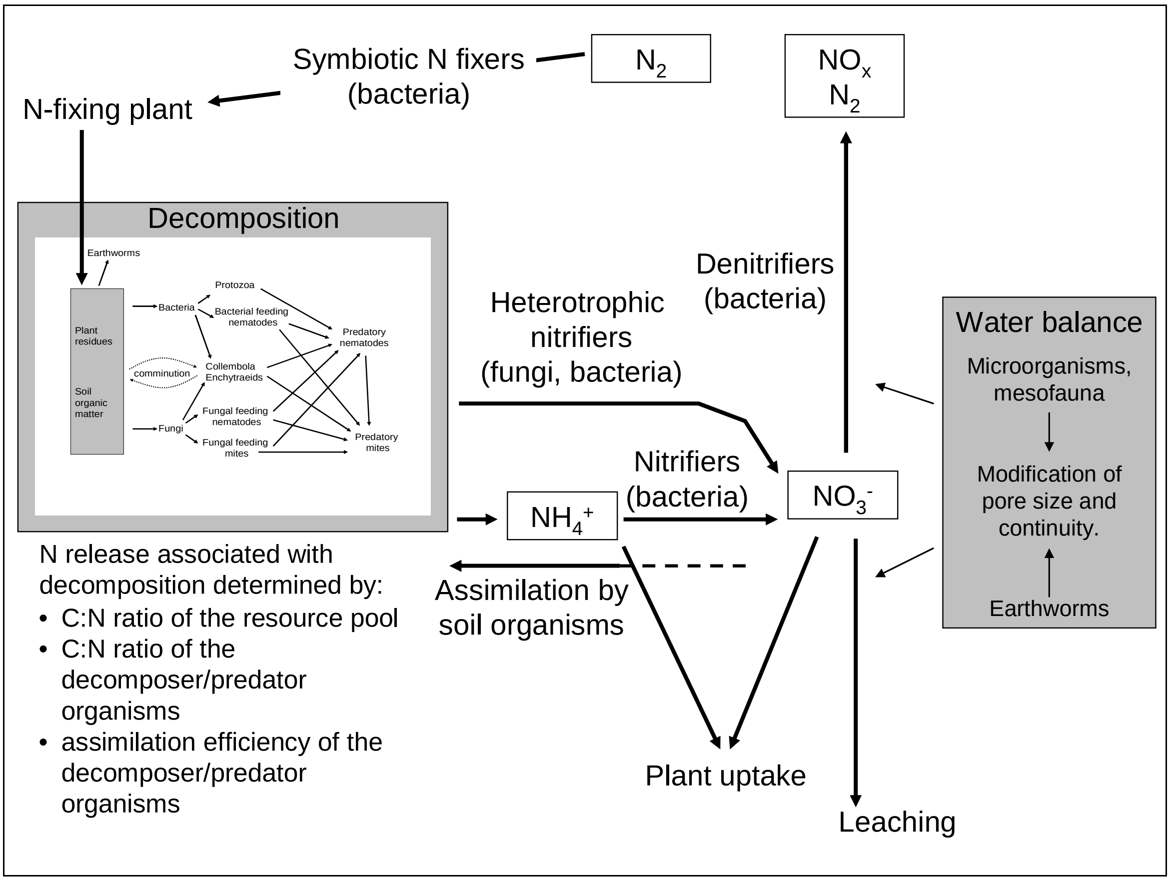


Figure 2.6 Key soil organisms involved in the soil N cycle which in part results from the interaction of decomposition processes and the soil water balance.

The N content of soil can be increased indirectly through symbiotic N fixing processes (Figure 2.6). 40 million tonnes of N are estimated to be fixed by field crops and pasture species globally each year (Jenkinson 2001). In agricultural systems, much of the N fixed is harvested but a significant proportion will enter the soil OM (via crop residues, excreta of grazing animals etc) and be subject to decomposition. N fertilisers are manufactured by industrial N fixation and largely contain immediately plant available forms of N. Incorporation of crop residues, manures etc may increase soil N content in a particular place, but such applications usually represent transfers of N within/ between farming systems, rather than imports of N.

The gross release of N into mineral forms (NH_4^+) during decomposition (gross mineralisation) depends on the C:N ratio of the resource and the C assimilation efficiency of the decomposers (Hart *et al.* 1994). The net release also depends on the C:N ratio and nutritional status of the decomposer/predator (Figure 2.6), which controls the rate of assimilation (immobilisation) of N released. Mineralisation processes also occur within the guts of larger soil animals.

Organic materials applied to soil can have a wide range of C:N ratios; it has been estimated that in arable soils the critical C:N ratio of added materials is c. 20. At larger C:N ratios additional mineral N is needed to support decomposition. Soil organisms are able to immobilise NO_3, but NH_4 immobilisation is more energetically favourable (Recous *et al.* 1990). Critical C:N ratios will differ depending on whether the decomposer sub-system is bacterially or fungally dominated (Figure 2.2); fungi and bacteria have different assimilation efficiency and C:N ratios. In grassland soils the role of cycling through NH_4 as described above, may be reduced in importance compared to the release and uptake of small soluble organic N compounds (Murphy *et al.* 2000).

Nitrification is the biological oxidation of nitrogen in soil to a more oxidised form. Nitrification in soils is dominated by the chemoautotrophic oxidation of NH_4 to NO3 via nitrite. This is a two step process mediated by *Nitrosomonas* and *Nitrobacter* respectively; rates are usually limited by the availability of NH_4 (Moore *et al.* 2004). Heterotrophic nitrification which releases NO_3 directly from organic N without NH_4 as an intermediary is also known to occur, particularly under acid uncultivated situations (Pennington and Ellis 1993).

The soil NO_3 pool is particularly vulnerable to loss from the soil. Leaching losses of NO_3 are related to the rates of drainage from soils. In contrast NH_4 is prevented from leaching in drainage as a result of cation exchange on the surfaces of clays and and/or humified organic matter. Increasing numbers and connectivity of transmission pores, such as created by earthworm burrows and plant roots, can increase the rates of leaching loss; tillage often disrupts the continuity of pores from the surface into the subsoil (Young and Ritz 2000). The modification of pore size distribution through the activity of mesofauna can increase the amount of water retained in the soil at field capacity, particularly for coarse textured sandy soils, and may reduce drainage and hence leaching. Denitrification, discussed further below in relation to trace gas balance from soil, also leads to losses of NO_3 from soil. NO_3 replaces oxygen as the terminal electron acceptor in respiration where oxygen concentrations are limiting for a wide range of bacterial groups that are facultative anaerobes.

Phosphorus (P) cycling

Total P in soils in the UK averages c. 700 $\mu g\ g^{-1}$ (Cooke 1958). However, concentrations in soil solution are very low (c. 0.1 $\mu g\ g^{-1}$) and only a very small fraction of total P is available to plants. Many studies of P cycling in soil have focussed on the complex chemical equilibria that control plant available phosphate (H_2PO_4). However, about 25% of soil P is held in organic forms (Wild, 1988); the release of this P pool is linked to decomposition processes, as for N (Figure 2.7).

Some bacteria, fungi and plant roots excrete organic acids which modify the chemical equilibria controlling P availability from calcium phosphates (including rock phosphate) by reducing pH and/or chelating calcium, thus increasing the solubilisation of P (Figure 2.7). Plant uptake of P is also facilitated where AM fungal associations with roots are present. H_2PO_4 concentrations in soil solution are not large enough to mean that mass flow (i.e. the movement of nutrients in solution to the root in response to transpiration demand) can meet plant requirements, and diffusion of H_2PO_4 is very slow. Hence increased apparent root surface area mediated by root hairs and AM fungal associations gives plants access to an increased volume of soil solution and more available P. Where soil becomes very dry, diffusion rates are reduced further, consequently changes in pore size distribution which increase water holding capacity as a result of biological activity, particularly in sandy soils, can also improve the P availability for plants.

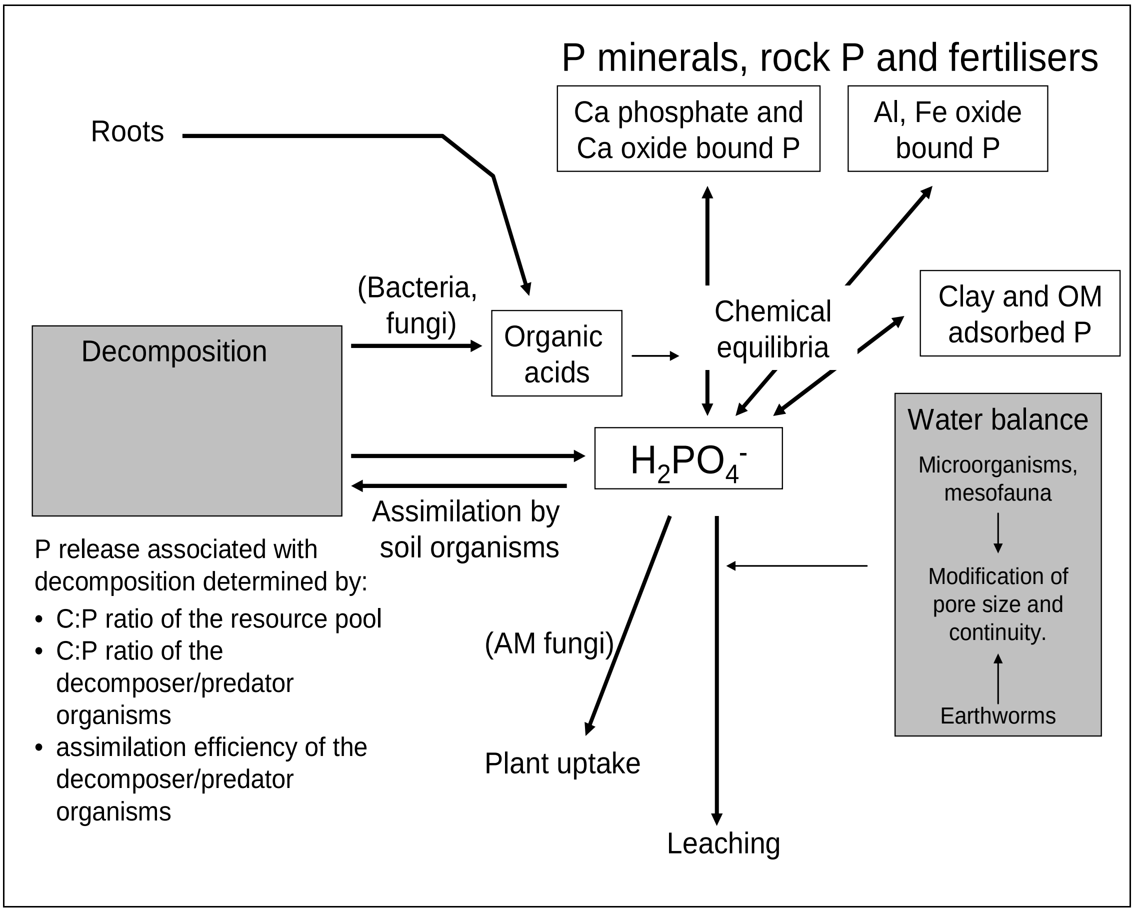


Figure 2.7 Key soil organisms involved in the soil P cycle, which results from the interaction of mineral equilibria, decomposition processes, soil structure and water balance.

Phosphorous is often lost from soils as a result of sediment transport in runoff; surface soils are often the most enriched in P, as a result of fertiliser application and plant residue return. Factors affecting the balance between infiltration and runoff, including biological activity as discussed above, therefore control erosive losses of P. Where adsorption sites for H_2PO_4 along the walls of transmission pores become saturated then leaching of P may also become a significant route for loss (Heckrath *et al.* 1995; Fortune *et al.* 2005). As for all leaching losses, continuity of transmission pores, facilitated by earthworm activity, has a large role in facilitating loss.

Plant disease

Direct damage to plant roots by root feeding and parasitic organisms can be considerable (particularly beetle larvae and nematodes). In Britain, the most important plant parasitic nematodes are the root knot nematodes (*Meloidogyne*) and the cyst nematodes (*Globodera*). In addition a wide range of soil micro-organisms that can cause plant disease, dominant amongst them fungi, have been gradually identified and their ecology described (Garrett 1956; Bruehl 1987; Hillocks and Waller 1997). Generalisations about management or causes between diseases are not possible. Saprophytic survival of the pathogens and infectivity with regard to the host are affected by a range of factors. In most cases the exact mechanism remains unknown and it has been observed that the more that is written about the subject the less we really know (Bruehl 1987). The interaction between plant species and soil-organisms is critical – non chemical approaches to the control of soil borne disease include careful crop rotation design and increasingly the selection of resistant crop varieties.
As well as the occurrence of naturally resistant crop varieties, some soils also seem to be naturally disease suppressive (Menzies 1959). Certain soil micro-organisms have direct inhibitory effects on various soil pathogens. For example *Coniothyrium minitans* has been shown to paralyse a wide range of sclerotia in soil (Adams and Ayers 1983) and *Gliocladium virens* shows parasitic inhibition of *Rhizoctonia solani* and antagonistic effects against *Pythium ultium* (Howell 1982). Hyperparasitism of pathogenic nematodes by fungi has been shown to be an important factor in the development of nematode suppressive soils and also as a successful biocontrol strategy (Kerry 1981; Atkins *et al.* 2003). Links have been made between actinomycete population size and the suppression of some Pythium root rots (Cooper and Chilton 1950). It has been suggested that antagonism to the take-all fungus (*Gaeumannomyces graminis var Tritici*) reside in a single bacterial species and depended on a single bacterial process (Keel *et al.* 1992). Antibiotic production *in situ* is likely to have some role in the development of suppressive soils (Smiley 1979). A wide range of mechanisms (including parasitism, direct and indirect antagonism) is now thought likely to operate in disease suppressive soil (Mazzola 2002) and interactions with soil chemical and physical properties also needed to be considered (Duffy *et al.* 1997). The complexity of these biological interactions, as well as our limited understanding of them, means that it has not been possible to draw out a simple interaction web that shows the links between below-ground ecology and plant disease.

2.5.2. Environmental interactions

Carbon balance

The carbon dioxide (CO_2) efflux from soils is the net effect of all heterotrophic aerobic decomposition processes (Figure 2.2). Carbon assimilation is also a net result of decomposition (Figure 2.8). Where the total amount of C input to soil is increased then both the total CO_2 efflux and C assimilated is likely to increase (e.g. Jacinthe *et al.* 2002). However, the quality of the C input and a range of other factors affect the partitioning between CO_2 production and C assimilated to biomass and humus. Microbial efficiency has been shown to be affected by soil texture (Schimel 1986), mineralising substrate/residue type (Hart *et al.* 1994, Mueller *et al.*, 1997) and temperature (Henriksen and Breland 1999). It is likely that a range of similar factors will affect assimilation efficiency at higher trophic levels. Decomposable carbon may also be protected from decomposition as a result of spatial location and a range of other mechanisms (Powlson 1980; Six *et al.* 2002) CO_2 consumption by soils is measurable but has only a small impact on overall C balance (Miltner *et al.* 2005). Methane (CH_4) fluxes, both efflux and oxidation, are very small compared to CO_2 and have almost no impact on C balance, except under completely waterlogged conditions where aerobic decomposition is almost completely suspended. Under waterlogged conditions rates of decomposition are slowed considerably and organic matter will accumulate, usually at the soil surface.

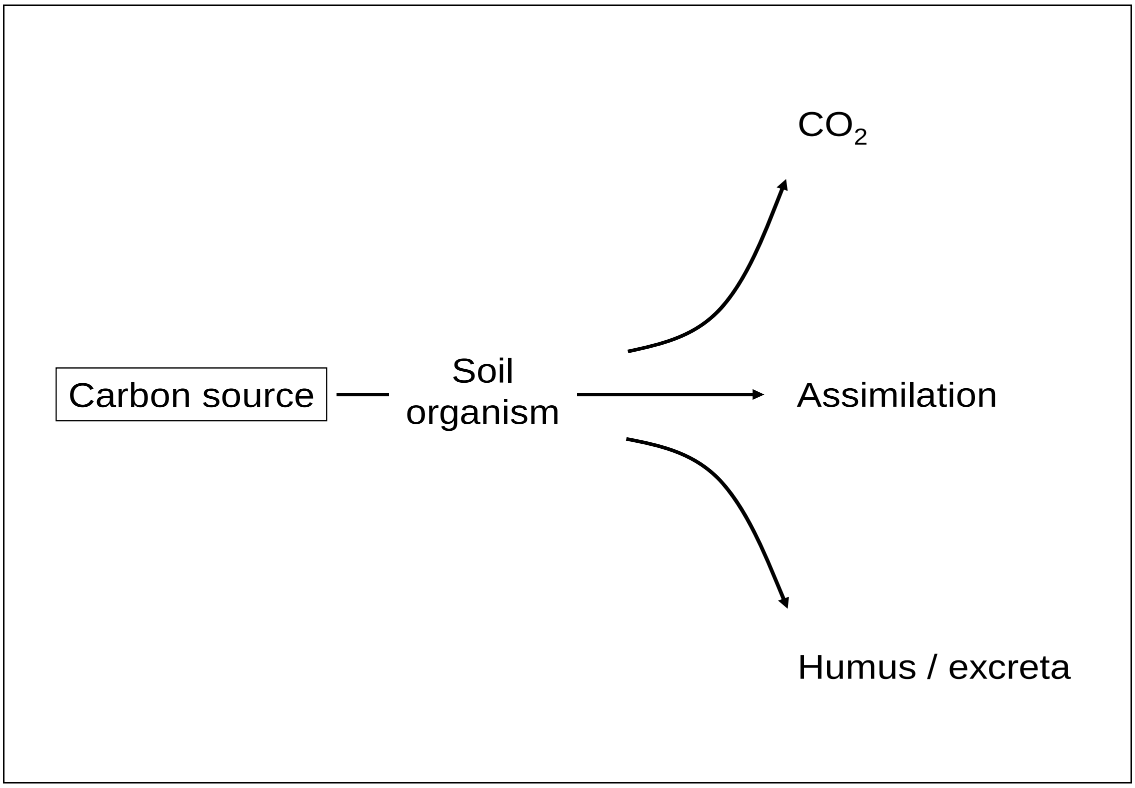


Figure 2.8 Carbon balance in heterotrophic decomposition processes in soil.

Trace gas balance

The gaseous end products of microbial activity include not only CO_2 discussed above, but also nitrogen gas, nitrous oxide, methane and many other volatile compounds (Figure 2.9). The trace gas balance mediated by soils is an important component of atmospheric regulation; many of these gases contribute to the greenhouse effect. The availability of oxygen for microbial metabolism is a key controlling factor in many of the biological processes controlling trace gas production and consumption. Consequently water balance is an important controlling factor.

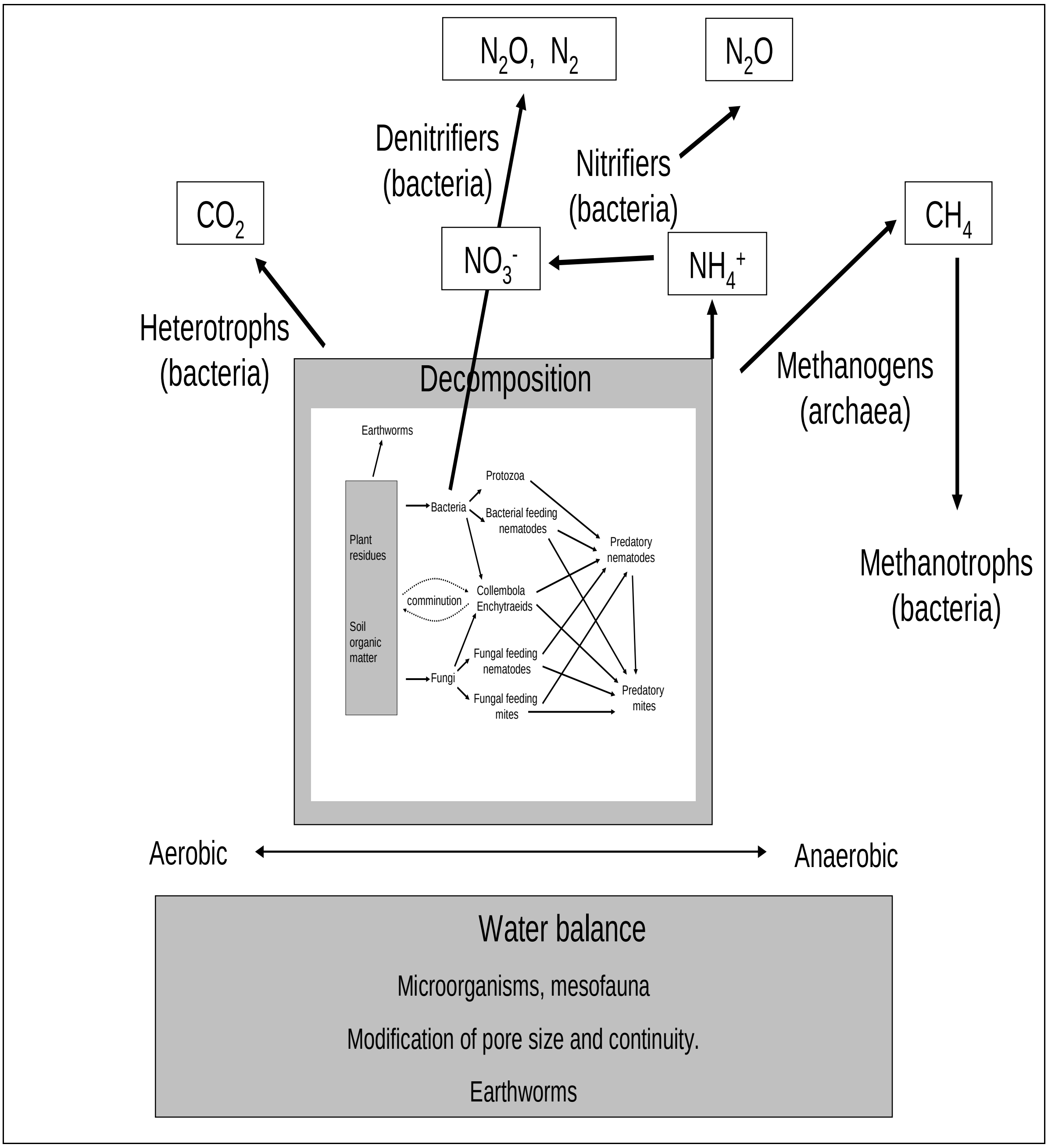


Figure 2.9 Key soil organisms involved in the trace gas balance mediated by soils which largely results from interactions of decomposition processes and water balance, which controls oxygen availability.

CO_2 release from soils is the consequence of heterotrophic decomposition processes carried out by a wide range of bacteria under aerobic conditions. Under anaerobic conditions the decomposition/fermentation of complex organic compounds by a variety of anaerobic bacteria releases organic acids. Methane production is then controlled by a highly specialised group of anaerobic microbes (methanogens) which further reduce these organic acids releasing methane (CH_4). Under aerobic conditions, autotrophic methanotrophs (methane oxidising bacteria) provide a biological sink for CH_4; to a much lesser extent NH_4 oxidisers also provide a pathway for methane oxidation in aerobic soil. Consequently even in permanently waterlogged soils, an aerobic surface layer can significantly reduce the net methane efflux to the atmosphere.

Denitrification, i.e. the biological reduction of nitrate to nitrous oxide and/or N_2, is driven by bacterial groups, which use nitrate or nitrite as the terminal electron acceptor in respiration under anaerobic conditions. Consequently the process dominantly occurs during decomposition where both nitrate and anaerobic conditions occur in soil. Hotspots of denitrification activity are often associated with pockets of organic matter in soil (Parkin 1987). There is evidence of high adaptability and ubiquity of bacterial groups with regard to the process of denitrification; denitrification is associated with a number of bacterial genera. (Knowles 1982). Nitrous oxide (N_2O) production can also occur during the chemoautotrophic nitrification of NH_4 to nitrite. Fluxes of nitrogen oxide and nitrogen gas in agricultural systems often occur soon after fertiliser application under warm and moist soil conditions (Clayton *et al.* 1997).

Water quality

Diffuse losses of N and P from agricultural soils as a result of transport in runoff (soluble forms and sediments) and through leaching can have very significant effects on the ecological quality of surface waters. The roles of below-ground ecology in these processes is discussed in previous sections.

The transport of bacteria (particularly human pathogens) from soil to surface waters is also a key water quality issue. Surface run-off from grazed fields has been identified as the main route by which pollution occurs (Vinten *et al.* 2004a). There is a higher pollution risk from grazing livestock than from application of slurry (Vinten *et al.* 2004b). Infiltration of water containing pathogenic bacteria into the soil tends to reduce pathogen loading. Within the soil competition with the existing bacterial population and adsorption of bacteria onto soil surfaces in pores (Camper *et al.* 1993) reduces pollution risk arising from leaching; routes through soil are not considered to be a major source of bacterial pollution to waters (Vinten *et al.* 2004a).

2.5.3. Support of ecological habitats and biodiversity

Much of the terrestrial biosphere is found in soil and the diversity in soils is several orders of magnitude higher than that seen in above ground ecosystems (Heywood 1995). A preliminary description of soil organisms was given in Section 2.1. Soil biodiversity is of value in its own right; thus there are over 100 species of soil invertebrates and fungi in UK Biodiversity Action Plans (mostly associated semi-natural and natural ecosystems).

However, it is not yet clear how goals can be set for soil as an ecological habitat in its own right and for soil biodiversity *per se* within nature conservation and heritage interests.

Farmland species of birds and other predators depend, at least in part, on below-ground ecology (Figure 2.10; Smedding and de Snoo 2003). Consequently management effects that impact on below-ground ecology will potentially impact on the support of biodiversity for a wide range of farmland species. The interaction between above and below-ground ecosystems is also key in maintaining particular habitat types e.g. heath where development of the characteristic vegetation, land management by grazing and podzolisation processes in the soil are closely linked (Nielsen *et al.* 1999).

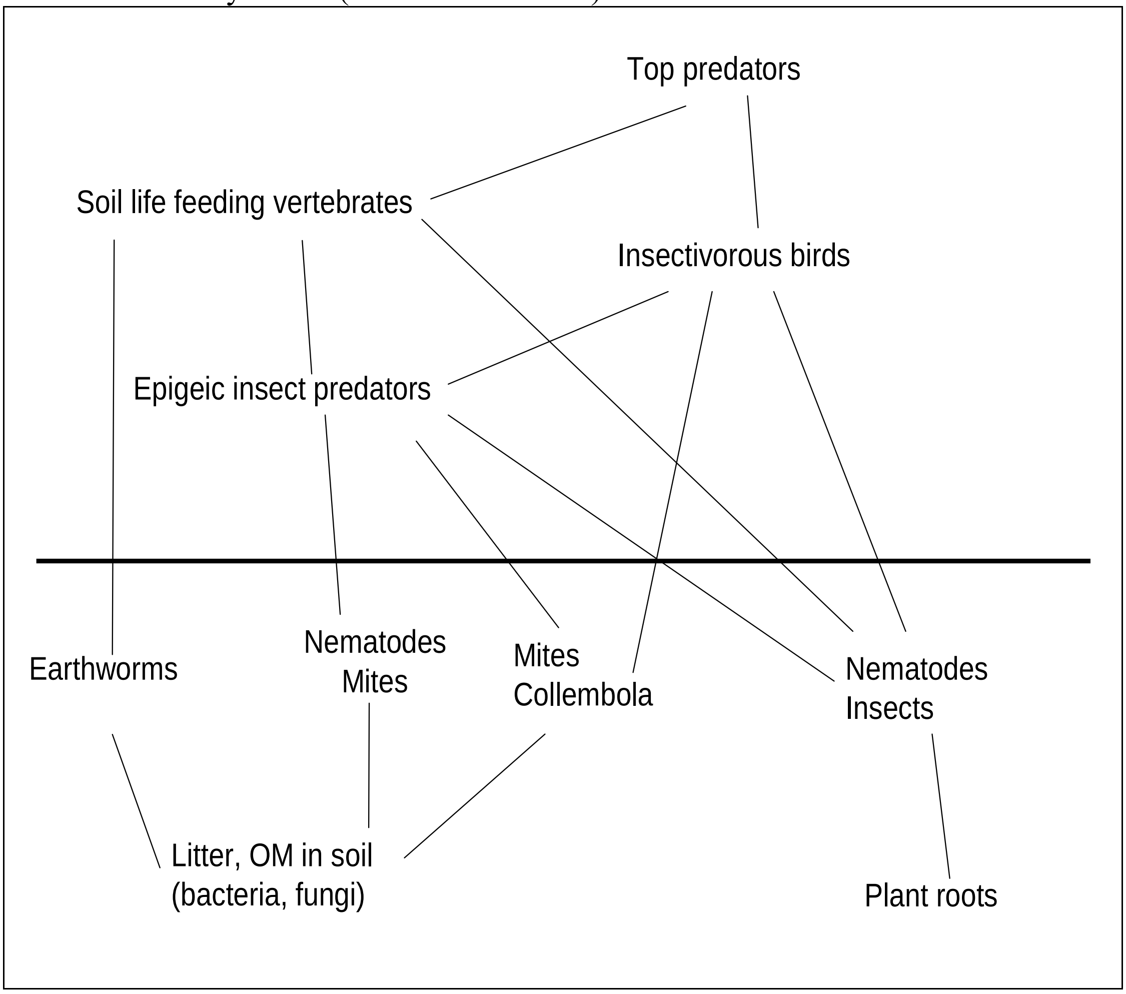


Figure 2.10 Interaction web showing the direct role of below-ground ecology in providing food sources for birds and mammals and hence one of the roles of below-ground ecology in delivering the "support of ecological habitats and biodiversity" function of soils. As the relationships shown are largely predator/prey interactions the lines represent the two-way regulating relationship. Adapted from Smedding and de Snoo 2003.

3. Impacts of agricultural management practices

The potential use of land for agriculture in any location is rarely unconstrained. Assessment of land use quality uses a number of relatively fixed site characteristics to define the quality of land (e.g. climate, slope, some soil factors). These site factors are largely unmanageable and consequently they set the boundary for the range of agricultural practices that are possible. For example, in the Agricultural Land Classification system used in England, land of Grade 5 is not suitable for cultivation, whereas it would be unusual for land of Grade 1-3a not to experience some at least rotational cropping with arable or horticultural crops. A similar range of fixed factors (climate, depth, stoniness, mineralogy, texture) has also been identified as controlling the maximum potential soil OM content (Ingram and Fernandes 2001; Dick and Gregorich 2004). Because of the close relationship between soil OM contents and the size of the soil microbial biomass pool (Wardle 1992), it is not unreasonable to suggest that a similar range of factors might define the potential size of the below-ground biomass populations. It is clear that these factors have a large direct influence on the composition, structure and flows between soil habitats identified above. 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.

The limited number of biogeographical studies of below-ground organisms (largely completed only for mites, e.g. Luxton 1996; collembola, e.g. Christiansen and Bellinger 1995 and earthworms, e.g. Reynolds 1994), show that site factors are often a major determinant in the development of below-ground communities:

- Black and Parekh (1998) compared soil mesofaunal and microbial diversity in winter wheat under integrated (reduced inputs and reduced tillage) and conventional farming systems at three sites across the UK and showed that climate and/or other site factors including differences in soil texture had significant effects on total mesofauna, collembola and mites;
- Fulthorpe *et al.* (1998) in studies of bacterial populations across scales from metres to continents showed some gross similarities between the composition of microbial populations at a coarse scale, nonetheless they also showed strong adaptation to locality;
- Marschner *et al.* (2001) showed that soil type (differences in texture, pH, P and K status) had a major effect (along with plant species) in determining the bacterial community structure in the rhizosphere;
- Sessitch *et al.* (2001) demonstrated that bacterial community structure was affected to a greater extent by the particle size fraction than by long-term fertilisation applied.

Fixed site factors have a major effect on below-ground biomass in terms of both size and activity. 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 combination of fixed site factors. However, it is also clear that a range of land management practices and other natural disturbances (e.g. fire) may also influence the below-ground biomass both directly (through physiological effects on populations) and indirectly through impacts on soil habitats and/or other organisms. (Figure 3.1). This model is clearly incomplete, but nonetheless we believe that it provides a useful simple framework within which to discuss the impacts of agricultural management on below-ground biodiversity.

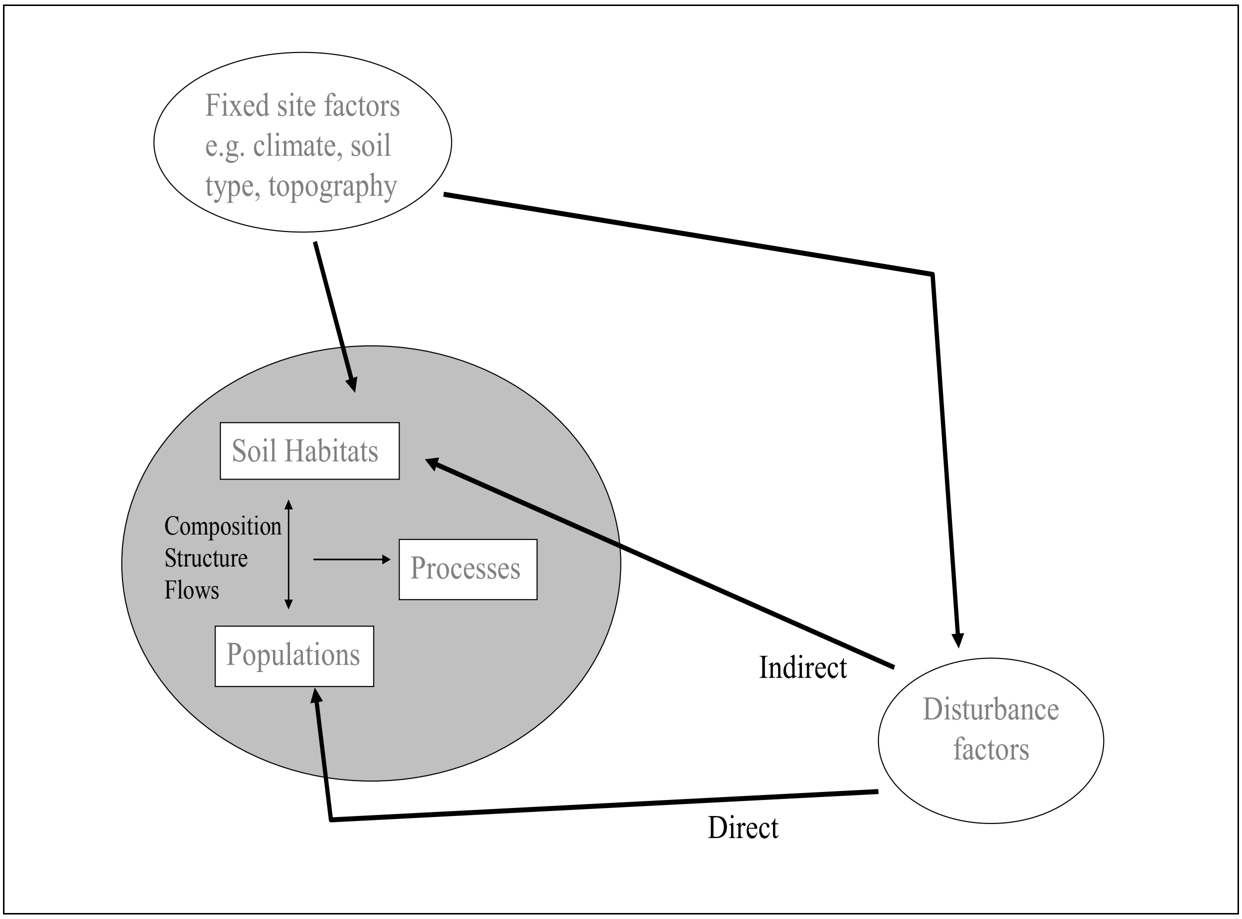


Figure 3.1 Simple schema showing the interaction of fixed site and disturbance factors in their effect on below-ground ecology.

Increased use of mechanical and manufactured inputs and increased specialisation of production often mark intensification of agricultural production; regulation of the agro-ecosystem through biological processes is replaced through regulation by inputs (Giller *et al.* 1997). Intensification of farming practices also tends to increase homogeneity of production systems and their associated landscapes above-ground (Benton *et al.* 2003). Consequent widespread decline in farmland biodiversity for birds, mammals, arthropods and flowering plants has been observed (e.g. Krebs *et al.* 1999). What about below-ground? There is evidence that agricultural systems are associated with simplified soil food webs compared to semi-natural systems; differences also occur between dominantly pastoral and arable agricultural systems (van der Putten *et al.* 2004). Compilation of the limited data on protozoan species richness in agro-ecosystems and neighbouring natural biotopes (Foissner 1997) suggested that the species richness of testate amoebae is invariably and distinctly reduced under agricultural production so that only a residue of the original more diverse population is retained. Nonetheless taking all species into consideration, Giller *et al.* (1997) found little evidence for agricultural intensification causing a loss of biodiversity in soil. Wardle *et al.* (1999) studied the impact of agricultural intensification over seven years and could also find little evidence to support the view that agricultural intensification need necessarily have adverse consequences for soil microbial biomass populations or activity.

Given the size and scope of this project, it was not possible to review the effects of all possible agricultural management practices. There are also a range of non-agricultural activities that potentially have an effect on agricultural soils e.g. off-road trail driving etc. However, these are beyond the scope of this review. Typical practices within the range of UK farming systems were identified (see Section 4 for a consideration of the systems themselves). These were tillage, crop and rotation selection, crop residue management, grazing (intensity and livestock species), inputs of veterinary medicines, herbicides, pesticides, lime, mineral fertiliser, organic amendments and drainage/irrigation. Some of these practices are directly targeted at managing soil-organisms (e.g. use of nematicides in potato production) and some have direct but un-intended effects (e.g. tillage). However, most agricultural practices are targeted at/affect other soil properties and hence have indirect effects on below-ground ecology. Table 3.1 shows the number of papers found within a large bibliographic database (1990-to date) covering this topic area. This was carried out on Web of Knowledge on 24 February 2006 using the search terms identified in the table e.g. Soil not (rice or tropical or forest*) and earthworm* and (farmyard manure or slurry or compost).

Individual combinations of management practice and group gave from 0 to 920 references. The focus has therefore been mainly on key review papers and recent developments with a particular focus on the impacts of common practices within UK farming systems rather than a full meta-analysis. It is perhaps surprising that there appear to have been very few studies addressing the impact of clover based leys on soil organisms. With the exception of impact on microbial biomass, the impact of crop residues themselves have also received surprisingly little attention. Studies of the impact of grazing by domestic species on belowground biodiversity are difficult to separate in the literature as the term ‘grazing’ is unsuitable due to its dual use for above and belowground grazing. The use of the term defoliation in the above search may therefore underestimate actual paper numbers.

Most studies compare established management practices rather than studying the impact of a change in management. Because agricultural systems are typically in dynamic non-equilibrium states, effects of disturbance are likely to take some time to settle down. There are also very few studies, which compare the relative impacts of different farming practices; most studies focus on the study of a particular practice using a range of intensities or plus/minus comparisons. It can be difficult to distinguish direct and indirect effects of land management practices from data collected in the field; consequently many of the studies reviewed here provide a report of the net result of both. Seasonal dynamics of below-ground populations in response to the dynamics of temperature and water are often not taken into account in studies comparing agricultural practices. These changes may have greater effects on populations than farming practice e.g. Spedding *et al.* (2004) observed larger seasonal variations in fungal and total microbial biomass content than effects that could be

Table 3.1 Results of Web of Knowledge search showing number of hits for each combination of group and management practice. [* = wildcard]

Management practice	crop residue	tillage	drainage	defoliation	pesticide	herbicide	insecticide	(worming or nematicide)	lime	fertil*	(farmyard manure or slurry or compost)	ley and clover	crop rotation	(green manure or cover crop)
Group														
+ collembola	2	34	0	2	19	17	22	1	1	30	13	1	8	3
+ earthworm*	18	169	29	1	51	36	34	2	27	174	76	3	23	15
+mite*	2	29	1	2	14	5	14	3	2	25	10	1	2	3
+protoz*	4	14	5	1	4	3	4	1	2	24	13	1	2	3
+nematode*	8	113	5	16	44	45	45	198	12	199	71	0	101	79
+ beetle*	3	54	2	24	21	17	66	0	1	50	7	0	9	6
+ invertebrat*	4	34	11	4	33	9	24	1	2	58	12	0	11	4
+rhizobi*	2	20	3	5	9	23	6	0	36	239	22	1	15	9
+bacteria	10	101	70	6	123	227	45	24	40	445	337	2	54	23
+archaea	0	1	0	0	1	0	1	0	0	1	1	0	1	0
+microbial biomass	75	472	36	23	65	107	12	3	43	763	228	2	111	110
+fung*	28	280	28	39	220	222	99	49	72	920	227	2	119	76
+mycorrhiz*	3	96	9	22	10	14	5	3	38	636	45	1	48	25
+enchytraeid*	0	10	0	4	5	4	3	0	2	8	4	1	2	0

attributed to the tillage or the residue treatments imposed. Smith *et al.* (2001) showed that while there was a relatively large stable population of micro-organisms in the soils studied, there were significant changes in components of the bacterial population through the season. Smith *et al.* (2001) also observed that different approaches to the study of the same population, here culturing and molecular approaches used to study bacterial populations, showed different patterns of response both to the agricultural practice and seasonally. Consequently care needs to be taken when comparing data collected in different ways, at different times of year and on different sites. The evidence base used here derives from a number of scientific studies employing a range of methodologies. This often makes direct comparison of the results of different studies difficult. Table 3.2 highlights observations made during this review on experimental design issues. The recent development and use of molecular techniques has allowed enormous developments in our knowledge of soil biology. AM fungi are an excellent example. In the past classification depended on the use of spore morphology, but not all species of AM fungi sporulate meaning that older studies of diversity of AM fungi in agroecosystems were unable to give a complete picture of the system (Douds and Millner 1999).

Table 3.2 Limitations of studies

Soil type – There are surprisingly few studies which examine the impact of soil on soil organisms. Furthermore, there are a number of published studies which contain limited or no information on soil type or soil properties.

Duration – Studies that take into account both short and long term effects of agricultural practices are needed. In many cases studies of the effect of a particular biocide have not been linked to the half-life of the chemical in soil. The full effects of some management practices e.g. tillage, may not be evident within one growing season and in some cases it may be important to study effects over a whole rotation.

Value of different scales of approach – Pot experiments can give useful and interesting information on behaviour of individual species. Care must however be taken in scaling up these studies to the less controlled field situation.

Biocides – In practice in the field, biocides are often applied as combinations of products and on multiple occasions during a growing season. The published literature often addresses the effect of one product in one application.

Field experimental design – This is addressed in more detail in Section 4.1.

Invalid comparisons - unbalanced nutrient additions.

Invalid comparisons of crops - e.g. Kahiluoto and Vestberg (1998) compared the effects of AM inoculation of P uptake in leek after conventionally managed continuous cereals with the effects of AM inoculation of P uptake an organically managed ley/arable rotation.

3.1. Tillage

Tillage is the manipulation, usually mechanical, of soil properties to modify conditions for crop production. Most tillage operations are performed to decrease soil density in the disturbed zone and practices can be grouped into those which invert, loosen, mix or crush the soil. Ennis (1979) suggested that more than half of tillage operations carried out within conventional systems were associated with weed management; tillage is also usually a key step in seedbed preparation. Tillage may also incorporate crop residues, fertilisers or other amendments. All of these operations are often combined within a single tillage operation in the field. However in the following section the effects solely of soil disturbance through tillage will be considered.

At the outset it is critical to recognise that the resultant soil properties following any tillage operation, even where the same implement is used, depend on a combination of equipment factors (including depth, energy input, speed) and soil factors (including water content, texture, residue cover). Consequently “it is difficult to visualise, let al.one predict, the soil conditions resulting from a given operation” (Unger and Cassel 1991). Nonetheless it is clear that the aim of any tillage operation is the modification of soil pore distribution (usually the aim is to increase macroporosity) and pore connectivity.

Wardle (1995) carried out an extensive review of the impacts of disturbance through tillage on food-webs in agro-ecosystems. The conclusions of his meta-analysis show that tillage tends to reduce large soil organisms (beetles, spiders, earthworms) more than the smallest ones (bacteria, fungi). On average some intermediate groups such as bacterial feeding nematodes, mites and enchytraieds even show small population increases. For most species groups the effects on populations are as a result of indirect effects arising as a result of the modification of soil habitats, particularly the continuity of water filled pores and water films (Winter *et al.* 1990). Consequent smaller impacts of changes in tillage practice are often seen on very sandy soils (Spedding *et al.* 2004).

Some direct inhibitory effects are seen for micro-arthropods (mites, collembolae) where tillage may kill individuals as a result of abrasion and trapping during inversion/mixing (Andren and Lagerlöf 1980). Petersen (2002) showed that tillage to the same depth but with different implements reduced the collebola population to about 1/3 of the pre-tillage level one week after cultivation. However, the stratification of the populations was affected differently depending on whether the soil was inverted or mixed by a tine sub-soiler to the same depth (Petersen 2002). Earthworms are highly sensitive to the direct effects of physical disturbance and injury as result of tillage (Rovira *et al.* 1989). Responses of earthworms to tillage show relatively low variance and hence reasonably predictable effects. While insects (such as beetles and other predatory meso-arthropods) are also inhibited directly as a result of the physical disturbance caused by tillage operations, the large variance of this response means that effects of tillage are often unpredictable (Wardle 1995). Dispersal characteristics of these species along with the presence of local reservoir habitat, will be important in determining the speed of return of populations after disturbance. Variation in these characteristics may explain the high variance observed. In regularly tilled systems indirect inhibition of soil macrofauna is probably also important in maintaining lower populations since there is less surface litter associated with regularly tilled systems and hence a reduction in food resource for both species groups. Shifts in species assemblages have also been seen under tillage for nematodes and earthworms (Wardle 1995). Invertebrate food resources for birds have been shown to be increased in no-till compared to conventionally tilled systems (Tucker 1992).

It is often predicted that fungal populations will be reduced more significantly than bacterial populations by tillage due to the disruption of the hyphal network (Doran and Linn 1994; Young and Ritz 2000). However, Wardle (1995) showed that under similar practices bacterial and fungal populations are reduced slightly and to the same extent by tillage in annual cropping. Petersen *et al.* (2002) showed that despite slight differences in microbial biomass between no-till and chisel ploughed systems, the seasonal dynamic interactions of soil conditions and microbiological properties were similar suggesting that common mechanisms regulate microbial dynamics in both tillage systems. Douds *et al.* (1995) and Jansa *et al.* (2003) showed changes in the community structure of arbuscular mycorrhizal fungi colonising maize roots in tilled and no-till systems. Oehl *et al.* (2003) showed that increased intensity of tillage led to preferential selection for AM fungi species that formed spores rapidly. However in these field experiments it wasn't possible to separate the direct effects of tillage on the hyphal network from indirect effects due to changes in nutrient availability or weed populations. Ahl *et al.* (1998) showed that ceasing inversion tillage and using reduced tillage methods tended to increase the total amount and proportion of fungal biomass in soil relative to bacteria; however, the effects of tillage and other system effects on plant factors were not distinguished. Where soil disturbance takes place in perennial crops e.g. between tree rows in an orchard, then fungal populations tend to show a greater impact of tillage (Wardle 1995).

Mechanical tillage is often also associated with compaction of soil and compaction to a depth of 50 cm has been observed as a result of trafficking (Whalley *et al.* 1995). Aritajat *et al.* (1977) showed clearly that compaction can significantly reduce the number of earthworms and microarthropods in soil under grass, with the extent of effect dependent of the soil type (i.e. to what extent pore size distribution was affected) and species. Populations took a period of several months to recover from a single incident of compaction (Aritajat *et al.* 1977) and it is likely that repeated compaction would have an increased and long-term effect.

In addition to the impacts of tillage *per se* comparisons of arable systems cultivated by conventional and no-till show differences in plant biomass, which may have indirect effects on below-ground ecology. Crop yields vary depending on climate and soil type, neither conventional nor reduced tillage systems have consistently higher yields. However, no-till systems consistently show higher root biomass near to the soil surface (Anderson 1987) as well as deep penetration of roots in earthworm burrows (Cheng *et al.*, 1990). It might therefore be expected that spatial patterns of root exudation, if not also total amounts, might vary between conventional and no-till systems. Placement of crop residues also differs between the systems and no-till systems often show an increased stratification of OM content in what was previously the plough layer (Kay and vandenBygaart 2002). These indirect effects of tillage regimes on plant root patterns may have as significant effect as tillage *per se* on below-ground ecology, not least in the promotion and persistence of greater small scale heterogeneity in no-till systems.

Table 3.3. Summary of tillage impacts on below-ground organisms

Species/group	Average impact of tillage or increased tillage intensity	Key references
Bacteria and archaea	Mild inhibition	Wardle 1995
Rhizobia	No evidence found	
Nitrifiers	Little evidence, stimulation of group 3 *Nitrosospira* (but not group 4) by cultivation	Mendum and Hirsch 2002
Fungi	Mild inhibition	Wardle 1995
Arbuscular mycorrhizal species	Inhibition of AM colonisation of roots and spore numbers	Gosling *et al.* 2006
Protozoa	Little evidence, minor impact	Foissner 1997
Nematodes	Little effect; mild stimulation of bacterial feeders, mild inhibition of fungal feeders and omnivores	Wardle 1995
Mites	Moderate to mild inhibition, some studies show stimulation	Wardle 1995
Collembola	Moderate to mild inhibition, some studies show stimulation	Wardle 1995
Enchytraeids	Little effect, as often stimulated as inhibited	Wardle 1995
Earthworms	Moderate to extreme inhibition	Wardle 1995
Insects	Moderate to extreme inhibition	Wardle 1995

3.2. Crop

Most below-ground organisms are heterotrophic and hence dependent on the decomposition of sources of C in soil rather than photosynthesis or autotrophic mechanisms for energy. Some studies have shown a correlation between increase in plant biomass production and total bacteria and fungi in soil e.g. Bardgett *et al.* (1999) in a gradient of increasing productivity in upland grasslands. Bare fallow established in a grassland system was shown to reduce some soil fauna (earthworms, collembola, predatory nematodes) but not mites, other nematode groups or bacterial populations (Wardle *et al.* 1999). The extensive review of Wardle (2002) showed that above–ground net primary production is not strongly or simply related to the biomass of bacteria and fungi (as primary decomposers) below-ground in all systems.

Earthworms have shown preference for certain litter types (Hendriksen 1990). Osler *et al.* (2000) showed that crop phenology e.g. leaf fall in the senescent crop phase and/or canopy structure with consequent exposure of soil, has a major effect driving the seasonal variation of soil mite communities under crops (wheat, lupin, oilseed rape) grown in rotation. Crops also differ in the mass of roots produced and in the depth and pattern of rooting (Gregory 2006). The properties of soil, particularly soil structure, are changed as a result (Angers and Caron, 1998). Roots tend to compress soil in their vicinity during radial expansion and hence decrease porosity and change pore size distribution in the rhizosphere; in the medium-term rooting increases soil macroporosity through the provision of continuous channels. Differences between crops in their impact on soil structure have been widely noted e.g. lucerne (Meek *et al.* 1989) and white clover (Mytton *et al.* 1993) have been shown to be particularly effective in improving soil structure. Differences even between cultivars have also been observed (Chantigny *et al.* 1997). These differences between crops may have short-term and/or long-term effects on below-ground ecology.

Differences between crop species and within cultivars of the same species on below-ground community structure, particularly in the short-term, are usually attributed to differences in the amount and quality of root exudates. Specifically Rumberger and Marschner (2003) showed that increased 2-phenylethylisothiocyanate concentrations in the rhizosphere of oilseed rape had a significant effect on the community structure of bacteria and overall fungal, algae and nematode community structure in the rhizosphere. The abundance and community structure of soil biomass populations are modified by plant species typical of grasslands (Griffiths *et al.* 1992, Groffman *et al.* 1996; Grayston *et al.* 1998; Bardgett *et al.* 1999; Porazinska *et al.* 2003) and cropping systems (Grayston *et al.* 1998; Porazinska *et al.* 2003). Marilley and Aragno (1999) used molecular techniques to show that the bacterial community structure associated with grass and clover roots within a sward varied markedly between species. Griffiths *et al.* (1992) also showed some indication of differences between plant species in their effect on below-ground species with *Poa annua* and *Poa pratensis* supporting larger bacterial numbers in the rhizosphere than *Lolium perenne* or *Festuca arundinacea*. There were also differences between species in their effect on protozoa, nematodes and enchytraeids. Marschner *et al.* (2001) showed that while rhizosphere communities of bacteria were largely plant species specific the development of the community was controlled by a complex interaction of soil and plant factors. The strong specificity of many microbe-plant relationships in the root, rhizoplane and rhizophere suggests that an increase in plant diversity, whether in space or time, is likely to lead to changes in the species dominance below-ground and perhaps also an increase in diversity (Lynch *et al.* 2004).

The presence of particular host crops is well known to be critical for the survival of certain root-associated species e.g. for rhizobia, AM fungi and pathogenic species. The presence of fallow periods or non-host crops for AM fungi (e.g. brassicaeous species) in a rotation significantly reduces propagule numbers, and AM colonisation of subsequent crops (Gosling *et al.* 2006). Species and even cultivars may show different root exudates or leachates that either stimulate (susceptible crops and varieties) or inhibit (resistant crops and varieties) the germination of specific pathogenic organisms (Navneet and Mehrotra 1988; Bateman and Kwasna 1999). Knowledge of the survival strategy of the particular organism in the absence of a host plant is important for rotation planning either to maintain populations (Rhizobia and AM fungi) or to break the pathogen/host cycle.

Plant productivity is coupled to below-ground ecology through amount and quality of litter or residues returned (considered in more detail in Section 3.4), root growth and exudation in the soil. Plants also compete for available nutrients, water and other resources below-ground and it is not clearly established to what extent this competition is an important regulatory mechanism in below-ground ecology (Wardle 2002). These mechanisms are difficult to separate in the field. The cultivation of different crops in arable systems is usually associated with a range of other changes in management practices, as well as differences in relation to duration of crop cover and growing season, amount and quality of OM inputs. Black (1998) could not distinguish any major differences in their measured impact on below-ground ecology of root and cereal crops in arable rotations. Differences between the composition and structure of below-ground populations in long-term pasture and arable crops are well known but relate to tillage (described above) as well as the plant factors also outlined here. Some agri-environmental schemes are encouraging the increased of arable/root crops in the uplands cultivation; inclusion of such crops may well provide alternative habitats and food sources for birds (Stevens and Bradbury 2006) but further consideration of the impacts of these crops and associated management (tillage etc) in upland soils is required.

3.3. Crop rotation

In most cropping systems mono-cropping is the exception and the majority of cropping systems include a distinct break crop to interrupt host/pathogen interactions. Crop rotation is a system where different crops are grown in a defined recurring sequence. Alternating legumes and nitrogen demanding crops can reduce the need for N fertiliser. Break crops may also be selected because of their impact on soil structure or other properties (e.g. Lucerne). However, selection of break crops is usually carried out in relation to their market value and potential yield. Any change in crop rotation is likely to also result in a number of other management changes, in addition to changes in crop order. In a long-term experiment, Houot and Chaussod (1995) showed that the effects on soil properties and below-ground ecology of changes to management practices in crop rotations can take a long time to reach equilibrium; hence studies of previous crop effects in the less-controlled “real world” are complicated by these temporal dynamics.

There are few studies of the impact of a previous crop or crops on below-ground ecology. However, some indications that these effects are significant in the field have been shown e.g. certain cultivars of red clover have been found to foster the development of endophytic bacteria that promote the growth of subsequent potato crops (Sturz and Christie 1998). It has recently been proposed that an additional benefit might be delivered by brassicaeous species (including oilseed rape) as a result of allelochemicals released into the rhizosphere during crop growth (Rumberger and Marschner 2003) and/or when the crop residues are decomposed in soil (Bending and Lincoln 1999). However, these changes may not be large enough to affect crop growth in the field. Smith *et al.* (2004) could show no evidence of benefits to following wheat crops as a result of biofumigation by brassicas or evidence of any significant changes in microbial community structure.

Perhaps not surprisingly rotation effects have been most studied in relation to the persistence and effectiveness of mycorrhizal fungi. Oehl *et al.* (2003) showed that increased cropping diversity coupled with reducing tillage within a cropping sequence (in a study using a gradient of sites from intensive mono-cropped maize to species rich grassland) led to a increase in the species richness of arbuscular mycorrhizal fungi. Mono-cropping seems to select for AM fungal species that offer limited benefits to the main crop plant (Johnson *et al.* 1992). Increasing the diversity of hosts by crop rotation generally increases the diversity of AM fungal species (Gosling *et al.* 2006), but it is also clear that non-mycorrhizal hosts within the rotation will have negative impacts.

Many of the effects of crop rotation are linked to the increased diversity (and sometimes also amount) of litter and/or crop residue return, which have a range of potential effects on below-ground ecology and are considered in the next section.

3.4. Crop residue management

Return of crop residues (in contrast to baling and removal) has been shown in some studies to make a larger contribution to the increase in size of the soil microbial biomass than decreasing intensity of tillage (Spedding *et al.* 2004). The relative magnitude of these effects is strongly dependent on the soil type (Spedding *et al.* 2004). Reductions in the microbial population density and diversity have been observed following stubble burning; this was linked to reductions in amount and availability of OM (Rasmussen and Rohde 1988). Increases in soil microbial biomass are commonly measured where residues are incorporated rather than removed or burnt (Powlson *et al.* 1987). However, increased organic matter input from plants has been linked to stimulation in the bacterial feeding microfauna (nematodes and protozoa) without a concomitant increase in the size of the bacterial population; the stimulation of the bacteria population is kept in check by grazing (Wardle 1995; an example of a tri-trophic effect within the soil food web). Christensen *et al.* (1992) showed a rapid but ephemeral (up to 20 days) increase in protozoa and bacterial feeding nematodes (to populations 80 and 30 times greater than the initial population sizes respectively) in the vicinity of a freshly-killed barley root. These increases did not stimulate larger predators, perhaps because of their short duration.

The amount and quality of crop residue returns are well known to affect mineralisation processes in soil (Swift *et al.* 1979); However, there has been much less study of the impact of residue returns on microbial community structure. Inclusion of cover crops with no other changes in the crop rotation, led to an increased size of the microbial biomass in a vegetable cropping system (Schutter and Dick 2002). There was also some evidence that inclusion of a cover crop also affected microbial community structure with an increase in the proportion of rapidly proliferating bacterial species e.g. pseudomonads (Schutter and Dick 2002). Bending *et al.* (2002) showed smaller differences than they had expected in microbial community functional diversity as a result of the addition of a range of crop residues. Residues with high lignin contents seemed to have a greater short-term influence on microbial community composition, whereas low lignin resides with a range of other characteristics showed increased population sizes but little difference in functional diversity (Bending *et al.* 2002). Bailey and Lazarovits (2003) assembled substantial evidence from the literature to show that rapidly decaying plant residues (with low C:N ratios) reduce the numbers of pathogenic species while at the same time increasing the total population of bacteria and fungi. They attributed the effect to the impact of high NH_3/NH_4^+ concentrations produced during mineralisation on pathogen populations rather than microbial competition.

It has been suggested that the microbial biomass population is adapted to "specialise" in decomposition of the dominant litter type (Cookson *et al.* 1998). However, more recent studies do not provide strong evidence to support this hypothesis (Ayres *et al.* 2006).

3.5. Herbicides

Herbicides are diverse group of chemicals developed to allow the treatment of unwanted vegetation. The most actively used herbicide formulations on arable crops in the UK in 2004 were glyphosate, isoproturon, fluroxypyr, mecoprop-P and trifluralin (Garthwaite *et al.* 2003). Many herbicides are rapidly broken down in contact with soil, and may even stimulate microbial activity in the short-term. Haney *et al.* (2002) showed that application of glyphosate (as the isopropylamine salt with associated formulants and surfactants) stimulated microbial activity in soil as well as giving short-term increases in the size of the microbial biomass. Plots, which had received applications of the herbicide linuron (for > 10 years), showed the presence of the Variovax bacterial ribotype, which has been previously associated with the degradation of this herbicide and was not present in an untreated soil (El Fantroussi *et al.* 1999).

Reviewing studies that had used typical field rates of herbicide applied to soil, Wardle (1995) could find no evidence for any detectable direct effects of herbicides on protozoa, collembolae, nematodes and earthworms. Black (1998) also found few studies which showed an impact of herbicides on soil microbial biomass and soil fauna, where negative impacts were observed these were for beetles, collembola. Some negative impacts have also been observed for protozoa (Foissner 1997). Hart and Brookes (1996) also showed no major effects of microbial biomass size or activity even after long-term application at usual rates. However, herbicides have a range of target effects on plant cover (restricting weed emergence and/or growth and stimulating crop growth) which are likely to result in a range of indirect effects on below-ground ecology e.g. by changing the amount and quality of root exudates. Earthworms seem to benefit more from weedy conditions more than other species groups (Tomlin and Fox 2003). Using molecular techniques (El Fantroussi *et al.* 1999) showed that plots with > 10 years use of the herbicide diuron showed the loss of the bacterial group Acidobacteria. Separately Seghers *et al.* (2003) showed the composition and diversity of the methanotrophic bacteria in soil was reduced following long term application of herbicides (20 years use of atrazine and metolachlor at typical field rates); the community structure of endophytic bacteria was also altered (Seghers *et al.* 2004). In both studies it is not possible to distinguish whether this is a direct effect of the herbicide or an indirect effect due to impact on ground cover. Manipulation of ground cover and removal of weed species may be of particular importance for rhizobial and AM fungi, where weed species may act as hosts; weeds are particularly important as bridges if the main crop is not a host species (Kurle and Pfleger 1996).

3.6. Pesticides

Pesticides are a diverse group of chemicals used to control insects and other organisms harmful to cultivated plants and animals. Collated results from the literature (Black 1998) show that non-target impacts of pesticides were greatest for soil fauna in arable systems (reflecting the much lower use of pesticides in grassland). The highest proportion of negative effects were seen for macrofauna - earthworms, beetles, collembola. Hart and Brookes (1996) showed little evidence of long-term harmful effects of the use of typical range of agricultural pesticides, singly or in combination, on the soil microbial biomass or its activity. (Hart and Brookes 1996). Studies of pesticide impacts usually consider applications of single components rather than the full diverse programme of an in-field pesticide regime; the majority of studies are carried out under controlled rather than field conditions. The timing of an application in relation to the life cycle of fauna is also critical in determining the impact on target and non-target species (Frampton and Çilgi 1996). Little is known about the impact of the formulation ingredients of pesticides e.g. adjuvants (dos Satos *et al.* 2005). It is therefore difficult to assess the likely impact of field use pesticides on below-ground ecology (Gosling *et al.* 2006).

3.6.1. Insecticides

The most extensively used insecticides in the UK belong to the pyrethroids, accounting for 88% of the insecticide-treated arable area (Garthwaite *et al.* 2003). Pyrethroid-neonicotinoid co-formulations, carbamates and organophosphates are also extensively used.

Of the 128 papers identified which used soil and pyrethroid as a key word none reported the impacts of pyrethroid insecticides on below-ground ecology under temperate conditions; 2 papers were identified which showed changes in microbial community composition under rice. Organophosphate insecticides have been shown to have negative impacts on collembola (Endlweber *et al.* 2005) with differences between the effects of different insecticides (chlorpyrifos had greater effects than dimethoate in field at typical application rates). Populations of collembola showed some recovery with time, but one year after application treated plots had smaller populations than the control plots (Endlweber *et al.* 2005). Insecticides showed differential effects on different collembolan species; while the overall species diversity was not affected, the dominance patterns between species were changed. Organophosphate insecticides have also shown negative impacts on earthworms (Panda and Sahu 2004) and led to changes in bacterial and fungal numbers (Pandey and Sigh 2004). Carbamate insecticides also had negative impacts on earthworm populations (Ribera *et al.* 2001). Foissner (1997) reviewed a range of data mostly collected in laboratory microcosms systems and showed that insecticides often disturb soil protozoa critically for periods of up to 2 months, but under field conditions populations then recover; in many cases fluctuations in other variables such as food resources and /or temperature had bigger effects than pesticide use.

3.6.2. Fungicides

Fungicides are applied to control fungal disease on crops. The most extensively used fungicide formulations used in arable crops in the UK are chlorothlonil (all crops except rye), epoxiconazole (cereals), azoxystrobin (all except triticale and sugarbeet), epoxiconazole/fenpropimorph/kresoxim-methyl (all cereals) and trifloxystrobin (wheat and barley) (Garthwaite *et al.* 2003). When applied at recommended rates to plants, few fungicides have been seen to have significant effects on mycorrhizal colonisation (Gosling *et al.* 2006). Where effects are seen these are often short-term e.g. Smith *et al.* (2000).

Long-term negative effects are seen where copper-based fungicides have been used for a number of years due to the accumulation in the soil of Cu to levels which are toxic. Most effects in the field are seen in orchards and vineyards where negative effects on earthworms have been recorded (Filser *et al.* 1995; Van Zweiten *et al.* 2004; Eijsackers *et al.* 2005; Loureiro *et al.* 2005). Use of Cu-based fungicides was also shown to lead to increased stress responses in microbial populations (Merrington *et al.*, 2002).

3.6.3. Fumigants

Fumigation of soils to control soilborne pathogens, nematodes and weeds is a tool associated with the intensive cultivation of some vegetable, fruit and nursery crops. The most common fumigant in recent times has been methyl bromide often applied in combination with chloropicin. However, the use of methyl bromide is being phased out, by international treaty as a result of its greenhouse gas potential, and alternative fumigants have been developed. Differences are often observed in the short-term response of soil organisms to fumigation since soil temperature and moisture content affect the efficacy of fumigation. However, in general, all fumigants show an immediate reduction (up to 1 week) of soil microbial activity (respiration and enzyme activities), but after 30 weeks there is little difference between fumigated and unfumigated soils (e.g. Klose and Ajwa 2004). However, under field conditions after multiple applications methyl bromide can be seen to have a significant negative impact on the enzymatic processes of the soil microbial biomass (Klose and Ajwa 2004). It has therefore been hypothesised that repeated fumigation may lead to long term adaptation of the microbial population with loss of sensitive species and selection for resistant species. Miller *et al.* (1997) showed an increase in the population of microbes able to use fumigants as a source of C and/or energy. Initial work suggests that the alternative fumigants developed to replace methyl bromide have smaller effects (Klose and Ajwa 2004). However, it should not really be a surprise that a blunt management technique such as fumigation should have significant effects on below-ground ecology.

3.7. Grazing livestock (intensity and species, use of vet medicines)

Grazing i.e. the above ground defoliation of grass and forb species by herbivores consumes up to half the annual above ground net primary productivity. Defoliation has been shown to reduce the amount of root exudation with consequent reductions in the activity of the cultivable soil bacterial population (Macdonald *et al.* 2004). Other studies have shown increases in exudation following defoliation (e.g. Hamilton and Frank 2001). Other impacts of the livestock, particularly the returns of dung and urine to the soil surface, confound the direct impact of defoliation within grazing management. Supplementary feeding of livestock during the grazing period may also increase inputs of C, N, P and other nutrients to the below-ground ecosystem via excreta significantly. These combined effects therefore mean that grazing affects the amount and quality of C (and other nutrient) input to the soil in quite a complex way (Bardgett *et al.* 1997) and often increases the size and activity, particularly of bacteria in soil. Clegg (2006) compared grazed and ungrazed swards and showed no differences in community profiles of fungi but modification as a result of grazing to the structure of the pseudomonad community; these are heterotrophic bacteria which responding rapidly to soluble C inputs, some ribotypes are associated with denitrification. Neilson *et al.* (2002) showed that grazing had a significant impact on trophic interactions below ground and consequently on C and N cycling.

Relationships between stocking density and below-ground ecology depend on the typical stocking density of the system. At very low levels of stocking density (such as seen in very sparsely grazed upland grasslands) an increase in stocking density leads to an increase in the microbial biomass, particularly the bacterial population (Yeates *et al.* 1997). Other soil fauna also show increases which have been strongly linked to the increase in carbon inputs and nutrient availability (Bardgett *et al.* 1993; Bardgett *et al.* 1997). Early results of the GRUB project suggest that increased grazing intensity significantly reduced the population of spiders, though not other insects (Macaulay Institute 2006).

High stocking rates such as typically seen in lowland grassland have a negative impact. Mulder *et al.* (2003) have shown a decline in populations (presence and abundance) of most nematode species in grassland with increasing livestock units (measure of intensification), but two species showed a reverse trend and increase with increasing livestock density (Mulder *et al.* 2003). Functional diversity of both bacterial and fungal feeding nematodes also decreases with increasing grazing pressure (Mulder *et al.* 2003). Overstocking has negative impacts, which probably arise due to increased compaction, poaching, disruption of sward and an increased proportion of bare ground in overstocked swards. For example, where soil bulk density increases as a result of compaction, AM fungal colonization has been shown to decrease (e.g. Entry *et al.* 1996). Increased urine returns in overgrazing situations may interact with poaching to exacerbate the impacts on below ground ecology and soil processes. Urine stimulates soil microbial turnover (Petersen *et al.* 2004) and in a study of the effects of excretal returns and soil compaction on nitrous oxide emissions Simek *et al.* (2006) has recently shown very high microbial biomass and pH associated with the areas of most severe compaction. Research in Ireland has shown a negative correlation between % cover of bare ground (caused by poaching) and carabid species richness (Ni Bhriain *et al.* 2002).

Veterinary medicines include a variety of nematicides, hormones and anti-microbials, which may impact on below-ground ecology as a result of deposition in grazing excreta or through application of manures. Direct application of anti-microbials and nematicides usually used as veterinary medicines to soil has a negative impact on soil microbial populations and impacts below-ground food webs (Westergaard *et al.* 2001; Svendsen *et al.* 2005; Jensen *et al.* 2003). There is some evidence of reduced numbers and activity of dung beetles where veterinary drugs are used regularly (Hutton and Giller 2003), retarded decomposition rates of dung are likely to have an impacts on other species.

In order to control ectoparasites in sheep, UK farmers use organophosphate or synthetic pyrethroid-based formulations resulting in around 200 million litres of spent sheep dip produced each year. Use and disposal of sheep dip may have significant ecological impacts in surface waters and soils. Boucard *et al.* (2004) showed that synthetic pyrethroid sheep dip was less toxic to protozoa than an organophosphate sheep dip. In both cases amoebic cysts remained viable and emerged from dormancy, which suggests the potential for recovery of protozoan communities in contaminated environments. Roychowdhury *et al.* (1999) showed changes in the microbial community structure with a reduction of actinomycete populations using the synthetic pyrethroid. Aitken *et al.* (2004) found no consistent effects of sheep dip on microbial biomass or respiration rate but indicated that here might be some effect on functional diversity.

Table 3.4 Summary of grazing impacts on below-ground organisms

Species/group	Average impact of increased grazing intensity	Key references
Bacteria and archaea	Significant increase (in upland) due to stimulation of exudation and increased nutrient returns through excretion	Yeates *et al.* 1997
Fungi	No impact of grazing	Clegg 2006
Arbuscular mycorrhizal species	Reduced if overgrazing causes compaction	Entry *et al.* 1996
Nematodes	Significant reduction in lowland	Mulder *et al.* 2003
	Significant increase in upland	Bardgett *et al.* 1997a
Mites	Significant increase in upland	Bardgett *et al.* 1993a
Collembola	Significant reduction in lowland	King and Hutchinson 1976
	Significant increase in upland	Bardgett *et al.* 1993a
Earthworms	Little to no impact, slight increase with grazing intensity	Hutchinson and King 1980; Muldowney *et al.* 2003
Insects	No impact or reduced by grazing	Ni Bhriain *et al.* 2002; Macaulay Institute 2006

3.8. Lime

The use of lime to maintain soil pH where soils are non-calcareous, is common practice in arable cultivation; it is uncommon to find arable land in UK routinely below pH 6. Liming of grassland, especially in the uplands, is less common. 8.5% of all tilled land and 3.7% of all grassland received lime in 2003 at 5.4 and 4.1 tonnes/ha of CaO equivalent (Defra 2004b). Consequently > 95% studies of studies reported in the literature have shown positive impacts of liming pasture on below-ground ecology, particular increasing populations and activity of earthworms (Black 1998). Liming acid soils also tends to increase mycorrhizal population density (Hamel *et al.* 1994) and to increase AM fingal colonisation of plant roots in a bioassay (Johnson *et al.*, 2005). In contrast, hyphal feeding nematodes show a positive correlation with increasing soil acidity (Mulder *et al.* 2003). The fungally-mediated channel of decomposition increases in importance with increasing acidity (decreasing pH). It is well documented that acid soils tend to show decreased faunal activity (hence reduced comminution and mixing of OM) and reduced microbial decomposition leading to the development of a mor humus form in uncultivated soils, where below-ground fauna are dominated by mites, enchytraeids and collembola (Killham, 1994).

3.9. Mineral fertilisers

Mineral fertilisers are a major input into UK agriculture to meet plant nutrient demand and maintain a balanced nutrient budget. Nitrogen use is lower on grassland than crops (Table 3.5) with the highest application rates being to winter wheat and oilseed rape (Table 3.6). On average higher rates of phosphate and potash are also applied to crops than grassland. Potatoes receive the highest phosphate and potash, although sugar beet also has a high potash requirement. In relation to grassland, management is clearly important in determining fertilizer application rate and rates vary from 0 in upland situations to between 61 and 164 kg/ha for all grazed and conserved only grassland respectively (Defra 2004b). There is a range of fertilisers in common use – ammonium nitrate (as solid and liquid, with additional urea), diammonium phosphate, triple superphosphate, muriate of potash (dominantly potassium chloride), also as compound fertilisers.

Table 3.5 Overall nitrogen, phosphate and potash use (kg/ha), Great Britain (1999-2003). (Defra 2004b)

	Nitrogen (kg/ha)		Phosphate (kg/ha)		Potash (kg/ha)	
	Tillage crops	Grass	Tillage crops	Grass	Tillage crops	Grass
1999	141	110	45	20	57	28
2000	149	99	47	20	55	26
2001	145	94	43	19	52	24
2002	152	89	44	20	57	25
2003	149	83	40	18	54	22

Table 3.6 Overall fertiliser use (kg/ha) on major tillage crops, Great Britain (2003). (Defra 2004b)

	Winter wheat	Spring barley	Winter barley	Maincrop potatoes[1]	Oilseed rape[2]	Sugar beet
Nitrogen	197	107	148	152	191	103
Phosphate	39	44	41	130	38	34
Potash	47	57	59	214	42	91

1. Includes second earlies
2. Combines winter and spring rape

It is often considered that there is a direct effect of high levels of soluble P in soil on colonisation of roots and propagule density of AM fungi. However, a small number of studies have reported contradictory results. Harrier and Watson (2003) suggest that the effect of soluble P fertilisers on the AM fungi-crop relationship is affected by the P status of both crop and soil. Different isolates of AM fungi differ in their sensitivity to soil P and consequently at low and moderate levels of soil P impacts of P fertiliser on colonisation and the effectiveness of the AM-root association may vary depending on the isolate involved (Harrier and Watson 2003). The literature also provides mixed evidence of the effects of N fertiliser on AM-root associations (Gosling *et al.* 2006). The use of rock phosphates (which are a very slow release source of P) has no effect on AM fungi (Ryan *et al.* 1994). The reduced effectiveness of nodulation and N fixation where large soil nitrate concentrations are present in soils were established early in the study of N fixation (Nobbe and Richter 1902) but soils receiving high applications of N fertiliser can still support large populations of rhizobia.

Application of some N and S fertilisers (particularly ammonium sulphate) is known to reduce soil pH. Sarathchandra *et al.* (2001) measured changes in nematode species composition resulting from pH changes after fertilisation. Earthworm populations are also reduced with increasing acidity (Edwards 1998). Where long-term acidification from fertilisation of grassland is not remedied development of a mor humus form will result due to the reduction in comminution and decomposition of plant litter (Thurston *et al.* 1976; Shiel 1986; van Bergen *et al.* 1998). P fertilisers often contain trace heavy metal contaminants (Cd, Hg, Pb; McLaughlin *et al.* 2000); where P fertilisers have been used regularly long-term chronic toxicity might arise. However, this is more often a problem with contaminated organic amendments (Giller *et al.* 1998).

Within fertiliser studies it is almost impossible to separate any direct effects on below-ground ecology from feedbacks as a result of plant nutrition (Dick 1992). Marschner *et al.* (2004) showed that impacts of P fertilisation were mainly mediated via changes in the amount and composition of root exudates. Increased N fertilisation (removing N deficiency) has been shown to reduce the proportion of plant assimilated C that is directed to root exudation in grassland e.g. Paterson and Sim 1999, which may lead to a larger more active soil microbial community in N-limited grassland (Yeates *et al.* 1997). Donnison *et al.* (2000) measured different responses of different soil fungi to the application of NPK fertilisers to grassland systems. Consequently it is not surprising that a range of effects on microarthropods and nematodes have been shown in grasslands as a result of fertiliser application (Bardgett and Cook 1998).

Long-term fertiliser treatments leading to consistent differences in yields (and residue returns) are usually associated with increases in SOM and microbial biomass (Marschner *et al.* 2004; Murphy *et al.* 2003) particularly where crop residues are returned. Changes in the size of the microbial biomass pool may also be detectable ahead of changes in SOM when practices are changed (Powlson *et al.* 1987). Su *et al.* (2004) showed that in arable plots which had received inorganic fertilisers (including N, P and K together and separately) for over 100 years had developed of different microbial communities where inorganic fertiliser rather than no addition or manure is used. In contrast, plots of wheat which have received long-term application of ammonium nitrate fertiliser showed only small differences in overall microbial population diversity (Lawlor *et al.* 2000). Peacock *et al.* (2001) also observed a significant change in the microbial community after 5 years application of ammonium nitrate fertiliser at typical farm rates with increased proportion of Gram +ve bacteria compared to the control. In this experiment soil pH was significantly lower in the fertilised plot which may have been a major driver in community change. Use of the nitrification inhibitor (DCD) showed no impact on the size of the total microbial biomass (Di and Cameron 2004). In the same wheat plots studied by Lawlor *et al.* (2000), Mendum and Hirsch (2002) found different dominant populations of autotrophic ammonia-oxidising bacteria (Nitrosospira, first step nitrifiers) than plots receiving no nitrogen fertiliser. The dominant populations in the low and high NH_4 conditions match the physiological distinctions observed in enrichment cultures (Kowalchuk *et al.* 2000). Changes in the nitrifier populations in response to fertilisation are relatively persistent (up to 1 season; Mendum and Hirsch, 2002; Okano *et al.* 2004) but long-term effects of withdrawal of fertiliser have not been studied. Black (1998; Figure3.2) found a large number of studies of showing impacts of fertiliser on earthworms, these were generally positive, with <8% of results showing negative impact on earthworms. There have been fewer studies on other insects. Nematodes and protozoa tend to show positive responses to fertiliser application, but little impact of application of fertiliser on soil microbial biomass, possibly a further example of a tri-trophic effect within the soil food web.

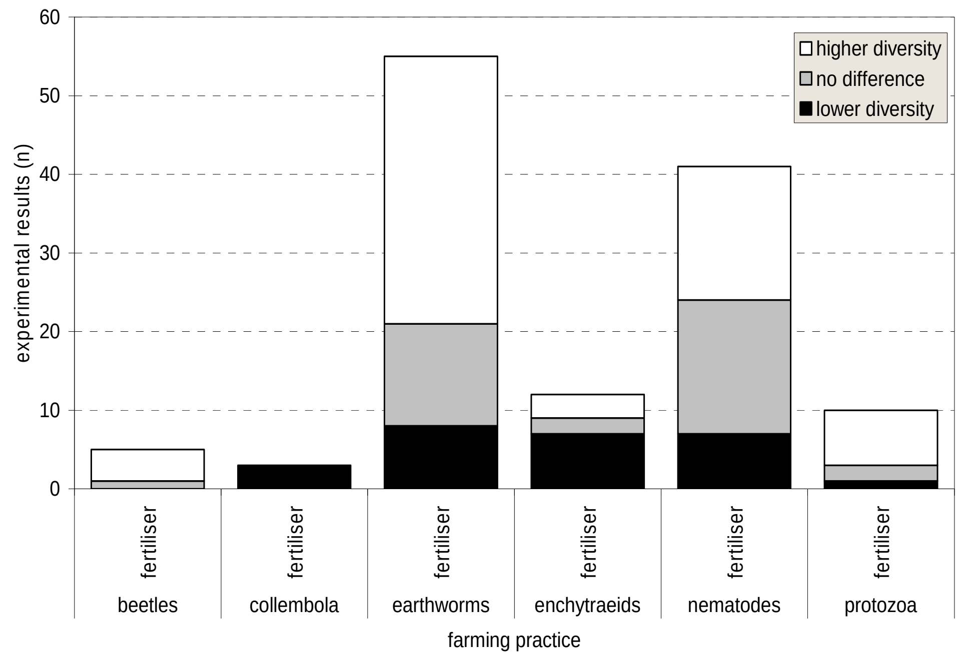


Figure 3.2 Frequency histogram of impacts of fertiliser on soil biodiversity (adapted from Black 1998). Number of studies that show higher diversity (□), no difference (■) and lower diversity (■) for a range of soil organisms

3.10. Organic amendments

Organic amendments used in agriculture include a diverse range of materials produced on and off-farm, here defined as including microbial, plant, and animal wastes, including by-products of the food processing industry. The most common wastes used on agricultural land within this category are farmyard manure and slurry; but increasing production of green waste composts and use on agricultural land. Organic amendments have been used on agricultural land partly to facilitate their disposal, but also to help meet plant nutrient demand and/or as soil conditioner. The variation in nutrient contents and nutrient availabilities in a range of animal manures is shown in Tables 3.7 and 3.8

Table 3.7. Typical nutrient content of animal manures (Anon 2000)

	DM	**Total** Nutrients			**Available** Nutrients (1)		
	%	Nitrogen	Phosphate	Potash	Nitrogen	Phosphate	Potash
Fresh FYM (2)			**kg/t**			**kg/t**	
Cattle	25	6.0	3.5	8.0	see Table 3.8	2.1	7.2
Pig	25	7.0	7.0	5.0		4.2	4.5
Poultry Manures			**kg/t**			**kg/t**	
Layer manure	30	16.0	13.0	9.0	see Table 3.8	7.8	8.1
Broiler/turkey litter	60	30.0	25.0	18.0		15.0	16.0
Slurries			**kg/m^3**			**kg/m^3**	
Dairy (3)	6	3.0	1.2	3.5	see Table 3.8	0.6	3.2
Beef (3)	6	2.3	1.2	2.7		0.6	2.4
Pig (3)	4	4.0	2.0	2.5		1.0	2.3
Separated cattle slurries (liquid)			**kg/m^3**			**kg/m^3**	
Strainer box	1.5	1.5	0.3	2.2	see Table 3.8	0.15	2.0
Weeping wall	3	2.0	0.5	3.0		0.25	2.7
Mechanical separator	4	3.0	1.2	3.5		0.60	3.2

(1) Nutrients that are available to the next crop
(2) Nitrogen and potash values will be lower if FYM is stored in the open or for long periods
(3) Adjust nutrient content if % DM is higher or lower

Table 3.8. Percentage of total nitrogen available to the next crop following applications of animal manures (% of total nitrogen) (Anon 2000)

Timing		Autumn (Aug-Oct)		Winter (Nov-Jan)		Spring (Feb-Apr)	Summer use on grassland
Soil type	DM(%)	Sandy/ shallow	Medium/ heavy	Sandy/ shallow	Medium/ heavy	All soils	All soils
Surface application							
Fresh FYM	25	5	10	10	15	20	n/a
Layer manure	30	10	20	15	30	35	n/a
Broiler/turkey litter	60	10	20	15	25	30	n/a
Dairy/beef slurries	6	5	15	20	30	35	20
Pig slurries	4	5	20	25	40	50	30
Separated slurries	1-4	5	20	25	40	50	35
Soil incorporation							
Fresh FYM	25	5	10	15	20	25	n/a
Layer manure	30	10	25	20	40	50	n/a
Broiler/turkey litter	60	10	25	20	40	45	n/a
Dairy/beef slurries	6	5	20	20	35	45	n/a
Pig slurry	4	5	20	20	45	55	n/a
Separated slurries	1-4	5	20	25	45	55	n/a

Amendment of soil with raw and composted organic amendments generally leads to an increase in the soil microbial biomass population e.g. Marschner *et al.* (2003). The duration of this effect depends on the amount and quality of OM added; sustained changes are most likely where organic amendment is regular. Black (1998) showed generally positive responses to the application of organic amendments for most species groups in both arable and pasture, where the amendment was FYM in most studies. There was little indication of changes for mites and beetles.

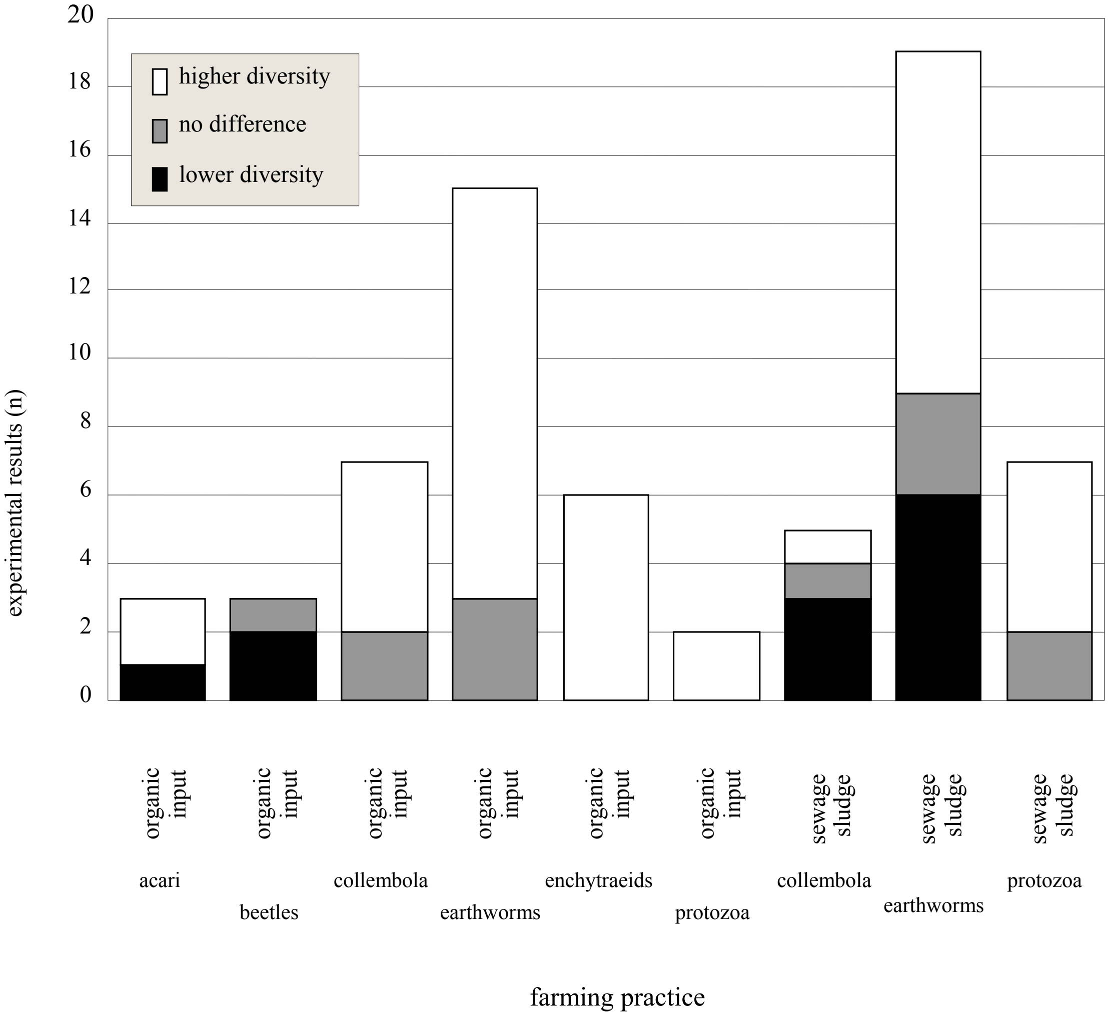


Figure 3.3. Frequency histogram of impacts of organic amendments on soil biodiversity (adapted from Black 1998). Number of studies that show higher diversity (□), no difference (■) and lower diversity (■) for a range of soil organisms.

Following more than 100 years of manure application Sun *et al.* (2004) showed increased soil bacterial diversity with increased evenness (reduction in the importance of the most dominant species) in comparison with plots receiving no additions. In a long-term trial in Sweden, Sessitch *et al.* (2001) also observed significant differences in the size and diversity of the bacterial population between plots receiving mineral fertiliser and those receiving green manure or well rotted farmyard manure. Increases in the size and changes in the structure of the bacterial community (but not that of the fungal community) occurred after long-term (> 30 years) low rate application of crop residues, additional straw, farmyard manure, sewage sludge (Marschner *et al.* 2003). Differences between the treatments were attributed to differences in the amount and composition of SOM and thus substrate availability associated with the organic amendments. Separately long-term amendment of plots with sewage sludge (metal contaminated) or peat also led to very distinct bacterial communities (Sessitch *et al.* 2001). Crecchio *et al.* (2004) demonstrated clearly using a range of molecular techniques that while application of municipal waste compost (for 6 years) slightly increased the size and activity of the bacterial community over this time period, its genetic diversity was not affected; well matured composts show OM quality very similar to that of humified soil OM. Zaller and Kopke (2004) showed significant increases in the microbial population size and activity after 9 years application of composted manure in an arable rotation; they also showed some smaller, but significant, differences between the impact of manures depending on whether and how biodynamic treatments had been used during the composting process. There is some evidence that these preparations affect the microbial community, which develops in the manure during the composting process (Carpenter-Boggs *et al.* 2000). However, few differences in the quality of the composted manures could be measured at application (Zaller and Kopke 2004).

Populations of protozoa, bacterivorous and fungivorous nematodes tend to show short-term increase after organic amendments, particularly where the amendments have low C:N ratio (e.g. Bongers and Ferris 1999; Griffiths *et al.* 1994; Porazinska *et al.* 1999). Populations of protozoa tend to increase more quickly and peak much earlier than nematode populations (Opperman *et al.* 1989). In contrast the use of organic amendments tends to reduce the numbers of plant feeding nematode species (Griffiths *et al.* 1994). Increased populations of bacterivorous nematodes can be linked directly to increased populations of bacteria associated with the input of organic amendments (Griffiths *et al.* 1998; Bulluck and Ristaino 2002). Long-term application of organic amendments has also been shown to increase nematode populations as a result of the increase in soil OM and soil microbial biomass (Corbett *et al.* 1969). Differences between the impacts of organic amendments are seen and this is most strongly related to the proportion of C in the added material that is readily available for microbial utilisation (Griffiths *et al.* 1998). Application of composted FYM without additional fertiliser for 9 years in an arable rotation showed a significant increase in earthworm casting activity (120% of untreated plots) but there were no significant differences in earthworm population size and no significant differences in species richness (Zaller and Köpke 2004). In a comparison of surface applied and slit injection of slurry on 12 farms in the Netherlands, slit injection negatively affected epigeic earthworms whereas its effect on anecic and endogeic earthworms was absent or even positive (De Goede *et al.* 2003).

The use of composted green waste as mulch (3 years) also significantly increased bacterial population compared to plots with wood chip as mulch or unmulched plots (Tiquia *et al.* 2002). However, molecular analysis of genetic diversity associated with roots of cucumber seedlings showed strong similarity between plots; some unique peaks suggested some differences in population under compost mulch. Surface applications of dairy manure (Peacock et al. 2001) increased the number and proportion of Gram –ve bacteria, able to respond rapidly to added soluble organic C. The largest effects were seen in the surface (0-5 cm) but changes in the microbial population were also seen at lower depths, probably due to increased leaching of soluble C and other nutrients (Peacock et al. 2001).

Bailey and Lazarovits (2003) showed that application of organic amendments that are rich in N may reduce soil-borne diseases. Bulluck *et al.* (2002) showed that numbers of beneficial soil micro-organisms (*Trichoderma* species, thermophilic bacteria) were increased and pathogenic micro-organisms (*Phytophthora* and *Pythium* species) were reduced in soils receiving organic amendments even in the first season of application. The most likely mechanism is the release of NH_4 during decomposition, but a range of other allelopathic and competition effects may also play a role particularly in soils which receive regular organic amendments (Bailey and Lazarovits, 2003).

The use of organic amendments including composts and FYM seems to have no negative and often a positive effect on AM fungi (Harrier and Watson 2003); but care is needed where amendments have high concentrations of soluble P. Few studies have been conducted on the effects of organic amendments on symbiotic N fixation and the effects are variable depending on host plant, type of amendment and environmental conditions. The application of composts may affect the population structure of the indigenous rhizobial population (Cousin *et al.* 2002). Heavy metal contamination in sewage sludge can reduce N fixation in clover due to negative impacts on numbers (Giller *et al.* 1998) and diversity of rhizobia populations (Hirsch *et al.* 1993). Increased heavy metal concentrations in soil have also been shown to reduce the size of the total microbial biomass (Brookes and McGrath 1984). Abaye *et al.* (2005) showed significant differences in the bacterial population in soils which had been contaminated with metals as a result of regular sewage sludge additions even 40 years after the application of sludge had ceased. Applications of sewage sludge can also increase soil concentrations of persistent organic pollutants which can show negative effects on below-ground ecology (Wilson *et al.* 1997. Hill 2005); there is generally insufficient data currently available to carry out appropriate risk assessments for this practice. Differences between soil microbial populations have also been seen even at relatively low rates of sludge application (Banerjee *et al.* 1997). In general long-term chronic toxicity of heavy metals and persistent organic pollutants is more common than immediate, acute toxicity and more often associated with contaminated organic amendments from urban or industrial sources (Giller *et al.* 1998).

Microbial inoculants are used in very limited circumstances in the UK. McInnes and Haq (2003) reviewed the factors affecting the establishment and proliferation of rhizobial inoculants; but in general microbial inoculants seem to have little impact on soil populations.

3.11. Drainage and irrigation

The main effect of irrigation and drainage on below-ground ecology is indirect. By regulating the seasonal effects of rainfall patterns, irrigation and/or drainage tend to stabilise the soil moisture regime away from extremes. Irrigation in Mediterranean climates is shown to reduce the disturbance effect of soil drying and increased the length of time during which the microbial biomass is active in soil; however in intensive cropping systems losses through leaching can also be increased significantly (Jackson *et al.* 1994). Irrigation can also support the persistence of soil fauna e.g. nematodes (Ferris *et al.* 2004). Soil drainage is more common in the UK. Drained soils tend to be better aerated and have a longer duration of microbial activity in the year, including nitrification, which is often strongly stimulated by drainage (Murphy *et al.* 2003).

3.12. Managing below-ground biodiversity in agriculture

Very few agricultural management practices have simple and/or generalisable impacts of agricultural management practices on below-ground ecology. The central role of decomposition and soil structural development and stabilisation processes (Section 2.5) in controlling the processes in soil which together lead to the key soil functions means that practices which impact on these interactions will have the largest effect on soil function. Modification of the inputs of OM to soil either through crop choice, rotation or amendment therefore has potentially large impacts. Tillage which intentionally manipulates soil structure also has major impacts. The impacts of increase grazing intensity are mainly mediated through a series of complex interactions between changes in amount and quality of C inputs and modification to soil structure by compaction. Other amendments to soil (fertiliser, herbicides, pesticides, lime etc) have far smaller impacts. The range of observed responses of below-ground systems is not surprising given the range of situations and starting points to which management practices are applied, and the range of direct and indirect effects that are associated with the management practices (Table 3.9).

Consequently there are no specific and practical management steps identified for farmers even on a region by region or system by system basis which might allow the reliable manipulation of below ground populations and habitats through changes in agricultural practices. Some guidance where inoculants of N fixing bacteria or biocontrol agents are used to indicate practices that are likely to support their effectiveness and persistence. Very occasionally proposals are made for the targeted and practical management of the soil food web. For example Ferris *et al.* (2004) demonstrated in California how the combined use of use of irrigation and the provision of a carbon source (cover crops and straw incorporation) within a modified agricultural system could support the persistence of the nematode population through late summer in a Mediterranean climate was able to increase microbial activity and N availability into the following spring to the direct benefit of the subsequent summer tomato crop. The combination of management practices within agricultural systems is considered in the following section.

Table 3.9 Summary of direct impacts of agricultural management practice on the soil population and indirect impacts as a result of impacts of soil habitats

	Direct effects on inhabitants	Indirect effects - effects on structure, composition and flows within the habitat mosaic							
Practice		Roots	Root surfaces	Rhizosphere	Organic residues	Chemical environment	Transmission pores	Storage pores	Residual pores
Tillage	Kills soil macrofauna, earthworms and beetles	Destroys/ damages root systems		Stimulates mineralisation	Mixes/blends But can slow decomposition rate	Aerates and allows oxidation	Reduces connectivity to depth, may decrease	Increases	Changes distribution
Rotation of a variety of crops		Diversity of structure and depth	Increase variety in space and time	Increase diversity of inputs in space/ time	Increase variety of materials. May lead to allelopathic effects.	Variety of nutrient uptake demand patterns in time and space	Changes will vary throughout the rotation may increase or decrease particular pore types. Inclusion of deep rooting crops will increase these pore sizes at depth.		
Grass/clover mixture	Habitat for rhizobium population to develop	Reduced biomass compared with grass only	Reduced area compared with grass only, nodules create different habitats	Reduced area compared with grass only. Different bacterial communities observed with grass than clover roots	Lower C:N than grass only	Legume root activity is more acidifying compared with grass only	Increase pore numbers and connectivity with clover compared to grass only		
Crop residues	Rapid decomposition can control some pathogens			Stimulate/ reduce mineralization depending on C:N ratio	Increase Location within soil depends on method of incorporation	Rapid decomposition can lead to development of anaerobic microsites and high N availability		May stimulate aggregation	

	Direct effects on inhabitants	Indirect effects - effects on composition, structure and flows within the habitat mosaic							
Practice		Roots	Root surfaces	Rhizosphere	Organic residues	Chemical environment	Transmission pores	Storage pores	Residual pores
Herbicides		Kills roots	Rapid change in chemical properties	Increases dead root materials	Increase		May increase formation due to higher turnover of roots		
Insecticide	Kills insects	Increases life span	May increases surface area	May change exudation patterns	Reduced input of OM to soil as dead roots				
Fungicide	Cu-based fungicides accumulate and have toxic effects				May change quality of residues returned	Accumulation of Cu in soil where Cu-based fungicides used			
Increasing grazing intensity		Fertiliser effect stimulates growth	Increase Where compaction occurs, change in root morphology	Defoliation stimulates exudation	Increased excretal returns	Hotspots of N, P, K associated with excreta	Reduce if compaction occurs	Distribution change if compaction occurs	Increase if compaction occurs
							Development of platy aggregates if compaction occurs, reduced pore connectivity		
Lime					Potential increase	Increase pH with subsequent effect on element availability	Improve structural stability in some soils		
Drainage	Installation kills larger organisms.	Deeper and more biomass in drained soils	Increase	Increase	More rapidly decomposed	Increase aeration Big +ve impact on nitrifiers	Depending on drainage method may modify pore size distribution and connectivity		

Tab;e 3.9. Continued

Practice	Direct effects on inhabitants	Roots	Root surfaces	Rhizosphere	Organic residues	Chemical environment	Transmission pores	Storage pores	Residual pores
Fertiliser	High soluble P restricts AM fungi	Increase surface area	Increase	Increase exudation	Increase in longer term as crop residues will increase	May decrease pH (particularly NH_4, S-based fertilisers) High short-term levels of soluble nutrients following application			
FYM		Fertiliser effect stimulates growth	Increase	Increase volume	Increase Stimulate/ reduce mineralization depending on C:N ratio	Usually raises pH Increase N,P, K availability. Medium term availability	Stimulates structural formation processes after disturbance. Improve structural stability in some soils		
Slurry	High NH_4 levels can control some pathogens	Fertiliser effect stimulates growth	Increase		Increase	Increase N,P,K availability Short to medium term availability			
Compost		Improved rooting distribution	Increase		Increase Usually little impact on mineralisation depending on C:N ratio	Raise or lower pH Increase P, K availability	Stimulates structural formation processes after disturbance. Improve structural stability in some soils. Tends to increase stability of transmission and structural pores and/or increase water holding capacity depending on soil type.		
Sludge	May be toxicity effect after number of applications	May be a fertiliser effect to stimulate growth			Increase	Possible toxicity of metals and persistent organics			

4. Agricultural systems

In the UK, most agricultural land is under grass (Figure 4.1), with over half of that grassland largely unimproved for agriculture (Figure4.2). England has a much larger proportion of cropping land, dominated by the cropping of wheat, barley and oilseed rape (OSR). In contrast rough upland grazing is much more significant in Wales and Scotland. Lowlands on the western side of the UK are well suited climatically to grass production and consequently often associated with intensive grasslands used to support dairy production. Agricultural systems in the UK can therefore be roughly divided into upland livestock systems, usually producing beef and sheep, intensive lowland grassland systems, usually dairy systems, and lowland cropping systems, which include small areas of very intensive horticultural cultivation. Other systems can be identified with some increasing in importance e.g. livery and use of grassland for horse grazing. However, this is beyond the scope of this review as it has not been possible to consider all systems here. Agricultural systems represent integrated collections of structures and practices set within their local environmental constraints; however, particular practices (discussed individually in Section 3) are more associated with certain farming systems than others (Figures 4.5 to 4.9). These differences in practices are also likely to lead to quantitatively different characteristics at landscape, farm and field scales e.g. higher weed incidence in organic arable and horticulture systems, hedge lengths per unit area of land (Fuller *et al.* 2005).

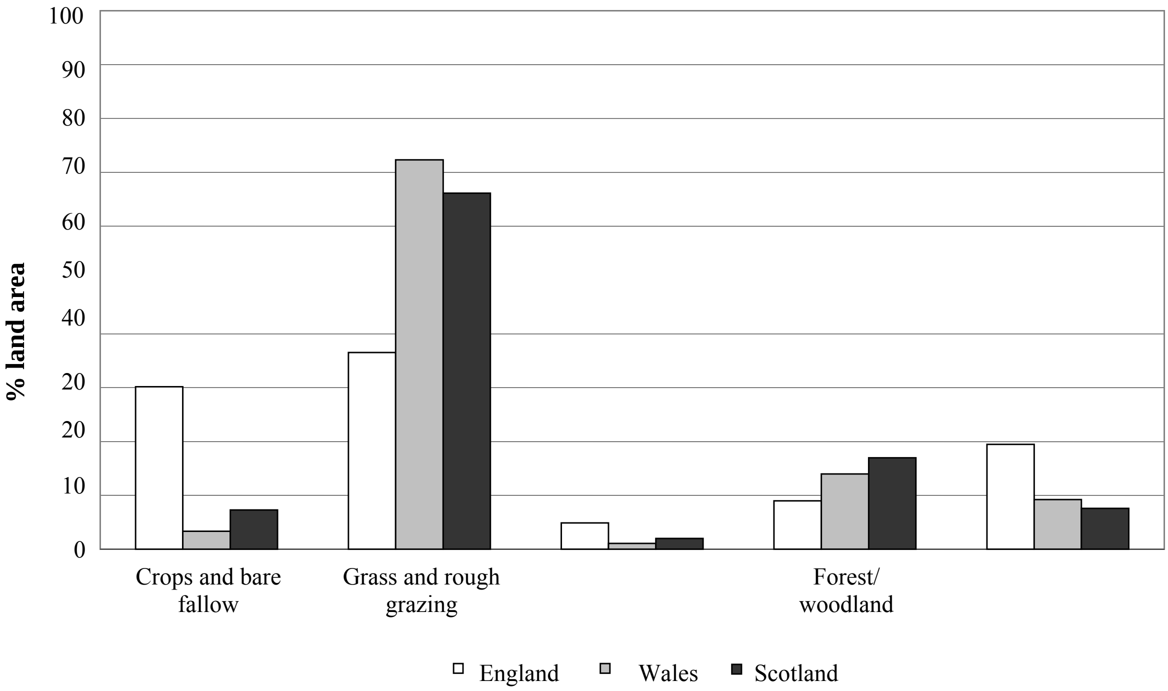


Figure 4.1. Land by agricultural and other uses in England, Wales and Scotland Source: Defra; Ordnance Survey; Forestry Commission; Forest Service (Accessed 1st March 2006).

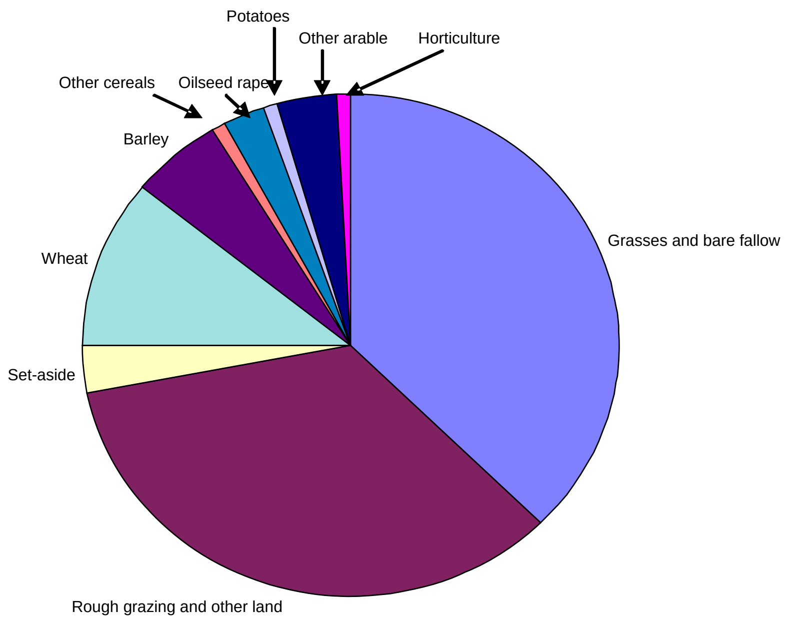


Figure 4.2. Agricultural Land Use in the UK 2004 Source: Defra (Accessed 1st March 2006)

The different agricultural practices identified within section 3 occur with varying degrees of intensity or frequency within different farming systems. Figure 4.3 sets out the relationship between practice and farming system; a rating of 3 stars indicates a common practice, two stars is infrequent and one star is rare. The term grazing refers to both species and stocking density; manure refers to solid farm yard (straw based) manures, rather than slurries and composts; fertilizer encompasses permitted supplementary nutrients within organic farming systems; similarly for biocides and veterinary medicines; species diversity refers to the range of species and varieties grown within the whole farming system.

	Pasture Intensive	Pasture Extensive	Ley/arable	Arable	Horticulture
Grazing	***	**	**		
Manure	***	*	***	*	*
Fertilizer	**	*	**	***	**
Lime	*	*	**	**	**
Vet medicines	**	***	**		
Drainage	**	*	**	**	**
Species diversity	*	**	***	*	**
Biocides	*	*	**	***	***
Tillage	**	*	**	***	***

Figure 4.3. Comparison of frequency or intensity of different agricultural practices within different farming systems.

Table 4.1. Shows the range of agricultural practices described in section 3 and the degree to which they are permitted, restricted or prohibited within organic farming systems

Practice	Permitted/Restricted/Prohibited	Comment on occurrence
Tillage	Permitted	Common within ley arable, predominately arable and horticultural systems. Less common in intensive grassland. Restricted on permanent pastures.
Rotation of a variety of crops	Permitted	Diversity in space and time encouraged as good practice.
Grass/clover mixture	Permitted	Most common on ley-arable systems and intensive grassland. May be found on in bye land on upland/extensive systems. Arable and horticultural systems may use pure clover and or other legumes in preference.
Crop residues	Permitted	Incorporation of crop residues encourage were tillage is practicable.
Herbicides	Prohibited	
Pesticide application	Restricted	Use restricted to a narrow range of products predominately used in intensive horticultural systems.
Grazing intensity / Stocking rates	Restricted	Land related activity, in so far as the number of animals relate to the land area available without causing problems of over-grazing, erosion and to allow for the spreading of livestock manures without adverse effects on the environment. Livestock units equivalent must not exceed 170 kg N ha.
Lime	Permitted	May be applied if crop nutrition and soil condition can not be maintained through rotation and recycling composts, FYM etc.
Fertiliser	Restricted	May be applied if crop nutrition and soil condition can not be maintained through rotation and recycling composts, FYM etc.
FYM	Restricted	Need recognised by inspection body and does not exceed 170 kg N ha per year of agricultural area used.
Slurry	Restricted	Need recognised by inspection body and does not exceed 170 kg N ha per year of agricultural area used.
Compost	Restricted	Product derived from source that has been submitted to either composting or anaerobic fermentation.
Sewage Sludge	Prohibited	
Drainage/ Irrigation	Restricted	Need recognised by inspection body.

From Defra (2003a)

4.1 Making comparisons between farming systems-methodological issues

There have been a number of studies that have explicitly compared the impacts of a range of farming systems on below-ground ecology. The major difficulties of all such comparative studies lies in the validity of the basis of the comparison and then subsequently in the applicability of the results to other sites and as a guide for practical farm management. One of the difficulties of interpreting comparative experiments, and indeed making comparisons between separate studies on farming systems, is the lack of definition of the terms “low input”, “integrated” and “conventional” farming. Integrated systems is the most common term in Europe and usually refers to reduction is fertiliser and herbicide and pesticide inputs coupled with reduced tillage approaches in arable cultivation. Organic farming systems are defined in law, and hence easier to identify in practice. However, there is a very wide variation in farming practices within all systems and where a large number of paired sites are compared it is usual to find a continuum rather than any sharp divisions in farming practice between systems (van Diepeningen *et al.* 2006; Elmholt and Labouriau 2005).

The two main approaches taken in studies which seek to compare farming systems are: i) sampling within the plots of well-established comparative systems trials or ii) identifying paired field/farm comparisons within which the difference is due to the change in system alone. It is an extremely complex task to select sites to allow an effective on-farm comparison; to be able to draw conclusions that relate to the system or other planned differences such as soil texture, there is a crucial dependence on the premise that these are the only varying factors (Yeates *et al.* 1997). Often a balance of the two approaches is taken with studies sampling field scale plots from comparative trials or fields from experimental farms alongside on-farm comparisons to simplify site selection and ensure good field records are available e.g. Shannon *et al.* (2002), Elmholt and Labouriau (2005). As far as possible, comparison should only take place where the same crop (or at least rotational stage) is grown on the same soil type and sampled at the same growth stage in the same season; where these factors are also allowed to vary alongside other system components such as fertilisation, tillage intensity, herbicide use, then any conclusions about farming system effects are considerably weakened (e.g. Girvan *et al.* 2003).

In on-farm situations agricultural systems rarely have completely stable management practices and hence it is unlikely that any differences observed in below-ground ecology will represent the true potential/equilibrium difference between such systems. Comparative systems trials usually increase in scientific value with the length of time they are established (e.g. Powlson and Johnston 1994) and some such trials may indeed reaching an equilibrium. However, it is important to realise that agriculturally managed systems even under stable long term management are not the climax ecological situation for their environment, instead they are regulated by a series of human managed perturbations and represent something closer to pulse stabilised systems (Odum 1969). Therefore any change in management within the system may cause a set of complex and interacting changes in below-ground ecology. The design of long-term experiments needs reflect farm practices but in a controlled way and to be flexible enough to reflect evolving changes in husbandry practices in the “real world” yet stable enough to provide continuity (Powlson and Johnston 1994). It is important to separate out those aspects of the system which need to be assessed at the whole systems level i.e. those which are dominated by interactions or large scale ecological processes, and those which can be compared at the small plot scale (Atkinson and Watson 2000). Comparison of crop rotations is an interesting example. As soon as the crops or even varieties within a rotation are changed the impact of that rotation will change, regardless of the production system. However, it is also a fact that under given soil and climatic constraints the most productive choice of crops and varieties in a rotation will differ depending on whether the system is managed conventionally or organically. Thus, are any differences between the biophysical aspects of the rotation due to the system or the rotation? The DOC Trial at FiBL has compared the same crop rotation under different systems of manuring and pest management since 1978 (Mäder *et al.* 2002). This has provided a wealth of interesting information on soil properties and crop protection and production but the question remains as to how applicable this information is in the context of practical farming. The integration of grazing livestock into trials is difficult and despite the reliance on forage legumes for fertility building in many organic systems, surprisingly few trials actually include grazing livestock. Many trials utilise livestock manure to mimic whole systems, but these can never truly represent realistic grazing situations where there is constant interaction between soils and plant and animal production. To reduce the problems associated with small plot management a number of systems trials have taken whole field or split field approaches e.g. in the UK, the LINK-IFS comparison of conventional and integrated farming systems (Ogilvy 1996), the Focus on Farming project at CWS Stoughton and the Rhone-Poulenc Boarded Barns Project which both compared organic, integrated and conventional systems (Higginbotham *et al.* 1996).

There are a number of medium to long-term studies of farming systems within the UK (Table 4.2); some of these trials have been used to determine the impact of farming systems on below-ground ecology, results of those studies will be drawn out in the following sections. In addition there are a number of long-term studies under similar climatic conditions and using similar crop rotations and/or livestock systems where measurements relevant to assessment of impacts of systems on below-ground ecology have been made (Table 4.3) and these results will also be described in full.

Table 4.2. Relevant long-term experiments comparing farming systems in the UK

Country	Plot/ field	Soil type	Crop rotation	Comp	Key reference
Rhone-Poulenc Boarded Barns study	Field	Not specified	C: w. wheat, OSR, beans O: w. wheat, ley, others I: w. wheat, OSR, beans	C, I, O	Higginbotham *et al.* 1996
CWS Focus on Farming practice study	Field	Not specified	C: w. wheat, OSR, beans w. wheat, OSR, ley I: w. wheat, OSR, beans O: w.wheat, oats, beans, gm (stocked) w.wheat, oats, beans, ley	C, I, O	Leake 1996
LINK/IFS project	Field	Six sites range of soils	Geographically relevant five course rotations	C, I	Ogilvy 1996
SAC Organic rotation trials	Replicate plots	Two sites sandy loam, loamy sand	Grass-clover, oats, swedes, oats (stocked)	O	Watson and Younie 1995
ADAS Terrington	Plots	Silty clay loam	Wheat, potatoes, beans wheat, gm	O Surrounding C fields sometimes taken as comparison	Cormack 1997
Ty Gwyn	Field/ farm	Not specified	Some permanent pasture. Reseed leys Cereal production on ss system only	O comparing Purchased feed system (pc) with a self sufficient system(ss)	Weller Pers Comms
ADAS Redesdale	Field/ farm	Peat Peaty mineral soils	Inbye (c 5%) Grass-clover intensive management. Reseeded hill pasture (c 20%) Rough grass with g/c Hill (> 70%) Rough hill grazing *Mollinia and calluna* sp.	C, O at a range of stocking rates	Anon, 2001 Final report to MAFF
ADAS Pwllpeiran	Field	Not specified	Improved grassland 9.1 ha; < 10% of total holding Hill grazing	C O	Frost *et al.* 2002
HRI Wellesbourne	Plots	Sandy loam	Arable- including cereal and potatoes Vegetable – potatoes, onion, carrot, leek Vegetable-cereal - spring cereal, cabbage, onion, carrot, leek. Permanent ley – grass-red clover	4 x O	Bending *et al.* 2004
Nafferton University of Newcastle	Split dairy farm	Sandy (clay) loam	C: w. wheat, w. barley OSR, grass ley O: s wheat, grass-red c, barley, beans, grass-white c	C, O	University of Newcastle, 2006.
	Plots		Wheat, potato, cabbage, onion and iceberg lettuce	C, I, O	

Table 4.3. Relevant long-term experiments comparing farming systems under comparable climates where measurements relevant to assessment of impacts of systems on below-ground ecology have been made

Country	Plot/ field	Soil type	Crop rotation	Comp	Key reference for site description
Switzerland DOC	Plot	Loess soil, silty clay loam?	All: Potatoes, w wheat, fodder beet, winter wheat, ley	C, C (manure), O , B	Mäder *et al.* 2000
Germany FAM Scheyern	Field	Variable soils, fine to coarse loamy	I: Wheat, maize, potatoes, with catch crops O: Clover, potatoes, wheat, rye, lupin/sunflower	I, O	Schröder *et al.* 2002
Netherlands Soil Ecology of Arable Farming Systems. (SEAFS) Lovinkhoeve Experimental Farm	Field	Calcareous silt loam	Both: w. wheat, sugar beet, s barley, potato	C, O	Kooistra *et al.* 1989
Norway Apelsvoll	Plot (split)	Loamy sand, sandy loam	Forage – 3 year ley, Arable – O includes short-term ley, gm	C, I, O Arable /Forage	Eltun 1994
Sweden, Öjebyn project	Field/ farm	Fine sand / silty loams	Dairy herd. Ley, barley, potatoes	C, O	Jonsson 2000

However, there are two internationally important systems comparisons trials where a significant amount of work studying the impact of farming practice on below-ground ecology and soil processes has been carried out, whose results are largely not applicable to UK conditions due to significant climatic and cropping systems differences:

- the Sustainable Agricultural Systems (SAFS) project at the University of California Davis is a long-term multidisciplinary study established in 1989 comparing farm management systems under irrigation in a Mediterranean climate whose main cash crop is processing tomatoes (Temple *et al.* 1994). Detailed observations of microbial population size, composition and dynamics and populations of other below-ground fauna have been made (Bossio *et al.* 1998; Gunapala and Scow 1998; Berkelmans *et al.* 2003); the main factors causing observed increases of populations and activity under organic management are the size and quality of the C inputs;
- the Farming Systems Trial at the Rodale Institute was established in 1981. Treatments consist of three farming systems: conventional, organic based around animal production using manures, a stockless organic system using N fixing cover crops (Drinkwater *et al.* 1998). Crop rotations are based around the cultivation of maize and soy bean. Wander *et al.* (1995) showed only very small differences between the size of the microbial biomass

under field conditions, but significant differences between activity of the populations. All treatments showed similar levels of arbuscular mycorrhizal fungi (Franke-Snyder *et al.* 2001); all plots also had very high levels of available P.

A number of previous reviews have drawn together data on the impacts of farming systems on biodiversity. These comparisons often yield uncertain results: Foissner (1992) concluded that "It is increasingly evident that generalisations like – conventional farming destroys life in the soil – or- Ecofarming stimulates soil life – are only partially supported by the available data". However, the most recent reviews largely show positive impacts of organic farming in comparison with conventional systems. Bengstsson *et al.* (2005), as part of a meta-analysis of all aspects of biodiversity in organic farming systems, showed that soil organisms were generally more abundant in organic agriculture systems, but heterogeneity among studies was large. Positive impacts on earthworms, microarthropods (mites and collembola) and fungal populations were confirmed, whereas effects on bacterial biomass and activity were unclear (Bengstsson *et al.* 2005). Hole *et al.* (2005) showed that there were more studies showing positive than no difference and/or negative results for the below-ground species covered by their review (Figure 4.4). Trewavas (2004) suggests that comparison of organic and integrated farming systems would be more appropriate, wherever possible in the sections that follow comparisons between conventional, integrated and organic systems will be drawn.

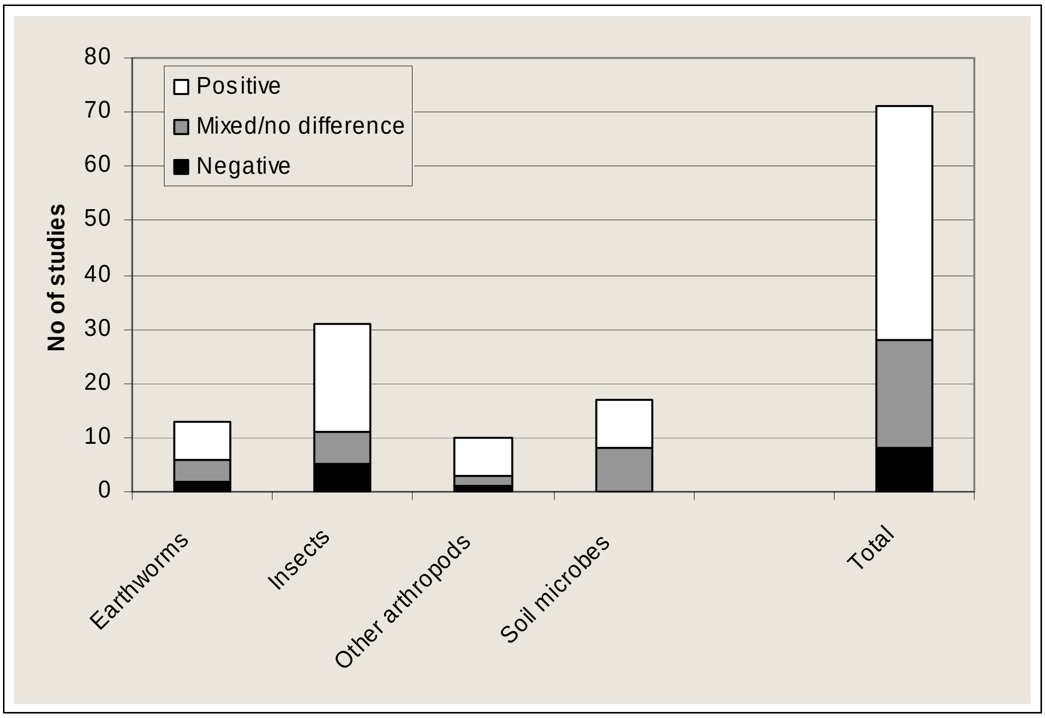
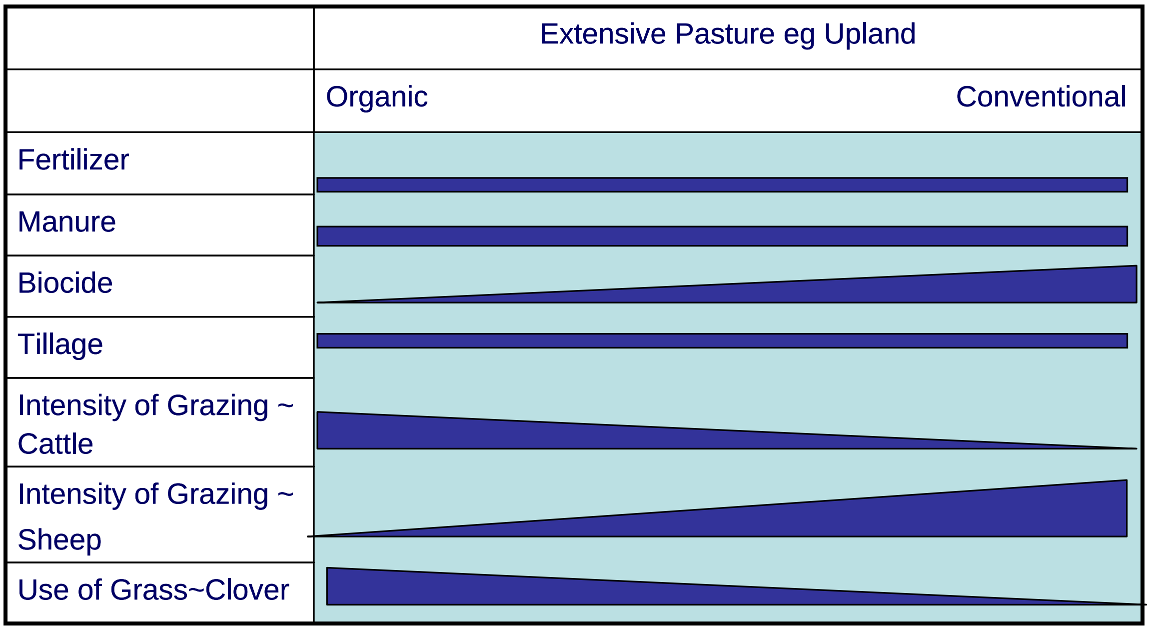


Figure 4.4 Number of studies showing impacts of organic farming systems on below-ground organisms adapted from data presented in Hole *et al.* (2005).

4.2. Comparisons in upland farming systems

Differences between management systems in the uplands are relatively small (Figure 4.5), with the largest differences seen in the management of the small areas of in bye land associated with a holding (Anon 2001). In bye is not discussed further in this section, as it is expected that the main impacts will largely match those seen in lowland pastures.



NB: Manure refers to solid farm yard manures

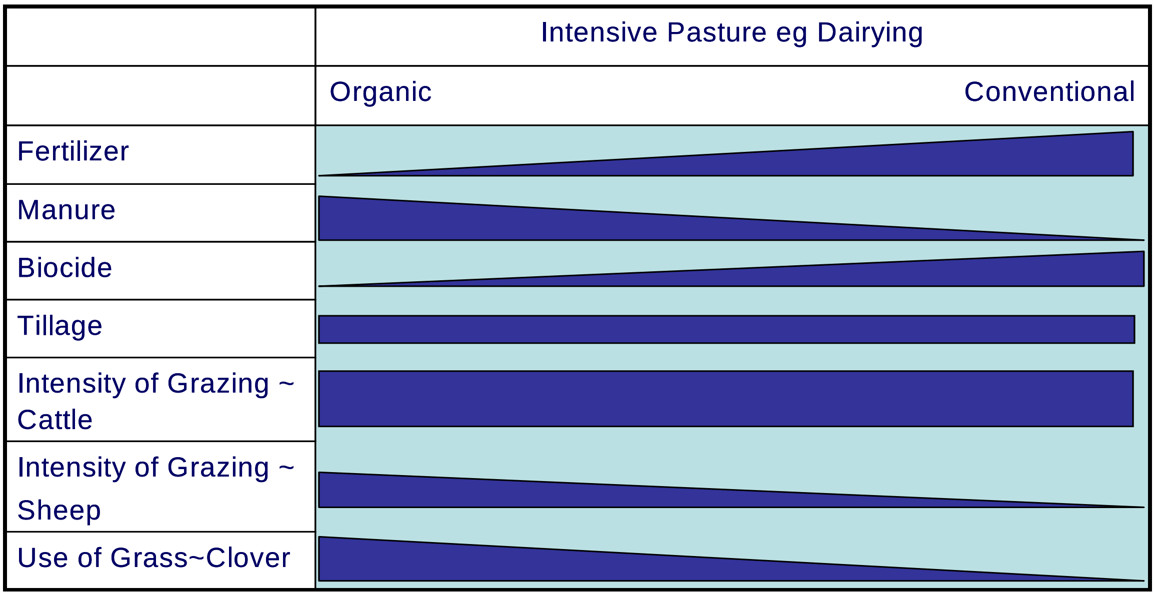
Figure 4.5. Comparative weightings of different practices between organic and conventional farming systems: Upland or extensive grassland systems

As far as we could determine no comparative studies of below ground ecology have been carried out in the hill or improved hill pastures at ADAS Redesdale or ADAS Pwllpeiran. McCaig *et al.* (1999) showed no significant difference in bacterial diversity between improved and unimproved grassland despite differences in grazing intensity, fertiliser use and plant species; there was, however, some indication of an underlying difference in specific population components. Parfitt *et al.* (2005) studied soil biota along a fertility gradient in hill pastures in New Zealand, similar to good improved hill land in the UK. Sites were all permanent pasture, showed a range of plant species and a range of stocking rates (6-16 ewe equivalents) and fertiliser applications (to a maximum of 90 kg N and 33 kg P ha^{-1}). Bacteria and nematode populations showed an increase with increasing fertility; the same trends were found in organic and conventional pastures and the farming systems were not distinguishable in this study (Parfitt *et al.* 2005).

Almost no work has been done to study the impact of differences between farming systems in upland pastures. However, from information compiled on the impacts of farming practices, it might be expected that organic management, which is likely to reduce stocking density on unimproved grazing (Anon 2001) will have a negative impact on below-ground ecology in these areas (Bardgett *et al.* 1993; Bardgett *et al.* 1997 Section 3.7). There is some evidence of reduced numbers and activity of dung beetles where veterinary drugs are used regularly in lowland grassland systems (Hutton and Giller 2003); anecdotal evidence suggests that dung decomposes more slowly where wormers have been used, but there is currently no evidence of the impact of the use of veterinary medicines on below-ground ecology in upland systems.

4.3. Comparisons in intensive grassland

There are much larger differences between the extremes of organic and conventional practices in dairying and its associated grassland management. (Figure 4.6) Here there the diagram suggests more use of straw based (solid) manure in the organic system where as the conventional system will have a greater proportion of liquid slurries. Both systems will use drainage where necessary to improve accessibility to pasture and lime to maintain an optimum pH for grassland and cultivation of forage crops. However, there are marked differences in the use of mineral fertiliser (Figure 4.6); in addition organic farming systems are more likely to handle animal waste as farmyard manure rather than slurry. Organic dairy systems are built around the use of grass-clover leys, rather than rye-grass only swards. However, an increasing number of conventional farmers are also developing highly productive systems based on grass-clover swards (MDC 2000). Stocking density may be slightly lower in organic systems 1.14 to 1.83 LSU/ha (Weller, pers. comm.)



NB: Manure refers to solid farm yard manures

Figure 4.6. Comparative weightings of different practices between organic and conventional farming systems: Intensive Pasture.

Yeates *et al.* (1997) compared 3 paired conventional (c. 5 year grass-clover swards receiving NPK fertilisers) and organic (> 10 year grass-clover swards with a significant proportion of other grass species, slurry applied at 1 site) grasslands on three contrasting soil types (silt, loam and sand). Microbial biomass, micro-, mesofauna and earthworm populations were determined. Fungal and total nematode populations were increased at all sites. Other species showed an interaction between site and management effect (loam soils often showed a different pattern). Bacterial, mites and tardigrade populations tended to be increased under organic management, whereas earthworm populations were reduced. Foissner (1992) and Younie and Armstrong (1995) found no significant differences between management systems in grassland on earthworms

Detailed analysis of the nematode population showed that fungal feeders were increased at all sites under organic management, whereas bacterial feeders, predatory and plant feeding species showed strong interactions with site and no clear effect of management (Yeates *et al.* 1997). Results from a gradient of grassland sites in the Netherlands (Mulder *et al.* 2003) showed higher diversity of bacterial and fungal feeding nematodes under organic than intensive grassland management systems – a gradient of practice related to management intensity including increasing livestock density, increasing use of mineral fertilisers and biocides and reducing use of FYM was established. The highest nematode diversity was recorded on an organic farm (Mulder *et al.* 2003). There was a strong relationship between higher bacterial populations and bacterial feeding nematodes, both increasing with intensity of management. Hyphal feeding nematodes show much lower resilience than bacterial feeders to increasing intensity of farming practices with lower number of taxa and fewer individuals in general in all the grassland systems studied (Mulder *et al.* 2003). Oehl *et al.* (2003) studying nematodes across a management intensity gradient in Germany and Switzerland showed similar trends to those of Mulder *et al.* (2003).

Eason *et al.* (1999) compared the AM fungal spore densities and infection potential of soil under conservation/grazing pastures in organic (6) and conventional farming systems (7), in total 24 fields were sampled; all with a clay loam (Denbigh series) soil. Organic farms had high proportions of clover in the sward, than their conventional pair. Soils under organic management had 3 times greater spore density and 30% greater AMF infection of roots than conventional swards. This higher infectivity was also shown in greenhouse tests with trap cropping (Eason *et al.* 1999). No measurements were made of species diversity.

The limited data available comparing conventional and organic management of lowland grassland suggests that with increasing intensity of grassland management (inorganic fertiliser N use, increasing stocking density) there is an increasing dominance of the bacterially mediated decomposition pathway indicated by nematode channel ratio (Mulder *et al.* 2003). It is likely that AMF increase as a result of decreasing P availability in organic systems, conventional dairy pastures often have very high P availability (Haygarth *et al.* 1998). Decomposition pathways under organic grasslands are therefore likely to be more complex/ diverse than under high intensity conventional grassland with consequent effects for the higher trophic levels of the food web. However, there are few studies of the impact of organic management on meso- and macrofauna other than earthworms in lowland grassland systems.

4.4. Comparisons in cropping systems

Most studies comparing the effects of organic, integrated and conventional systems have been carried out in arable systems. In these systems, there are large differences between organic and conventional management (Figures). All systems use lime to maintain an optimum pH (around 6.5). However, there are marked differences in the use of mineral fertiliser, herbicides and pesticides. Organic livestock-based ley-arable systems use grass-clover leys and manures to maintain soil fertility whereas stockless farms use N fixing green manure crops (Watson *et al.*, 2000). Integrated farming systems use reduced tillage approaches, whereas both conventional and organic farms tend to use more intensive cultivation systems (Jordan *et al.* 2000).

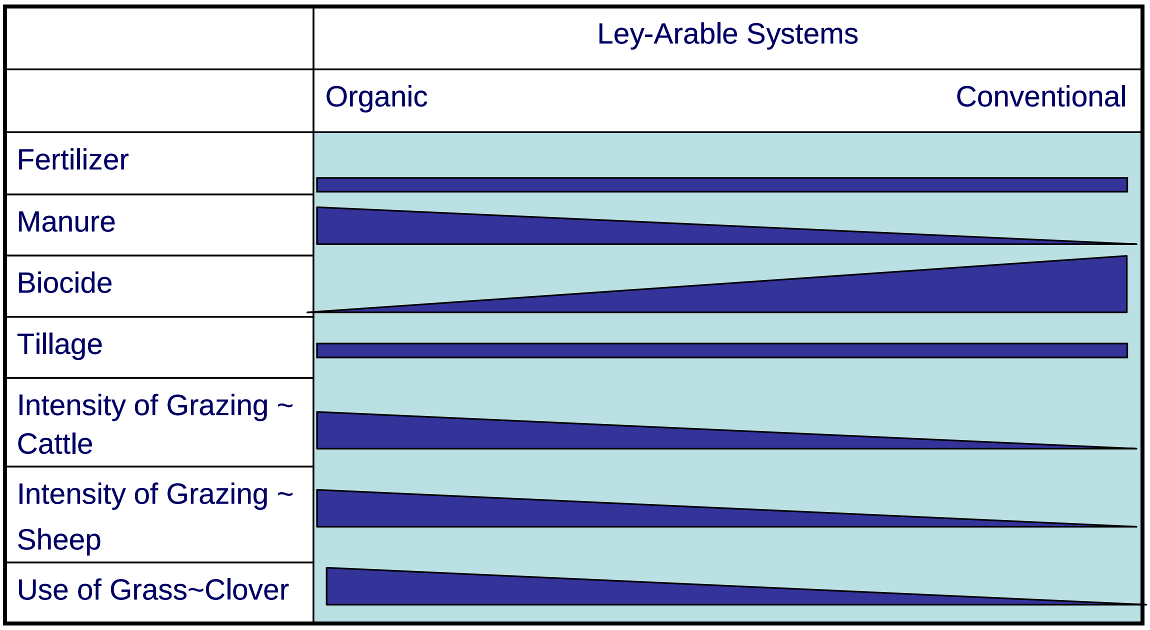
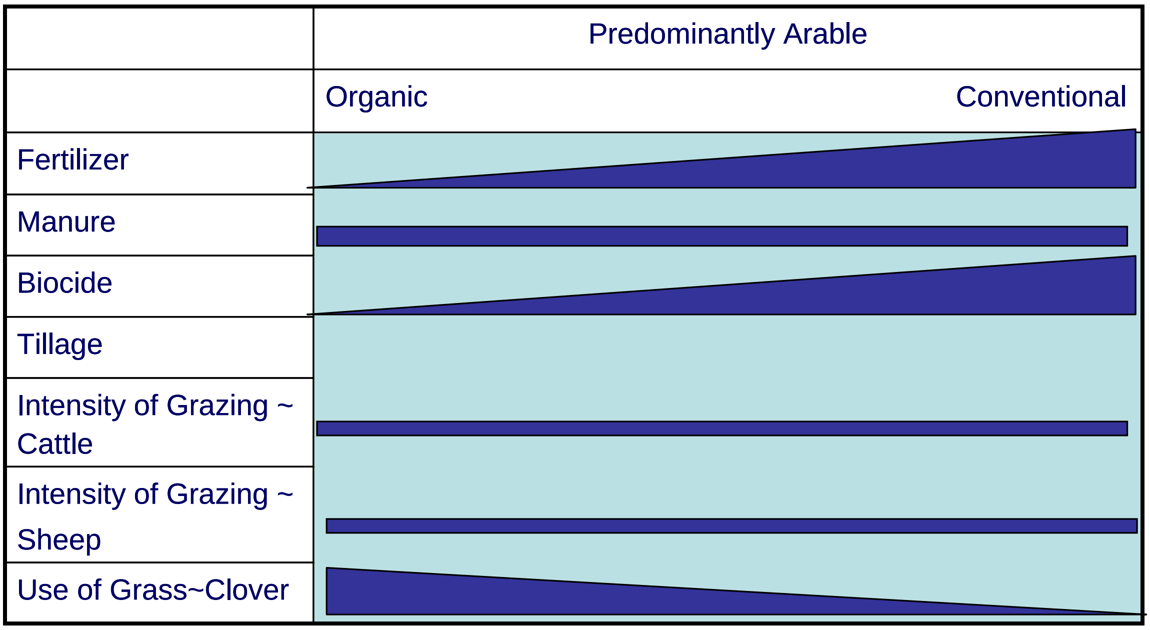


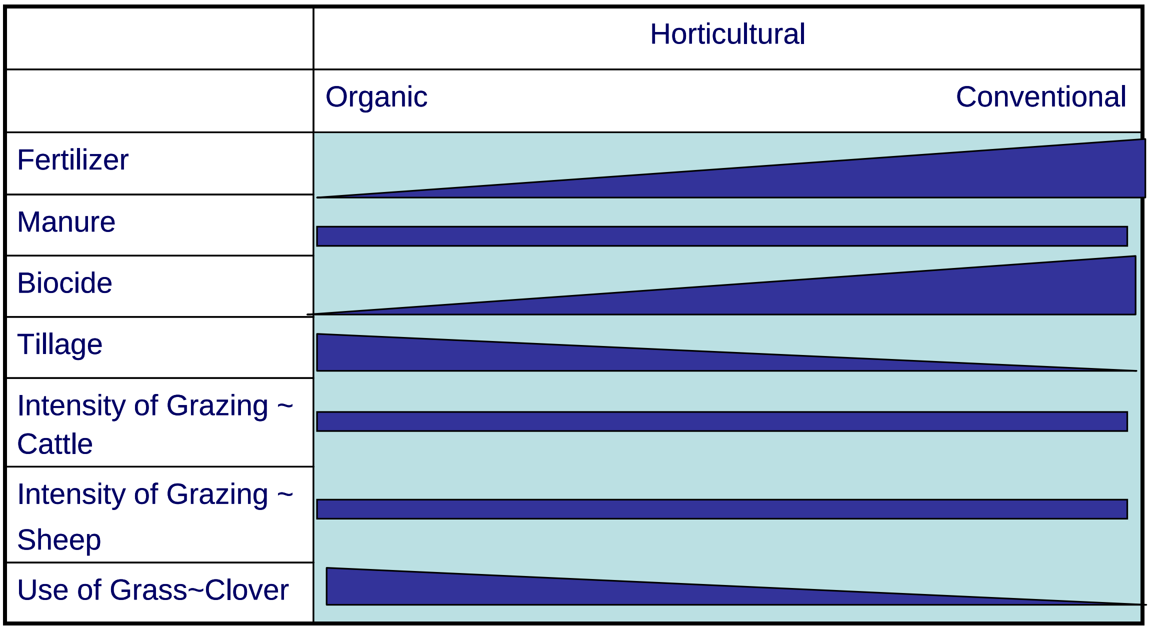
Figure 4.7 Comparative weightings of different practices between organic and conventional farming systems: Ley-Arable.

Bloem *et al.* (1994, SEAFS study) showed higher bacterial and fungal biomass in the surface soil (0-10 cm) of soil under integrated compared with conventional management. Filser *et al.* (2002; FAM Scheyern) showed no differences between microbial biomass under integrated and organic management systems. Paired farm studies in New Zealand showed higher microbial biomass in organic than conventional systems (Reganold *et al.* 1993; Murata and Goh 1997). However, van Diepeningen *et al.* (2006) showed a larger effect of soil type (sand v clay) on the size of the microbial biomass than farming system in a comparison of 13 paired sites. Breland and Eltun (1999; Apelsvoll) showed increased microbial biomass in forage than arable rotations in conventional and integrated systems. However, microbial biomass in organic systems did not differ significantly between organic forage and arable systems, which include a ley phase and green manure crops). Organic arable rotations were much better able to buffer changes in soil microbial biomass resulting from increased tillage, probably as a result of increased organic matter inputs; these was also an interaction with development of more stable soil structure (Breland and Eltun 1999). Watson *et al.* (1996; SAC rotations trial) showed that microbial biomass population and respiration rates were highest in the grass-clover ley phase of the organic rotations, which tends to support the proposed mechanism of change at Apelsvoll.



NB: Manures refers to solid farm yard manures

Figure 4.8. Comparative weightings of different practices between organic and conventional farming systems: Predominately Arable



NB.Manures refers to solid farm yard manures

Figure 4.9. Comparative weightings of different practices between organic and conventional farming systems: Horticulture

Jensen *et al.* (2000) compared ergosterol (as an indicator of fungal biomass) in soils from a biodynamic, a conventional arable and conventional livestock farm. The conventional livestock farm had greater ergosterol contents than the conventional arable farm, the biodynamic system which had a mixed cropping rotation, and intermediate inputs of OM had intermediate ergosterol contents; spatial heterogeneity in ergosterol also increased with pool size. However, Shannon *et al.* (2002) showed few (and no consistent) differences in bacterial or fungal biomass between organic and conventional systems (soils sampled from CWS Focus on Farming practice study; ADAS Terrington). Elmholt and Labouriau (2005) showed that variation in fungal abundance within farming systems (C, I, O) was as great (and often greater) than differences between farming systems.

Black and Parekh (1998) measured substrate utilisation patterns (Biolog GN) by soil inocula which indicated a higher activity of microbial biomass under integrated than conventional systems (LINK-IFS). However, site characteristics interacted strongly with farming system. Fliessbach and Mäder (1997; DOC trial) measured higher functional diversity (Biolog GN) coupled with lower qCO_2 in organically managed plots. Shannon *et al.* (2002) suggested that organically managed soils showed some indication of a larger population of viable but non-culturable micro-organisms. A study by Girvan *et al.* (2003) gives some indication of increased diversity of bacterial populations under organic management; however, a large number of factors varied between the sites studied.

Sattelmacher *et al.* (1991) and Ryan *et al.* (1994) showed higher arbuscular mycorrhizal colonisation in organic than conventional paired cropping systems. Scullion *et al.* (1998) showed that arbuscular mycorrhizal fungi (AMF) in organic paired farms had a higher infectivity to roots than conventional systems (compared at the same inoculum levels); the largest difference was seen on an inherently low P status soil. Scullion *et al.* (1998) also found that AMF inoculation effectiveness increased with time since conversion to OF. Bending *et al.* (2004) found a much lower AMF colonisation potential in conventional wheat than under four contrasting OF management regimes (HRI Wellesbourne); there were no differences between the different organic management treatments. Sattelmacher *et al.* (1991) and Oehl *et al.* (2004, DOC trial) also showed a higher diversity of spores under organic management regimes. Gosling *et al.* (2006) summarised evidence from 13 available studies showing greater root colonisation, larger numbers of AMF spores and greater diversity of AMF in organically managed soils. However, they also identified poor performance of AMF in some organic systems; the cause could not be identified because of differences in the details of management practices used in organic systems and contrasting land management at the sites before conversion.

Foissner (1992) showed no differences in protozoan populations between organic and conventional farming systems in Austria. In contrast, Foissner (1992) and Neher (1999) showed that overall nematode abundance was higher under organic management than in comparable conventional systems. However, van Diepeningen *et al.* (2006) showed a larger effect of soil type (sand v clay) in the Netherlands on the size of the total nematode population than farming system in a comparison of 13 paired sites. Oehl *et al.* (2003) observed that organically managed arable land in crop rotation maintained nematode species diversity and species type more similar to low intensity grass than comparable arable systems. Neher and Olson (1999) showed that nematode community varied with fertiliser and crop protection systems. Different components of nematode biomass responded differently to different management practices (Neher and Olson 1999). De Ruiter *et al.* (1993

SEAFS) showed higher populations of predatory and bacterial–feeding nematodes in the topsoil (0-10 cm) of integrated compared with conventional systems; whereas herbivorous nematodes were increased in conventional systems.

Long-term system studies in the Netherlands, USA and Sweden, have found fundamental differences in soil food web structure between conventional and less intensive farming systems (Brussaard 1994, the Netherlands), Crossley *et al.* (1989, USA) and Andren *et al.* (1990, Sweden). De Ruiter *et al.* (1993) showed that microbes, protozoa and nematodes contributed more to the total amount of N mineralised in the integrated system, while in the conventional system mites and enchytraeids made a larger contribution to mineralization. Contribution to mineralisation is a function of both species abundance and turnover (De Ruiter *et al.* 1993). It has been suggested that as in grassland systems, in conventional systems, the bacterial community dominates the microbial component while, in less intensive systems, the fungal community was the dominant microbial component. There is some indication that this is so, however, it is less certain than in grasslands. Such differences influence nutrient cycling and have implications for the efficient use of nutrient inputs and leaching potential (De Ruiter *et al.* 1994; Pankhurst *et al.* 1994).

Long-term system studies in the Netherlands, USA and Sweden (Brussaard 1994, Crossley *et al.*1989 and Andren *et al.* 1990 respectively) have found most soil faunal groups were more diverse or unchanged in less intensive farming systems (organic and integrated) than in conventional systems. Black and Parekh (1998) showed little difference in mesofaunal abundance between conventional and integrated (reduced tillage reduced inputs) plots under winter wheat at three sites across the UK (LINK IFS). Filser *et al.* (2002) also found no difference in collembolan population size or species abundance between integrated and organic farming systems. Integrated data from LINK-IFS, Boarded Barns and CWS Focus on Farming Practice sites (Higginbotham *et al.* 2000) showed an increased population size of collembola in integrated systems and no difference between organic and conventional. In the same systems’ trials, Alvarez *et al.* (2001) showed that many collembolan taxa were ubiquitous in arable farming systems, but taxa responded differently to farming systems; integrated management systems had greater total collembolan populations. The most important source of variation was local differences between management practice rather than regional variation. Alvarez *et al.* (2001) suggest that collembolan species showing increase in organic farming systems are those that prefer increased humidity and hence higher weed populations.

Holland *et al.* 1996 (LINK IFS) found little difference in carabid populations between integrated and conventional systems; there were similar interactions with site as seen for collembola (Black and Parekh, 1998) and a strong influence of crop on carabids (a significant reduction of carabids was seen with seed potatoes). Mäder *et al.* (2002, DOC trial) reported significantly higher carabid populations in organic management regimes. Doring and Kromp (2003) carried out a meta-analysis of all the data collected on comparisons of carabids in organic and conventional systems in Southern Germany and Switzerland and showed that on average there were 34% more species found in organically managed winter wheat; carabid species also responded to management systems differently with some species showing increases and some decreases in organic systems. Identification of key species traits and requirements is needed if optimum management practices for any particular species are to be developed e.g. *Carabus auratus* often shows increased populations in organic farming systems. However, it is not found is very sandy soils, and shows a strongly negative response to mechanical weeding in spring due to its long larval stage (Doring and Kromp, 2003).

A long term study comparing conventional and reduced rates of pesticide application at 3 sites across the UK (SCARAB, Tarrant *et al.* 1997) showed large differences in earthworm populations between sites, but negligible differences due to the contrasting pesticide management systems. An increase in earthworm population the integrated farming system at Boarded Barns (I > O > C; Higginbotham *et al.* 2000) was linked to the reduced cultivation system, since there was a particular increase in shallow burrowing species. Earthworms are generally found to be higher in organic than conventional systems (Reganold *et al.* 1993; Mäder *et al.* 2002, DOC trial). Within organic rotations (Watson *et al.* 1999) earthworm numbers were found to be highest in the 2nd year of grass-clover ley, the population declined through the tillage phase of the rotation with temporal distance from the ley phase.

Given our limited ability to apply robust taxonomic classification systems to below-ground groups, it is probably not surprising that a range of positive and negative effects on below-ground ecology are observed as a result of the application of contrasting cropping systems. For groups, which can be resolved to the species level e.g. collembola, carabids, differential effects of systems are found on different species. However, on average, organic and integrated system have positive effects on below-ground ecology. Tillage intensity seems to have the largest effect, but impacts of cropping management (particularly the amount and quality of OM returned) can moderate the impact of even quite severe tillage operations and seem to increase the resilience of below-ground ecosystems.

4.5. Integrated effects of combinations of management practices

While the impacts of separate management practices on soil organisms can be distinguished (Section 3), in practice, there will be interactions between different management practices for most organisms. Taking management practices individually it is almost impossible to derive recommendations in relation to how they should be optimised for belowground ecology and the delivery of soil multi-functionality. Consequently, while authors of reviews such as this argue that maintenance and enhancement of soil biological fertility is of benefit within all agricultural systems (e.g. Doran and Smith 1987; Beauchamp and Hume 1997; Clapperton *et al.* 2003), they highlight in their conclusions how this can be put into practice at a farming system level only in very general way. For example, Clapperton *et al.* (2003) conclude: "Ideally agroecosystems should be managed to maintain the structural integrity of the [soil] habitat, increase SOM and optimise the C:N ratios in SOM using cover crops and/or crop sequence to synchronise nutrient release and plant uptake." But a farmer might well ask how many cover crops and which ones, where the right balance (economic as well as ecological) is between minimising tillage and optimising weed control … and many more.

It is widely recognised that it is differences in the quality as much as the quantity of organic matter input that have the driving impact on the microbial community in soil and on decomposition and cycling of C and N (e.g. Janssen 1984). Maintenance of below-ground diversity and a major part of the ecosystem services are also controlled, at least in part, by the nature of the plant community (Swift *et al.* 2004). Plants are also the main point at which humans intervene in agro-ecosystems determining the species richness, genetic variability and organisation in space and time of crops, if not of weeds. The impacts of OM inputs are then mediated by the impact of tillage and other residue management practice and the particular climate/soil conditions at any site (Doran and Smith 1987). Where plant communities are managed carefully (e.g. through return of residues, mulching etc) it has been shown that agricultural intensification does not adversely affect microbial and arthropod communities e.g. (Wardle *et al.* 1999; Yeates *et al.* 1999).

Taking AM fungi as an example (Table 4.4), reduced plant species diversity (and modern cultivars), the use of non-mycorrhizal crops, fallow and excessive tillage are all likely to contribute to a negative impact on mycorrhizal species diversity and infectivity. Rotational cropping using a rage of appropriate hosts with reduced tillage intensity and regular inputs of OM is likely to be generally positive for AM fungi. Hence advice targeted at improving AM fungal populations would stress the positive and advise minimisation of the negative, which would lead to general advice much like that indicated above. Reports in the literature of the impacts of single practices on AMF are often hedged with significant caveats in their reporting such as:

- dependent on the type of soil …,
- there is likely to be a trade off between ….,
- the point within the crop rotation at which organic amendments are applied is critical ..,
- variations both within and between seasons may lead to …

Table 4.4. Summary of impacts of agricultural practices on AM fungi (for more detail see Section 3 and Harrier and Watson 2003, Gosling *et al.* 2006)

Direction of effect	Practice
Positive	Rotations Weeds
Negative	Monoculture Non-host in rotation Bare fallow = no host Modern cultivars Intensive tillage Increased soil soluble P
Variable	Intercrops N fertilisation Organic amendments Biocides (herbicides, pesticides) Grazing

Authors of review papers are also very wary of drawing out general principles that would support practical field management:

- "though generally regarded as beneficial, the activity of mycorrhizal fungi in agroecosystems is neither easily predictable nor always beneficial" (Gosling *et al.* 1996);
- "translating the results of result into practical recommendations is difficult because of the interaction between factors involved in the plant mycorrhizal symbiosis and the separation of cause and effect" (Harrier and Watson 2003).

It is not clear that the increased understanding of physiology and function in the detail that is felt necessary to support effective guidance for practical management will ever be achieved. It is therefore by no means clear to what extent e.g. tillage intensity needs to be reduced to mitigate its effects, nor to what extent other factors e.g. organic matter inputs may moderate the impacts of tillage and hence how these practices could be optimised simultaneously to the benefit of AMF. Nonetheless interactions between practices are often the focus of farm management decisions.

Data for earthworm abundance in the rotations trials at SAC (Figure 4.10) shows the significant impact of tillage on earthworm populations; the very intensive cultivations associated root crops have a particularly negative effect. The dynamic nature of this population through the rotation makes it difficult to apply concepts such as resistance and resilience. What is the equilibrium position for this system – is it the maximum abundance achieved under grass-clover? Or must each rotational phase be compared to the identical phase in the previous rotation to see if the population is stable or changing? i.e. the rotational average through time is considered. There is also likely to be an interaction with increased organic matter inputs supporting the growth of the earthworm populations in the grass-clover phase of the rotation.

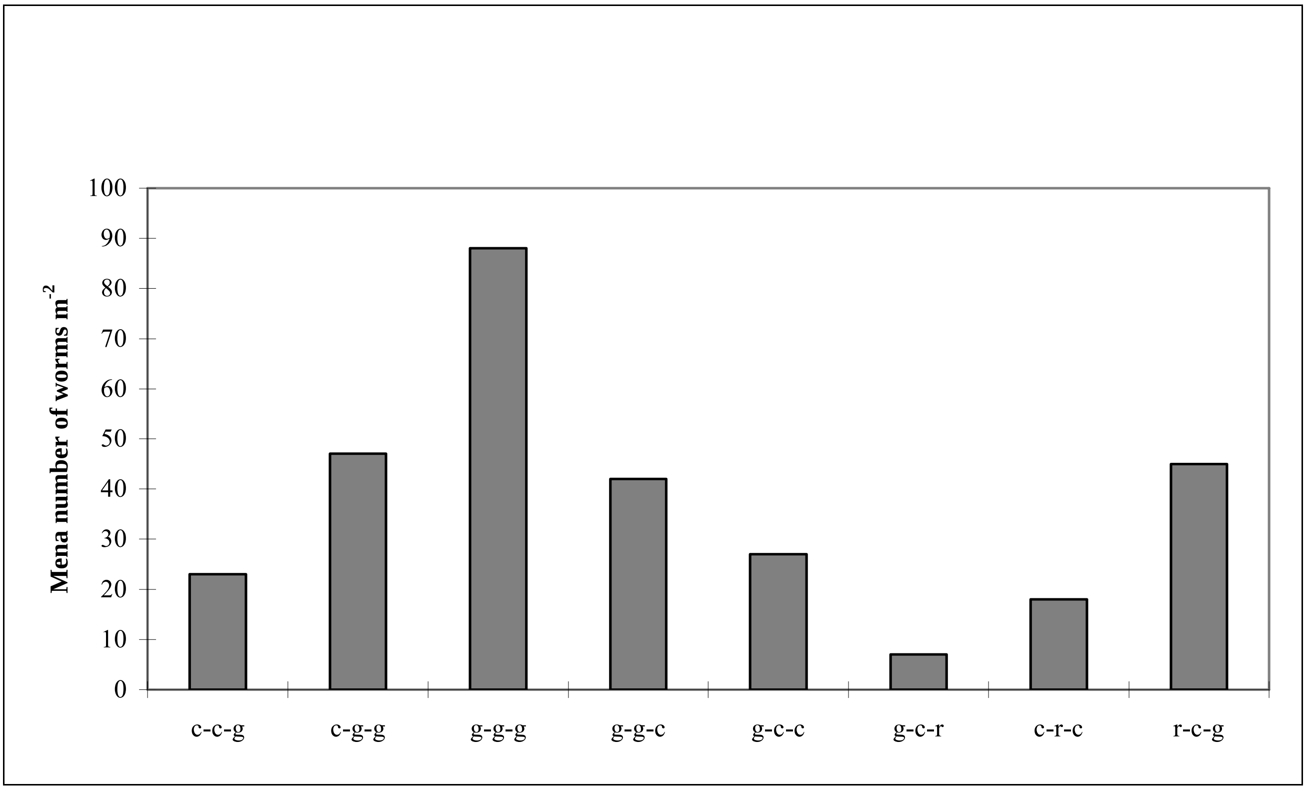


Figure 4.10. Earthworm abundance in the rotations trial; cropping sequence indicates current and two previous crops, where g – grass-clover ley, c – cereal and r – root crop. Hence g-c-r indicates that the plot is currently under root crop and was previously under a cereal crop preceded by a grass-clover ley.

These data (Figure 4.10) show that three years of a grass-clover ley allow a large earthworm population to accumulate relative to the cultivated phases; but would the population increase further after 4 years and if it did would the population level under the cereal phases be higher or is the population seen here the maximum that is achievable given the intensity and frequency of the tillage disturbance. It is likely that reductions in the earthworm numbers in the cultivated phases would reduce the development of earthworm burrows which represent important transmission pores; but might this have benefits in reducing rapid throughflow of water.

It is tempting to suggest that rotational cropping, particularly where rotational phases include no tillage such as the ley-arable farming systems once common in the arable areas of the UK, builds restorative phases for below-ground organisms into the system and hence increases the resistance and/or resilience of the below-ground ecosystem over the rotation as a whole. However, the evidence base is not in place to draw such a broad conclusion. Where rotations are designed in relation to N management it is common to talk about N fertility-building versus exploitative phases of the rotation; it may be useful to think in this way in relation to below-ground populations. Factors and management practices that increase resilience are also likely to vary for different below-ground organisms – for insect species the maintenance of a source of organisms able to re-invade may be critical (i.e. size, proximity and connectivity with an unaffected community). In some circumstances, management practices can also provide a reservoir population e.g. using tree saplings inoculated with appropriate mycorrhizal fungi. For grassland systems, balancing grazing intensity in space and time, together with the considered use of fertilisers, lime organic inputs are the main routes to allow agricultural management to support below-ground ecology. The detailed application of these broad principles will be site –specific and hence are dependent on the communication of the scientific principles to land managers in a way that allows pro-active site-adapted management. For all the farming systems the principles discussed above pose challenges, e.g. to what extent can minimum tillage approaches be adopted in organic farming systems? How can conventional intensive arable systems include increased inputs of appropriate organic amendments?

5. Conclusions

Much of the literature on below-ground ecology is necessarily divorced from a consideration of practical management whilst it goes in search of detailed understanding of taxonomy, physiology and/or function. However, there does seem to be a significant disparity between the advances that have been made in our understanding of the importance, interactions and even impacts of management practices on below-ground ecology and the transference of this knowledge into practical guidance for farmers and land managers. Some of this knowledge transfer is hampered by failings in experiment design and/or reporting of studies of below-ground ecology. For example how can it be determined whether soil texture or other factors are of significance if they are not reported clearly? We would recommend agreement of a minimum data set of site description factors e.g. topsoil texture (% sand, silt and clay), topsoil pH, soil series or equivalent linked to a relevant international classification scheme; climate, crop at time of sampling and its management in detail; abbreviated records of previous cropping and management.

Studies on the impact of individual crops have to date largely been very focussed on identifying mechanisms and usually been carried out in microcosms. Consequently there is a need for an increased understanding of whether the suite of management practices associated with particular crops is generally beneficial or detrimental to below-ground ecology. Where detrimental impacts are known e.g. as a result of the intensive tillage associated with potato cultivation, mitigating strategies should be sought and tested in the field (e.g. the approach taken in California in extension of the SAFS work, Jackson *et al.* 2004). Economic impacts also need to be more clearly addressed (Harrier and Watson 2003). There are some emerging management practices, which while small scale in terms of land covered, may have a very significant affect where they occur e.g. the increasing use of land-based systems for monogastrics.

It is clear that farm management practices do alter below-ground biodiversity and ecosystem function. However, it is much less clear what steps could or should be taken to prevent or mitigate these effects. Mulder *et al.* (2003) suggest that we have sufficient data to be able to conclude that increased intensity of management practices act on most taxa to reduce diversity within functional groups, and hence also possibly to reduce the resilience of these managed ecosystems. There is also sufficient data to indicate that reducing the intensity of use of mechanical and manufactured inputs and (re)-discovering cost-effective ways to integrate biological inputs, will benefit below–ground biodiversity, particularly in lowland grassland and cropping systems. Benefits are seen from both organic and integrated systems; the evidence base is not strong enough to conclusively distinguish the benefits of these approaches from one another in lowland arable systems. The diversity of UK farming systems means that great care need to be taken if the enhancement of below-ground biodiversity is to be included as part of any agri-environmental scheme. Best practice is likely to be farm, and even micro-site, specific.

5.1. Key questions arising from the review

How to use farmer knowledge together with scientific understanding; advisory role as mediator?

Future work

1. Uplands: question – do we want increased soil biodiversity and from what baseline?
2. Sustainable Land Management: upland advice needs to be specific.
3. Need to address the ideas of ecologically sustainable systems: and social/economic impacts of managing for belowground biodiversity
4. There is a need to address how science/policy can use the knowledge base held by farmers.
5. What is the most effective way to provide advice/information to land managers on managing soil biodiversity?

6. Acknowledgements

The authors wish to acknowledge the financial support of English Nature, Countryside Council for Wales, Department for Environment, Food and Rural Affairs and Scottish Natural Heritage.

7. References

ABAYE, D.A., LAWLOR, K., HIRSCH, P.R., & BROOKES, P. C. 2005. Changes in the microbial community of an arable soil caused by long-term metal contamination. *European Journal of Soil Science*, **56**, 93-102.

ADAMS, P.B. & AYERS, W.A. 1983. Histological and physiological aspects of infection of sclerotia of Sclerotinia species by two mycoparasites. *Phytopathology,* **73**, 1072-1076.

AHL, C., JOERGENSEN, R.G., KANDELER, E., MEYER, B., & WOEHLER, V. 1998. Microbial biomass and activity in silt and sand loams after long-term shallow tillage in central Germany. *Soil and Tillage Research*, **49**, 93-104.

AITKEN, M.N., CAMPBELL, C.D., BURGESS, S.D., SYM, G. DICKSON, J.W., & ELRICK, D. 2004. Persistence, mobility and effects on soil properties of sheep dip (Diazonin and Cypermethrin) application. *In:* D. LEWIS & L. GAIRNS, eds. *Water Framework Directive and Agriculture. Proceedings of the SAC and SEPA Biennial Conference*. Edinburgh: SAC/SEPA, 136-142.

ALVAREZ, T., FRAMPTON, G.K., & GOULSON, D. 2001. Epigeic Collembola in winter wheat under organic, integrated and conventional farm management regimes. *Agriculture Ecosystems & Environment*, **83**, 95-110.

AMATO, M. & LADD, J.N. 1992. Decomposition of C-14-Labeled Glucose and Legume Material in Soils - Properties Influencing the Accumulation of Organic Residue-C and Microbial Biomass-C. *Soil Biology & Biochemistry*, **24**, 455-464.

ANDERSON, E.L. 1987. Corn root-growth and distribution as influenced by tillage and nitrogen-fertilization. *Agronomy Journal*, **79**, 544-549.

ANDERSON, I.C. & CAIRNEY, J.W.G. 2004. Diversity and ecology of soil fungal communities: increased understanding through the application of molecular techniques *Environmental Microbiology*, **6**, 769-779.

ANDERSON, J.M. 1975. The enigma of soil animal species diversity. *In:* J. VANEK ed. *Progress in Zoology*. Prague: Academia, 51-58.

ANDREN, O. & LAGERLOF, J. 1980. The abundance of soil animals (microarthropa, enchytraeids, nematoda) in a crop rotation dominated by ley and in a rotation with varied crops. *In* : D. L. DINDAL ed. *Soil Biology is related to land-use practices.* Washington: Environmental Protection Agency, 274-279.

ANDREN, O., BRUSSAARD, L., & CLARHOLM, M. 1999. Soil organism influence on ecosystem-level processes bypassing the ecological hierarchy? *Applied Soil Ecology*, **11**, 177-188.

ANDREN, O., LINDBERG, T., PAUSTIAN, K. & ROSSWALL, T. 1990. Ecology of arable land – organism, carbon and nitrogen cycling. *Ecological Bulletins*, **40**, 1-222.

ANGERS, D.A. & CARON, J. 1998. Plant-induced changes in soil structure: Processes and feedbacks. *Biogeochemistry*, **42**, 55-72.

ANON, 2000. *Fertiliser recommendations for agricultural and horticultural crops* (RB209). Ministry of Agriculture, Fisheries and Food. London: HMSO.

ANON, 2001. Organic sheep and beef production in the uplands. Report of research project OF0147. Accessed on 25 February 2006 via www2.defra.gov.uk/research/project_data/Default.asp

ARITAJAT, U., MADGE, D.S., & GOODERHAM, P.T. 1977. Effects of Compaction of Agricultural Soils on Soil Fauna .1. Field Investigations. *Pedobiologia*, **17**, 262-282.

ATKINS, S.D., HIDALGO-DIAZ, L., KALISZ, H., MAUCHLINE, T.H., HIRSCH, P.R. & KERRY, B. R. 2003. Development of a new management strategy for the control of root knot nematodes (*Meloidogyne spp*) in organic vegetable production. *Pest Management Science*, **59**, 183-189.

ATKINSON, D. & WATSON, C.A. 2000. The research needs of organic farming: distinct or just part of agricultural research?In: BCPC Conference, 2000 - Pests & Diseases. Alton: BCPC, 151-158.

AYRES, E., DROMPH, K.M. & BARDGETT, R.D. 2006. Do plant species encourage soil biota that specialise in the rapid decomposition of their litter? *Soil Biology & Biochemistry*, **38**, 183-186.

BAILEY, K.L. & LAZAROVITS, G. 2003. Suppressing soil-borne diseases with residue management and organic amendments. *Soil & Tillage Research*, **72**, 169-180.

BANERJEE, M.R., BURTON, D.L., & DEPOE, S. 1997. Impact of sewage sludge application on soil biological characteristics. *Agriculture Ecosystems & Environment*, **66**, 241-249.

BARDGETT, R.D. 2002. Causes and consequences of biological diversity in soil. *Zoology*, **105**, 367-374.

BARDGETT, R.D. & COOK, R. 1998. Functional aspects of soil animal diversity in agricultural grasslands. *Applied Soil Ecology*, **10**, 263-276.

BARDGETT, R.D., WHITTAKER, J.B. & FRANKLAND, J.C. 1993. The diet and food prefercnes of *Onychiurus procampatus* (collembola) from upland grassland soils. *Biology and Fertility of Soils*, **16**, 296-298.

BARDGETT, R.D., LEEMANS, D.K., COOK, R., & HOBBS, P.J. 1997. Seasonality of the soil biota of grazed and ungrazed hill grasslands. *Soil Biology & Biochemistry*, **29**, 1285-1294.

BARDGETT, R.D., LOVELL, R.D., HOBBS, P.J., & JARVIS, S.C. 1999. Seasonal changes in soil microbial communities along a fertility gradient of temperate grasslands. *Soil Biology & Biochemistry*, **31**, 1021-1030.

BARDGETT, R.D., MAWDSLEY, J.L., EDWARDS, S., HOBBS, P.J., RODWELL, J.S., & DAVIES, W. J. 1999. Plant species and nitrogen effects on soil biological properties of temperate upland grasslands. *Functional Ecology*, **13**, 650-660.

BARNS, S.M., TAKALA, S.L., & KUSKE, C.R. 1999. Wide distribution and diversity of members of the bacterial kingdom Acidobacterium in the environment. *Applied and Environmental Microbiology*, **65**, 1731-1737.

BATEMAN, G.L. & KWASNA, H. 1999. Effects of number of winter wheat crops grown successively on fungal communities on wheat roots. *Applied Soil Ecology*, **13**, 271-282.

BEARE, M.H., PARMELEE, R.W., HENDRIX, P.F., CHENG, W.X., COLEMAN, D.C. & CROSSLEY, D.A. 1992. Microbial and faunal interactions and effects on litter nitrogen and decomposition in agroecosystems. *Ecological Monographs*, **62**, 569-591.

BEARE, M.H., COLEMAN, D.C., CROSSLEY, D.A., HENDRIX, P.F., & ODUM, E.P. 1995. A Hierarchical Approach to Evaluating the Significance of Soil Biodiversity to Biogeochemical Cycling. *Plant and Soil*, **170**, 5-22.

BEAUCHAMP, E.G. & HUME, D.J. 1997. Agricultural soil manipulation: The use of bacteria, manuring and plowing. *In:* J. D. VAN ELSAS, J. T. TREVORS & E. M. H. WELLINGTON eds. *Modern Soil Microbiology*. New York: Marcel Dekker, 643-664.

BELNAP, J., HAWKES, C.V., & FIRESTONE, M.K. 2003. Boundaries in miniature: Two examples from soil. *Bioscience*, **53**, 739-749.

BENDING, G.D. & LINCOLN, S.D. 1999. Characterisation of volatile sulphur-containing compounds produced during decomposition of Brassica juncea tissues in soil. *Soil Biology & Biochemistry*, **31**, 695-703.
BENDING, G.D., TURNER, M.K., & JONES, J.E. 2002. Interactions between crop residue and soil organic matter quality and the functional diversity of soil microbial communities. *Soil Biology & Biochemistry*, **34**, 1073-1082.
BENDING, G.D., TURNER, M.K., RAYNS, F., MARX, M.C., & WOOD, M. 2004. Microbial and biochemical soil quality indicators and their potential for differentiating areas under contrasting agricultural management regimes. *Soil Biology & Biochemistry*, **36**, 1785-1792.
BENGTSSON, J. 1998. Which species? What kind of diversity? Which ecosystem function? Some problems in studies of relations between biodiversity and ecosystem function. *Applied Soil Ecology*, **10**, 191-199.
BENGTSSON, J., AHNSTROM, J., & WEIBULL, A.C. 2005. The effects of organic agriculture on biodiversity and abundance: a meta-analysis. *Journal of Applied Ecology*, **42**, 261-269.
BENTON, T.G., VICKERY, J.A., & WILSON, J.D. 2003. Farmland Biodiversity. *Trends in Ecology & Evolution*, **18**, 182-188.
BERKELMANS, R., FERRIS, H., TENUTA, M., & VAN BRUGGEN, A.H.C. 2003. Effects of long-term crop management on nematode trophic levels other than plant feeders disappear after 1 year of disruptive soil management. *Applied Soil Ecology*, **23**, 223-235.
BLACK, H.I.J. 1998. Impact of farming practices on soil biodiversity: A literature review. In: The impact of farming practices on sustainable use of soil. Final Contract Report for the UK Ministry of Agriculture Fisheries and Food-funded project, OC9403 (CSA2845).
BLACK, H.I.J. & PAREKH, N.R. 1998. *A field assessment of the impact of farming practices on soil biodiversity*. Final Contract report for the UK Ministry of Agriculture Fisheries and Food-funded project, CSA2845.
BLOEM, J., LEBBINK, G., ZWART K.B., BOUWMAN, L.A., BURGERS, S.L.G.E., DE VOS, J.A., DE RUITER P.C. 1994. Dynamics of microorganisms, microbiovores and nitrogen mineralisation under conventional and integrated management. *Agriculture, Ecosystems and Environment*, **51**, 129-143.
BONGERS, T. & BONGERS, M. 1998. Functional diversity of nematodes. *Applied Soil Ecology*, **10**, 239-251.
BONGERS, T. & FERRIS, H. 1999. Nematode community structure as a bioindicator in environmental monitoring. *Trends in Ecology & Evolution*, **14**, 224-228.
BORNEMAN, J., SKROCH, P.W., OSULLIVAN, K.M., PALUS, J.A., RUMJANEK, N.G., JANSEN, J.L., NIENHUIS, J. & TRIPLETT, E.W. 1996. Molecular microbial diversity of an agricultural soil in Wisconsin. *Applied and Environmental Microbiology*, **62**, 1935-1943.
BOSSIO, D.A., SCOW, K.M., GUNAPALA, N. & GRAHAM, K.J. 1998. Determinants of soil microbial communities: Effects of agricultural management, season, and soil type on phospholipid fatty acid profiles. *Microbial Ecology*, **36**, 1-12.
BOUCARD, T.K., PARRY, J., JONES, K. & SEMPLE, K.T. 2004. Effects of organophosphate and synthetic pyrethroid sheep dip formulations on protozoan survival and bacterial survival and growth. FEMS Microbiology Ecology **47**, 121-127.

BRELAND, T. A. & ELTUN, R. 1999. Soil microbial biomass and mineralization of carbon and nitrogen in ecological, integrated and conventional forage and arable cropping systems. *Biology and Fertility of Soils*, **30**, 193-201.
BROOKES, P.C. & MCGRATH, S.P. 1984. Effects of Metal Toxicity on the Size of the Soil Microbial Biomass. *Journal of Soil Science*, **35**, 341-346.
BROWN, J.H., MEHLMAN, D.W. and STEVENS, D.G.C. 1995. Spatial variation in abundance. *Ecology*, **76**, 2028-2043.
BRUEHL, G.W. 1987. *Soilborne plant pathogens*. New York: Macmillan.
BRUNEAU, P.M.C., OSTLE, N., DAVIDSON, D.A., GRIEVE, I.C. and FALLICK, A.E. 2002. Determination of rhizosphere C-13 pulse signals in soil thin sections by laser ablation isotope ratio mass spectrometry. *Rapid Communications in Mass Spectrometry*, **16**, 2190-2194.
BRUSSAARD, L. 1998. Soil fauna, guilds, functional groups and ecosystem processes. *Applied Soil Ecology*, **9**, 123-135.
BRUSSAARD, L. ed. 1994. Soil ecology of conventional and integrated farming systems. *Agricultural Ecosystems and Environment*, **51**.
BRUSSAARD, L., BAKKER J-P, & OLLF, H. 1996. Biodiversity of soil biota and plants in abandoned arable fields and grasslands under restoration management. *Biodiversity and Conservation*, **5**, 211-221.
BRUSSAARD, L., BEHAN-PELLETIER, V.M., BIGNELL, D.E., BROWN, V.K., DIDDEN W.A.M., FOLGARAIT, P.I., FRAGOSO, C., FRECKMAN, D.W., GUPTA, V.V.S.R., HATTORI, T., HAWKSWORTH, D., KLOPATEK, C., LAVELLE, P., MALLOCH, D., RUSEK, J., SÖDERSTRÖM, B., TIEDJE, J.M. & VIRGINIA, R.A. 1997. Biodiversity and ecosystem functioning in soil. *Ambio*, **26**, 563-570.
BUCKLEY, D.H., GRABER, J.R., & SCHMIDT, T.M. 1998. Phylogenetic analysis of nonthermophilic members of the kingdom *Crenarchaeota* and their diversity and abundance in soil. *Applied Environmental Microbiology*, **64**, 4333–4339.
BULLUCK, L.R. & RISTAINO, J.B. 2002. Effect of synthetic and organic soil fertility amendments on Southern blight, soil microbial communities and yield of processing tomatoes *Phytopathology*, **92**, 181-189.
BULLUCK, L.R., BROSIUS, M., EVANYLO, G.K., & RISTAINO, J.B. 2002. Organic and synthetic fertility amendments influence soil microbial, physical and chemical properties on organic and conventional farms. *Applied Soil Ecology*, **19**, 147-160.
CAMPER, A. K., HAYES, J.T., STURMAN, P.J., JONES, W.L. & CUNNINGHAM, A.B. 1993. Effects of motility and adsorption rate coefficient on transport of bacteria through saturated porous media. *Applied and Environmental* Microbiology, **59**, 3455-3462.
CARPENTER-BOGGS, L., KENNEDY, A.C. & REGANOLD, J.P. 2000. Organic and biodynamic management: Effects on soil biology. *Soil Science Society of America Journal*, **64**, 1651-1659.
CHAMBERLAIN, D.E., WILSON, J.D. & FULLER, R.J. 1999. A comparison of bird populations on organic and conventional farm systems in southern Britain. *Biological Conservation*, **88**, 307-320.
CHANTIGNY, M.H., ANGERS, D.A., PREVOST, D., VEZINA, L.P.,& CHALIFOUR, F.P. 1997. Soil aggregation and fungal and bacterial biomass under annual and perennial cropping systems. *Soil Science Society of America Journal*, **61**, 262-267.
CHENG, W., COLEMAN, D.C., & BOX, J.E. 1990. Root Dynamics, Production and Distribution in Agroecosystems on the Georgia Piedmont Using Minirhizotrons. *Journal of Applied Ecology*, **27**, 592-604.

CHRISTENSEN, S., GRIFFITHS, B.S., EKELUND, F. & RONN, R. 1992. Huge Increase in Bacterivores on Freshly Killed Barley Roots. *Fems Microbiology Ecology*, **86**, 303-309.
CHRISTIANSEN, K. & BELLINGER, P. 1995. The biogeography of collembola. *Bulletin of Entomology Pologne*, **64**, 279-294.
CLAPPERTON, M.J., CHAN K.Y. & LARNEY, F.J. 2003. Managing the soil habitat for enhanced biological fertility. *In*: *Soil Biological Fertility – A Key to sustainable Land Use in Agriculture.* Eds. L. K. Abbott & D.V. Murphy. Dordrecht, Netherlands: Kluwer Academic. 203-224.
CLAYTON, H., McTAGGART, I.P., PARKER, J., SWAN, L. & SMITH, K.A. 1997. Nitrous oxide emissions from fertilised grassland: A 2 year study of the effects of N fertiliser form and environmental conditions. *Biology and Fertility of Soils*, **23**, 252-260.
CLEGG, C.D. 2006. Impact of cattle grazing and inorganic fertiliser additions to managed grasslands on the microbial community composition of soils. *Applied Soil Ecology*, **31**, 73-82.
COOKE, G.W. 1958. The nation's plant food larder. *Journal of the Science of Food and Agriculture*, **9**, 761-772.
COOKSON, W.R., BEARE, M.H. & WILSON, P. E. 1998. Effects of prior crop residue management on microbial properties and crop residue decomposition. *Applied Soil Ecology*, **7**, 179-188.
COOPER, W.E. & CHILTON, S.J.P. 1950. Studies on antibiotic soil organisms I Actinomycetes antibiotic to *Pythium arrhenomanes* in sugar cane soils of Louisiana. *Phytopathology*, **40**, 544-552.
CORBETT, D.C.M., WINSLOW, R.D., & WEBB, R.M. 1969. Nematode population studies on Broadbalk. 224-228.
CORMACK, W.F. 1997. Testing a stockless arable organic rotation on a fertile soil. *In*: J.E. OLESEN, R. ELTUN, M.J. GOODING, E.S. JENSEN, & U. KOPKE, eds. *Designing and Testing Crop Rotations for Organic Farmin, DARCOF Report No 1. Foulum:DARCOF,* 115-124.
Council of Europe (1972) *European Soil Charter*. Council of Europe:Strasbourg.
COUSIN, C., GRANT, J., DIXON, F., BEYENE, D. & VAN BERKUM, P. 2002. Influence of biosolids compost on the *Bradyrhizobial* genotypes recovered from cowpea and soybean nodules. *Archives of Microbiology*, **177**, 427-430.
CRECCHIO, C., CURCI, M., PIZZIGALLO, M.D.R., RICCIUTI, P. & RUGGIERO, P. 2004. Effects of municipal solid waste compost amendments on soil enzyme activities and bacterial genetic diversity. *Soil Biology & Biochemistry*, **36**, 1595-1605.
CROSSLEY, D.A. Jr, COLEMAN, D.C. & HENDRIX, P.F. 1989. The importance of the fauna in agricultural soils.: research approaches and perspectives. *Agriculture Ecosystems and Environment*, **27**, 47-55.
CROSSLEY, D.A. Jr, MUELLER, B.R. & PERDUE, J.C. 1992. Biodiversity of microarthropods in agricultural soils: relations to processes. *Agriculture Ecosystems and Environment*, **40**, 37-46.
DALIAS, P., KOKKORIS, G.D. & TROUMBIS, A.Y. 2003. Functional shift hypothesis and the relationship between temperature and soil carbon accumulation. *Biology and Fertility of Soils*, **37**, 90-95.
DAVIDSON, D.A., BRUNEAU, P.M.C., GRIEVE, I.C. & YOUNG, I.M. 2002. Impacts of fauna on an upland grassland soil as determined by micromorphological analysis. *Applied Soil Ecology*, **20**, 133-143.

Defra 2002. Policy Commission Report on the Future of Farming and Food http://archive.cabinetoffice.gov.uk/farming/pdf/PC Report2.pdf Page last modified: not given. Viewed 10 May 2006

Defra 2003a. Compendium of organic standards http://www.defra.gov.uk/farm/organic/legislation-standards/index.htm Page last modified: 3 May 2006. Viewed 10 May 2006.

Defra 2003b. Action Plan to Develop Organic Food and Farming in England. http://www.defra.gov.uk/farm/organic/actionplan/index.htm Page last modified: 7 February 2005. Viewed 1 March 2006

Defra 2004a. Soil Action Plan for England http://www.defra.gov.uk/environment/land/soil/sap/index.htm Page last modified: 6 April 2006 Viewed 10 May 2006.

Defra 2004b. British Survey of Fertiliser Practice *Fertiliser use on farm crops 2003*. Defra: London.

Defra 2005. Organic land statistics. http://statistics.defra.gov.uk/esg/statnot/orguk.pdf Page last modified 25 October 2005, viewed 1st March 2006

DE GOEDE, R.G.M., BRUSSAARD, L. & AKKERMANS A.D.L. 2003. On-farm impact of cattle slurry manure management on biological soil quality. *Netherlands Journal of Agricultural Science*, **51**, 103-133.

DE RUITER, P.C., MOORE, J.C., ZWART, K.B., BOUWMAN, L.A., HASSINK, J., BLOEM, J., DEVOS, J.A., MARINISSEN, J.C.Y., DIDDEN, W.A.M., LEBBINK, G. & BRUSSAARD, L. 1993. Simulation of Nitrogen Mineralization in the Belowground Food Webs of 2 Winter-Wheat Fields. *Journal of Applied Ecology*, **30**, 95-106.

DE RUITER, P.C., NEUTEL, A.N. & MOORE, J.C. 1994. Modelling food webs and nutrient cycling in agroecosystems. *Trends in Ecology and Evolution*, **9**, 378-383.

DI, H.J. & CAMERON, K.C. 2004. Effects of temperature and application rate of a nitrification inhibitor, dicyandiamide (DCD), on nitrification rate and microbial biomass in a grazed pasture soil. *Australian Journal of soil Research*, **42**, 927-932.

DICK, R.P. 1992. A Review - Long-Term Effects of Agricultural Systems on Soil Biochemical and Microbial Parameters. *Agriculture Ecosystems & Environment*, **40**, 25-36.

DICK, W.A. & GREGORICH, E.G. 2004. Developing and maintaining soil organic matter levels. *In* : P. SCHJONNING, S. ELMHOLT & B.T. CHRISTENSEN ed. *Managing soil quality. Challenges in modern agriculture.* Wallingford: CABI Publishing, 103-120.

DIDDEN, W.A.M.1993. Ecology of terrestrial Enchytraeidae. *Pedobiologica*, **37**, 2-29.

DONNISON, L.M., GRIFFITH, G.S., & BARDGETT, R. D. 2000. Determinants of fungal growth and activity in botanically diverse hay meadows: effects of litter type and fertilizer additions. *Soil Biology & Biochemistry*, **32**, 289-294.

DORAN, J.W. & SMITH, M.S. 1987. Organic Matter Management and Utilization of soil and Fertilizer Nutrients. In: *Soil Fertility and Organic Matter as Critical Components of Production Systems*. Soil Science Society of America Special Publication 19. Madison, Wisconsin: Soil Science Society of America and American Society of Agronomy, 53-72.

DORAN, J.W. & LINN, D.M. 1994. Microbial ecology of conservation management systems. *In* : J. L. HATFIELD & B. A. STEWART ed. *Soil biology effects on soil quality.* Boca Raton: Lewis Publishers, 1-27.

DORING, T.F. & KROMP, B. 2003. Which carabid species benefit from organic agriculture? - a review of comparative studies in winter cereals from Germany and Switzerland. *Agriculture Ecosystems & Environment*, **98**, 153-161.
DOS SATOS, J.B., FERREIRA, E.A., KASUYA, M.C.M., DA SILVA, A.A. & PROCOPIO, S.D.O. 2005. Tolerance of Bradyrhizobium strains to glyphosate formulations. *Crop Protection*, **24**, 543-547.
DOUDS D.D. & MILNER, P.D.1999. Biodiversity of arbuscular mycorrhizal fungi in agroecosystems. *Agriculture Ecosystems and Environment*, **74**, 77-93.
DOUDS, D.D., GALVEZ, L., JANKE, R.R. & WAGONER, P. 1995. Effect of tillage and farming system upon populations and distribution of vesicular-arbuscular mycorrhizal fungi. *Agriculture Ecosystems & Environment*, **52**, 111-118.
DRINKWATER, L.E., WAGGONER, P. & SARRANTONIO M. 1998. Legume based cropping systems have reduced carbon and nitrogen losses. *Nature*, **396**, 262-265.
DUFFY, B.K., OWNLEY, B.H. & WELLER, D.M. 1997. Soil chemical and physical properties associated with suppression of take-all of wheat by Trichoderma koningii. *Phytopathology*, **87**, 1118-1124.
EASON, W.R., SCULLION, J., & SCOTT E.P. 1999. Soil parameters and plant responses associated with arbuscular mycorrhizas from contrasting grassland management regimes. *Agriculture Ecosystems & Environment*, **73**, 245-255.
EDWARDS, C.A. ed. 1998. *Earthworm Ecology*. New York, NY: St Lucie Press.
EIJSACKERS, H., BENEKE, P., MABOETA, M, LOUW, J.P.E. & REINECKE, A.J. 2005. The implications of copper fungicide use in vineyards for earthworm activity and resulting sustainable soil quality. *Ecotoxicology and Environmental Safety*, **62**, 99-111.
EKSCHMITT, K. & GRIFFITHS, B.S. 1998. Soil biodiversity and its implications for ecosystem functioning in a heterogeneous and variable environment. *Applied Soil Ecology*, **10**, 201-215.
EL FANTROUSSI, S., VERSCHUERE, L., VERSTRAETE, W., & TOP, E.M. 1999. Effect of phenylurea herbicides on soil microbial communities estimated by analysis of 16S rRNA gene fingerprints and community-level physiological profiles. *Applied and Environmental Microbiology*, **65**, 982-988.
ELMHOLT, S. & LABOURIAU, R. 2005. Fungi in Danish soils under organic and conventional farming. *Agriculture Ecosystems & Environment*, **107**, 65-73.
ELTUN, R. 1994. The Apelsvoll cropping system experiment. I. Background, objectives and methods. *Norwegian Journal of Agricultural Sciences*, **8**, 301-315.
ENDLWEBER, K., SCHADLER, M., & SCHEU, S. 2006. Effects of foliar and soil insecticide applications on the collembolan community of an early set-aside arable field. *Applied Soil Ecology*, **31**, 136-146.
ENNIS, W.B. 1979. *Introduction to crop protection.* Madison, Wisconsin: American Society of Agronomy.
ENTRY, J.A., REEVES, D.W., MUDD, E., LEE, W.J., GUERTAL, E., & RAPER, R.L. 1996. Influence of compaction from wheel traffic and tillage on arbuscular mycorrhizae infection and nutrient uptake by Zea mays. *Plant and Soil*, **180**, 139-146.
ENVIRONMENT AGENCY 2004. State of Soils in England and Wales http://www.environmentagency.gov.uk/subjects/landquality/776051/775200/775473/?version=1&lang=_e Last modified not given, viewed 1 March 2006.
ETTEMA, C.H. & WARDLE, D.A. 2002. Spatial soil ecology. *Trends in Ecology and Evolution*, **17**, 177-183.

EUROPEAN COMMISSION 2002. EU Thematic Strategy for Soil Protection http://ec.europa.eu/comm/environment/soil/index.htm Page last modified 27 February 2006. Viewed 1 March 2006

EUROPEAN COMMISSION 2004. 'European Action Plan for Organic Food and Farming' http://europa.eu.int/comm/agriculture/qual/organic/plan/index_en.htm Last modified, not given, viewed 1 March 2006.

FEBER, R.E., FIRBANK, L.G., JOHNSON, P.J., MACDONALD, D.W. 1997. The effects of orgnaic farming on pesst and non-pest buterfly abundance. *Agricultural Ecosystems & Environment*, **64**, 133-139.

FERNANDEZ, N. 2005. Spatial patterns in European rabbit abundance after a population collapse. *Landscape Ecology*, **20**, 897-910.

FERRIS, H., VENETTE, R.C. & SCOW, K.M. 2004. Soil management to enhance bacterivore and fungivore nematode populations and their nitrogen mineralisation function. *Applied Soil Ecology*, **25**, 19-35.

FILSER, J., FROMM, H., NAGEL, R. & WINTER, K. 1995. Effects of previous intensive agricultural managmenet on microorganisms and the biodiversity of soil fauna. *Plant and Soil*, **170**, 123-129.

FILSER, J., MEBES, K. H., WINTER, K., LANG, A., & KAMPICHLER, C. 2002. Long-term dynamics and interrelationships of soil Collembola and microorganisms in an arable landscape following land use change. *Geoderma*, **105**, 201-221.

FITTER, A.H. 2005. Darkness visible: reflections on underground ecology. *Journal of Ecology*, **93**, 231-243.

FITTER, A.H., GILLIGAN, C.A., HOLLINGWORTH, K., KLECZKOWSKI, A., TWYMAN, R.M., PITCHFORD J.W. and the members of the NERC Soil Biodiversity Programme. 2005. Biodiversity and ecosystem function in soil. *Functional Ecology*, **19**, 369-377.

FLIESSBACH, A. & MADER, P. 1997. Carbon source utilization by microbial communities in soils under organic and conventional farming practice. *In*: H. INSAM & A. RANGGER, eds. *Microbial Communities- Functional versus structural approaches*. Berlin: Springer-Verlag, pp109-120.

FOISSNER, W. 1992. Comparative-Studies on the Soil Life in Ecofarmed and Conventionally Farmed Fields and Grasslands of Austria. *Agriculture Ecosystems & Environment*, **40**, 207-218.

FOISSNER, W. 1997. Protozoa as bioindicators in agroecosystems, with emphasis on farming practices, biocides, and biodiversity. *Agriculture Ecosystems & Environment*, **62**, 93-103.

FORTUNE, S., LU, J., ADDISCOTT, T.M. & BROOKES, P.C. 2005. Assessment of phosphorus leaching losses from arable land. *Plant and Soil*, **269**, 99-108.

FOX, C.J.F. 1964. The effects of five herbicides on the numbers of certain invertebrate animals in grassland soil. *Canadian Journal of Plant Science*, **44**, 405-409.

FRAMPTON, G.K. & ÇILGI, T. 1996. How do arable rotations influence pesticide side-effects on arthropods? *Aspects of Applied Biology*, **47**, 127-135.

FRANKE-SNYDER, M., DOUDS, D.D., GALVEZ, L., PHILLIPS, J.G., WAGONER, P., DRINKWATER, L., & MORTON, J.B. 2001. Diversity of communities of arbuscular mycorrhizal (AM) fungi present in conventional versus low-input agricultural sites in eastern Pennsylvania, USA. *Applied Soil Ecology*, **16**, 35-48.

FRANKLIN, R.B. & MILLS, A. L. 2003. Multi-scale variation in spatial heterogeneity for microbial community structure in an eastern Virginia agricultural field. *FEMS Microbiology Ecology*, **44**, 335-346.

FROST, D. MCLEAN, B.M.L. & EVANS, D.E. 2002. Eight years of organic farming at Pwllpeiran – livestock production and the financial performance of organic upland farms. *UK Organic Research 2002: Proceedings of the COR conference.* Aberystwyth: Organic Centre Wales, 259-262.

FULLER, R.J., NORTON, L.R., FEBER, R.E., JOHNSON, P.J., CHAMBERLAIN, D.E., JOYS, A.C., MATHEWS, F., STUART, R C., TOWNSEND, M.C., MANLEY, W.J., WOLFE, M.S., MACDONALD, D.W., & FIRBANK, L.G. 2005. Benefits of organic farming to biodiversity vary among taxa. *Biology Letters*, **1**, 431-434.

FULTHORPE, R.R., RHODES, A.N. & TIEDJE, J.M. 1998. High levels of endemicity of 3-chlorobenzoate-degrading soil bacteria. *Applied and Environmental Microbiology*, **64**, 1620-1627.

GARRETT, S.D. 1956. *Biology of root-infecting fungi.* Cambridge: Cambridge University Press.

GARTHWAITE, D.G., THOMAS, M.R., DAWSON, A. & STODDART, H. 2003. *Pesticide Usage Survey Group Report 187. Arable Crops in Great Britain*. Defra, SEERAD. Accessed on 25 February 2006 at: www.csl.gov.uk/science/organ/pvm/puskm/arable2002.pdf

GERMIDA, J.J., SICILIANO, S.D., DE FREITAS, J.R., & SEIB, A.M. 1998. Diversity of root-associated bacteria associated with held-grown canola (Brassica napus L.) and wheat (Triticum aestivum L.). *FEMS Microbiology Ecology*, **26**, 43-50.

GILLER, K.E., BEARE, M.H., LAVELLE, P., IZAC, A.M.N., & SWIFT, M.J. 1997. Agricultural intensification, soil biodiversity and agroecosystem function. *Applied Soil Ecology*, **6**, 3-16.

GILLER, K.E., WITTER, E., & MCGRATH, S.P. 1998. Toxicity of heavy metals to microorganisms and microbial processes in agricultural soils: A review. *Soil Biology & Biochemistry*, **30**, 1389-1414.

GIRVAN, M.S., BULLIMORE, J., PRETTY, J.N., OSBORN, A.M., & BALL, A.S. 2003. Soil type is the primary determinant of the composition of the total and active bacterial communities in arable soils. *Applied and Environmental Microbiology*, **69**, 1800-1809.

GOSLING, P., HODGE, A., GOODLASS, G., & BENDING, G.D. 2006. Arbuscular mycorrhizal fungi and organic farming. *Agriculture, Ecosystems & Environment*, **113**, 17-35.

GRAYSTON, S.J., WANG, S.Q., CAMPBELL, C.D., & EDWARDS, A.C. 1998. Selective influence of plant species on microbial diversity in the rhizosphere. *Soil Biology & Biochemistry*, **30**, 369-378.

GREENLAND, D.J. 1977. Soil damage by intensive arable cultivation – temporary or permanent. *Philosophical Transactions of the Royal Society Series B*, **281**, 193-208.

GREGORY, P.J. 2006. Roots, rhizosphere and soil: the route to a better understanding of soil science? *European Journal of Soil Science*, **57**, 2-12.

GRIFFITHS, B.S. 1994. Microbial feeding nematodes and protozoa in soil and their effects on microbial activity and nitrogen mineralisation in decomposition hotspots and the rhizosphere. *Plant and Soil*, **164**, 25-33.

GRIFFITHS, B S., WELSCHEN, R., VANARENDONK, J.J.C.M., & LAMBERS, H. 1992. The effect of nitrate-nitrogen supply on bacteria and bacterial-feeding fauna in the rhizosphere of different grass species. *Oecologia*, **91**, 253-259.

GRIFFITHS, B.S., RITZ, K., & WHEATLEY, R.E. 1994. Nematodes as indicators of enhanced microbiological activity in a Scottish organic farming system. *Soil Use and Management*, **10**, 20-24.

GRIFFITHS, B.S., WHEATLEY, R.E., OLESEN, T., HENRIKSEN, K., EKELUND, F., & RONN, R. 1998. Dynamics of nematodes and protozoa following the experimental addition of cattle or pig slurry to soil. *Soil Biology & Biochemistry*, **30**, 1379-1387.

GRIFFITHS, B.S., RITZ, K., BARDGETT, R.D., COOK, R., CHRISTENSEN, S., EKELUND, F., SØRENSEN, S.J., BÅÅTH, E., BLOEM, J., DE RUITER, P.C., DOLFING, J. & NICOLARDOT, B. 2000. Ecosystem response of pasture soil communities to fumigation-induced microbial diversity reductions: An examination of the biodiversity-ecosystem function relationship. *Oikos*, **90**, 279-294.

GRIFFITHS, B.S., BONKOWSKI, M., ROY, J., & RITZ, K. 2001. Functional stability, substrate utilisation and biological indicators of soils following environmental impacts. *Applied Soil Ecology*, **16**, 49-61.

GRIFFITHS, B.S., KUAN, H.L., RITZ, K., GLOVER, L.A., McCAIG, A.E. & FENWICK, C. 2004. The relationship between microbial community structure and functional stability, tested experimentally in an upland pasture soil. *Microbial Ecology*, **47**, 104-113.

GROFFMAN, P.M. & BOHLEN, P.J. 1999. Soil and sediment biodiversity: cross-system comparisons and large scale effects. *BioScience*, **49**, 139-148.

GROFFMAN, P.M., EAGAN, P., SULLIVAN, W.M., & LEMUNYON, J.L. 1996. Grass species and soil type effects on microbial biomass and activity. *Plant and Soil*, **183**, 61-67.

GUNAPALA, N. & SCOW, K.M. 1998. Dynamics of soil microbial biomass and activity in conventional and organic farming systems. *Soil Biology & Biochemistry*, **30**, 805-816.

HAMEL, C., DALPÉ, Y., LAPIERRE, C., SIMARD, R.R. & SMITH, D.L. 1994. Composition of the vesicular-arbuscular mycorrhizal fungal population in an old meadow as affected by pH, phosphorus and soil disturbance. *Agriculture, Ecosystems & Environment*, **49**, 223-231.

HAMILTON, E.W. & FRANK, D.A. 2001. Can plants stimulate soil microbes and their own nutrient supply? Evidence from a grazing tolerant grass. *Ecology*, **82**, 2397-2402.

HANEY, R.L., SENSEMAN, S.A., & HONS, F.M. 2002. Effect of roundup ultra on microbial activity and biomass from selected soils. *Journal of Environmental Quality*, **31**, 730-735.

HANSSON, A. 1990. Structure of the agroecosystem. *Ecological Bulletins*, **40**, 41-83.

HARRIER, L.A. & WATSON, C.A. 2003. The role of arbuscular mycorrhizal fungi in sustainable cropping systems. *Advances in Agronomy*, **79**, 185-225.

HART, M.R. & BROOKES, P.C. 1996. Soil microbial biomass and mineralisation of soil organic matter after 19 years of cumulative field applications of pesticides. *Soil Biology & Biochemistry*, **28**, 1641-1649.

HART, S. C., NASON, G. E., MYROLD, D. D. & PERRY, D. A. 1994. Dynamics of gross nitrogen transformations in an old-growth forest: the carbon connection. *Ecology*, **75**, 880-891.

HASSINK, J., BOUWMAN, L.A., ZWART, K.B., BLOEM, J. & BRUSSAARD, L. 1993. Relationships between soil texture, physical protection of organic-matter, soil biota, and C-mineralization and N-mineralization in grassland soils. *Geoderma*, **57**, 105-128.

HAYGARTH, P.M., CHAPMAN P.J., JARVIS, S.C. & SMITH, R.V. 1998. Phosphorus budgets for two contrasting grassland farming systems in the UK. *Soil Use and Management*, **14**, 160-167.

HECKRATH, G., BROOKES, P.C., POULTON, P.R. & GOULDING, K.W.T. 1995. Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk experiment. *Journal of Environmental Quality*, **24**, 904-910.
HENDRIKSEN, N.B. 1990. Leaf litter selection by detritivore and geophagous earthworms. *Biology and Fertility of Soils*, **10**, 17-21.
HENRIKSEN, T.M. & BRELAND, T.A., 1999. Decomposition of crop residues in the field: evaluation of a simulation model developed from microcosm studies. *Soil Biology & Biochemistry,* **31**, 1423-1434.
HEYWOOD, V.H. ed. 1995. *Global biodiversity assessment*. Cambridge: Cambridge University Press.
HIGGINBOTHAM, S., NOBLE, L. & JOICE, R. 1996. The profitability of integrated crop management, organic and conventional arable regimes. *Aspects of Applied Biology*, **47**, 327-333.
HIGGINBOTHAM, S., LEAKE, A R., JORDAN, V W., & OGILVY, S.E. 2000. Environmental and ecological aspects of integrated, organic and conventional farming systems. *Aspects of Applied Biology*, **62**, 15-20.
HILL, J. 2005. Recycling biosolids to pasture-based animal production systems in Australia: a review of evidence on the control of potentially toxic metals and persistent organic compounds recycled to agricultural land. *Australian Journal of Agricultural Research*, **56**, 753-773
HILLOCKS, R.J. & WALLER, R.M. eds 1997. *Soilborne disease of tropical crops.* Wallingford, UK: CAB International.
HIRSCH, P.R., JONES, M.J., MCGRATH, S.P., & GILLER, K.E. 1993. Heavy-Metals from Past Applications of Sewage-Sludge Decrease the Genetic Diversity of Rhizobium-Leguminosarum Biovar Trifolii Populations. *Soil Biology & Biochemistry*, **25**, 1485-1490.
HOLE, D.G., PERKINS, A.J., WILSON, J. D., ALEXANDER, I.H., GRICE, F., & EVANS, A.D. 2005. Does organic farming benefit biodiversity? *Biological Conservation*, **122**, 113-130.
HOLLAND, J.M., DRYSDALE, A., HEWITT, M.V. & TURLEY, D. 1996. The LINK-IFS project: the effect of crop rotations and cropping systems on Carabidae. *Aspects of Applied Biology*, **47**, 119-126.
HOOPER, D.U., CHAPIN, F.S., EWEL, J.J., HECTOR, A., INCHAUSTI, P., LAVOREL, S., LAWTON, J.H., LODGE, D.M., LOREAU, M., NAEEM, S., SCHMID, B., SETALA, H., SYMSTAD, A.J., VANDERMEER, J., & WARDLE, D. A. 2005. Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. *Ecological Monographs*, **75**, 3-35.
HOUOT, S. & CHAUSSOD, R. 1995. Impact of agricultural practices on the size and activity of the microbial biomass in a long-term field experiment. *Biology and Fertility of Soils*, **19**, 309-316.
HOWELL, C.R. 1982. Effect of *Gliocladium virens* on *Pythium ultium*, *Rhizoctonia solani* and damping off of cotton seedlings. *Phytopathology*, **72**, 496-498.
HUNT, H.W., COLEMAN, D.C., INGHAM, E.R., ELLIOTT, E.T., MOORE, J.C., ROSE, S.L., REID, P.P., & MORLEY C R. 1987. The detrital food web in a shortgrass prairie. *Biology and Fertility of Soils*, **3**, 57-68.
HUTCHINSON, K.J. & KING K.L. 1980. The effects of sheep stocking level on invertebrate abundance, biomass and energy-utilisation in a temperate, sown grassland. *Journal of Applied Ecology,* **17**, 369-387.

HUTTON, S. A. & GILLER, P. S. 2003. The effects of the intensification of agriculture on northern temperate dung beetle communities. *Journal of Applied Ecology*, **40**, 994-1007.
IFOAM 2006. The principles of organic agriculture. http://www.ifoam.org/about_ifoam/principles/index.html Last modified: not given. Viewed 1 March 2006.
INGRAM, J.S.I. & FERNANDES, E.C.M. 2001. Managing carbon sequestration in soils: concepts and terminology. *Agriculture Ecosystems & Environment*, **87**, 111-117.
JACINTHE, P.A., LAL, R. & KIMBLE, J.M. 2002. Carbon budget and seasonal carbon dioxide emission from a central Ohio Luvisol as influenced by wheat residue amendment. *Soil & Tillage Research*, **67**, 147-157.
JACKSON, L.E., STIVERS, L.J., WARDEN, B.T., & TANJI, K.K. 1994. Crop nitrogen-utilization and soil nitrate loss in a lettuce field. *Fertilizer Research*, **37**, 93-105.
JACKSON, L.E., RAMIREZ, I., YOKOTA, R., FENNIMORE, S.A., KOIKE, S.T., HENDERSON, D.M., CHANEY, W.E., CALDERÓN, F.J. KLONSKY, K. 2004. On-farm assessment of organic matter and tillage management on vegetable yield, soil, weeds, pests and economics in California. *Agriculture, Ecosystems & Environment*, **103**, 443-463.
JANSA, J., MOZAFAR, A., KUHN, G., ANKEN, T., RUH, R., SANDERS, I.R., & FROSSARD, E. 2003. Soil tillage affects the community structure of mycorrhizal fungi in maize roots. *Ecological Applications*, **13**, 1164-1176.
JANNSEN, B.H. 1984. A simple method for calculating decomposition and accumulation of 'young' soil organic matter. *Plant and Soil*, **76**, 297-304.
JENKINSON, D.S. 2001. The impact of humans on the nitrogen cycle, with focus on temperate arable agriculture. *Plant and Soil*, **228** 3-15.
JENSEN, J., KROGH, P.H., & SVERDRUP, L.E. 2003. Effects of the antibacterial agents tiamulin, olanquindox and metronidazole and the anthelmintic ivermectin on the soil invertebrate species Folsomia fimetaria (Collembola) and Enchytraeus crypticus (Enchytraeidae). *Chemosphere*, **50**, 437-443.
JENSEN, U.B., ELMHOLT, S., & LABOURIAU, R. 2000. Distribution of ergosterol in organically and conventionally cultivated agricultural soils. *Biological Agriculture & Horticulture*, **18**, 113-125.
JOHNSON, N.C., COPELAND, P.J., CROOKSTON, R.K., & PFLEGER, F.L. 1992. Mycorrhizae - Possible Explanation for Yield Decline with Continuous Corn and Soybean. *Agronomy Journal*, **84**, 387-390.
JOHNSON, D., LEAKE, J.R. & READ, D.J. 2005. Liming and nitrogen fertilisation affects phosphatase activities, microbial biomass and mycorrhizal colonisation in upland grassland. *Plant and Soil*, **271**, 157-164.
JOHNSTON, A.E. & POWLSON, D.S. 1994. The setting-up, conduct and applicability of long-term continuing field experiments in agricultural research. In: GREENLAND D. J. & SZABOLCS, I. Eds. *Soil Resilience and Sustainable Land Use*. Wallingford: CAB International. 395-421.
JONES, C.G., LAWTON, J.H. & SHACHAK, M. 1994. Organisms as ecosystem engineers. *Oikos*, **69**, 373-386.
JONES, D.L., HODGE, A. & KUZYAKOV, Y. 2004. Plant and mycorrhizal regulation of rhizodeposition. *New Phytologist*, **163**, 459-480.
JONES, D.L., SHANNON, D.,MURPHY, D.V. & FARRAR, J. 2004b. Role of dissoloved orgnaic nitrogen (DON) in soil N cycling in grassland soils. *Soil Biology & Biochemistry*, **36**, 749-756.

JONSSON, S. 2002. Crop yields in organic and conventional production – studies from the Öjebyn project. *UK Organic Research 2002: Proceedings of the COR conference.* Aberystwyth: Organic Centre Wales. 43-46.
JORDAN, V.W.L., LEAKE, A.R. & OGILVY, S. 2000. Agronomic and environmental implications of soil management practices in integrated farming systems. *Aspects of Applied Biology*, **62**, 55-61.
KAHILUOTO, H. & VESTBERG, M. 1998. The effect of arbuscular mycorrhiza on biomass production and phosphorus uptake from sparingly soluble sources by leek (*Allium porrum L.*) in Finnish field soils. *Biological Agriculture and Horticulture*, **16**, 65-85.
KAY, B.D. & VANDENBYGAART, A.J. 2002. Conservation tillage and depth stratification of porosity and soil organic matter. *Soil & Tillage Research*, **66**, 107-118.
KEEL, C., SCHNIDER, U., MAURHOFER, M., VOISARD, C., LAVILLE, J., BURGER, U., WIRTHNER, P., HAAS, D. & DEFAGO, G. 1992. Suppression of root diseases by Pseudomonas-fluorescens CHAO. Importance of the bacterial secondary metabolite 2,4-diacetylphloroglucinol. *Molecular Plant-Microbe Interactions*, **5**, 4-13.
KERRY, B. 1981. Fungal parasites: a weapon against cyst-nematodes. *Plant Disease*, **65**, 390-393.
KILLHAM, K. 1994. *Soil Ecology*. Cambridge: Cambridge University Press
KING, K.L. & HUTCHINSON, K.J. 1976. Effects of Sheep Stocking Intensity on Abundance and Distribution of Mesofauna in Pastures. *Journal of Applied Ecology*, **13**, 41-55.
KIRCHMANN, H. & GERZABEK, M.H. 1999. Relationship between soil organic matter and micropores in a long-term experiment at Ultuna, Sweden. *Journal of Plant Nutrition and Soil Science-Zeitschrift fur Pflanzenernahrung und Bodenkunde*, **162**, 493-498.
KLOSE, S. & AJWA, H.A. 2004. Enzyme activities in agricultural soils fumigated with methyl bromide alternatives. *Soil Biology & Biochemistry*, **36**, 1625-1635.
KNOWLES, R. 1982. Denitrification. *Microbiological Reviews*, **46**, 43-70.
KOOISTRA, M.J., LEBBINK, G. & BRUSSAARD, L. 1989. The Dutch programme on soil , ecology of arable farming systems. 2. Geogenesis, agricultural history field site characteristics and present farming systems at the Lovinkhoeve experimental farm. *Agriculture, Ecosystems & Environment*, **27**, 361-387.
KOWALCHUK, G.A., STIENSTRA, A.W., HEILIG, G.H.J., STEPHEN, J. R. & WOLDENDORP, J.W. 2000. Changes in the community structure of ammonia-oxidizing bacteria during secondary succession of calcareous grasslands. *Environmental Microbiology*, **2**, 99-110.
KREBS, J.R., WILSON, J.D., BRADBURY, R.B., & SIRIWARDENA, G.M. 1999. The second silent spring? *Nature*, **400**, 611-612.
KURLE, J.E. & PFLEGER, F.L. 1996. Management influences on arbuscular mycorrhizal fungal species composition in a corn-soybean rotation. *Agronomy Journal*, **88**, 155-161.
LAVELLE, P. 2000. Ecological challenges for soil science. *Soil Science*, **165**, 73-86.
LAVELLE, P., BLANCHART, E., MARTIN, A., MARTIN, S., SPAIN A., TOUTAIN, F., BAROIS I. & SCHAEFER R. 1993. A hierarchical model for decomposition in terrestrial ecosystems: application to soils of the humid tropics. *Biotropica*, **25**, 130-150.
LAVELLE, P., BARROS, E., BLANCHART, E., BROWN, G., DESJARDINS T, MARIANI, L. & ROSSI, J-P. 2001. SOM management in the tropics:Why feeding the soil macrofauna? *Nutrient Cycling in Agroecosystems*, **61**, 53-61.

LAWLOR, K., KNIGHT, B.P., BARBOSA-JEFFERSON, V.L., LANE, P.W., LILLEY, A.K., PATON, G.I., MCGRATH, S.P., O'FLAHERTY, S.M., & HIRSCH, P.R. 2000. Comparison of methods to investigate microbial populations in soils under different agricultural management. *FEMS Microbiology Ecology*, **33**, 129-137.

LEAKE, A.R. 1996. The effect of cropping sequences and rotational management: an economic comparison of conventional, integrated and organic systems. *Aspects of Applied Biology*, **47**, 185-195.

LEAKE, J.R., JOHNSON, D., DONNELLY, D.P., MUCKLE, G.E., BODDY, L. & READ, D.J. 2004. Networks of power and influence: the role of mycorrhizal mycelium in controlling plant communities and agroecosystem functioning. *Canadian Journal of Botany*, **82**, 1016-1045.

LOREAU, M., NAEEM, S., INCHAUSTI, P., BENGTSSON, J., GRIME, J.P., HECTOR, A., HOOPER, D.U., HUSTON, M.A., RAFFAELLI, D., SCHMID, B., TILMAN, D. & WARDLE, D.A. 2001. Biodiversity and ecosystem functioning: current knowledge and future challenges. *Science*, **294**, 804-808.

LOUREIRO, S., SOARES, A.M.V.M. & NOGUEIRA, A.J.A. 2005. Terrestrial avoidance behaviour tests as screening tool to assess soil contamination. *Environmental Pollution*, **138**, 121-131.

LOVELAND, P.J. & THOMPSON, T.R.E. 2002. Identification and development of a set of national indicators for soil quality. Environment Agency R&D Project Record P5-053/PR/02. Bristol: Environment Agency.

LUPWAYI, N.Z., CLAYTON, G.W., HANSON, K.G., RICE, W.A. & BIEDERBECK, V.O. 2004. Populations and functional diversity of bacteria associated with barley, wheat and canola roots. *Canadian Journal of Soil Science*, **84**, 245-254.

LUSSENHOP, P. 1992. Mechanisms for microarthropod – microbial interactions. *Advances in Ecological Research*, **23**, 1-33.

LUXTON, M. 1996. Oribatid mites of the British Isles: A check-list and notes on biogeography (Acari: Oribatida). *Journal of Natural History*, **30**, 803-822.

LYNCH, J.M., BENEDETTI, A., INSAM, H., NUTI M.P., SMALLA, K., TORSVIK, V. & NANNIPIERI, P. 2004. Microbial diversity in soil: ecological theories, the contribution of molecular techniques and the impact of transgenic plants and transgenic microorganisms. *Biology and Fertility of Soils*, **40**, 363–385.

MACAULAY INSTITUTE 2006. Grazing and Upland Birds. www.macaulay.ac.uk/projects/projectdetails.php?302797 Last updated, not given. Viewed, 13 March 2006.

MACDONALD, L.M., PATERSON, E., DAWSON, L.A., & MCDONALD, A.J.S. 2004. Short-term effects of defoliation on the soil microbial community associated with two contrasting Lolium perenne cultivars. *Soil Biology & Biochemistry*, **36**, 489-498.

MÄDER, P., EDENHOFER, S., BOLLER, T., WIEMKEN, A. & NIGGLI, U. 2000. Arbuscular mycorrhizae in a long-term field trial comparing low-input (organic, biological) and high-input (conventional) farming systems in a crop rotation. *Biology and Fertility of Soils*, **31**, 150-156.

MÄDER, P., FLIESSBACH, A., DUBOIS, D., GUNST, L., FRIED, P., & NIGGLI, U. 2002. Soil fertility and biodiversity in organic farming. *Science*, **296**, 1694-1697.

MARILLEY, L. & ARAGNO, M. 1999. Phylogenetic diversity of bacterial communities differing in degree of proximity of Lolium perenne and Trifolium repens roots. *Applied Soil Ecology*, **13**, 127-136.

MARSCHNER, P., YANG, C. H., LIEBEREI, R., & CROWLEY, D.E. 2001. Soil and plant specific effects on bacterial community composition in the rhizosphere. *Soil Biology & Biochemistry*, **33**, 1437-1445.
MARSCHNER, P., KANDELER, E. & MARSCHNER, B. 2003. Structure and function of the soil microbial community in a long-term fertilizer experiment. *Soil Biology & Biochemistry*, **35**, 453-461.
MARSCHNER, P., CROWLEY, D., & YANG, C.H. 2004. Development of specific rhizosphere bacterial communities in relation to plant species, nutrition and soil type. *Plant and Soil*, **261**, 199-208.
MAZZOLA, M. 2002. Mechanisms of natural soil suppressiveness to soil borne diseases. *Antonie van Leeuwenhoek International Journal of General and Molecular Microbiology*, **81**, 557-564.
MCCAIG, A.E., GLOVER, L.A., & PROSSER, J.I. 1999. Molecular analysis of bacterial community structure and diversity in unimproved and improved upland grass pastures. *Applied and Environmental Microbiology*, **65**, 1721-1730.
MCINNES, A. & HAQ, K. 2003. Contributions of Rhizobia to soil nitrogen fertility. *In* : L. K. ABBOTT & D.V. MURPHY ed. *Soil Biological Fertility – A Key to Sustainable Land Use in Agriculture.* Dordrecht: Kluwer, 99-128.
MCLAUGHLIN, M.J., HAMON, R.E., MCLAREN, R.G., SPEIR, T.W., ROGERS, S.L. 2000. Review: A bioavailability based rationale for controlling metal and metalloid contamination of agricultural land in Australia and New Zealand. *Australian Journal of Soil Research*, **38**, 1037-1086.
MDC 2000. Efficient use of grazed herbage by dairy cows – Maximising the contribution of nutrients from herbage. *MDC Project Report* 97/R1/13
MEEK, B.D., RECHEL, E.A., CARTER, L.M., & DETAR, W.R. 1989. Changes in Infiltration Under Alfalfa As Influenced by Time and Wheel Traffic. *Soil Science Society of America Journal*, **53**, 238-241.
MENDES, I.C. & BOTTOMLEY, P.J. 1998. Distribution of a population of Rhizobium leguminosarum bv. trifolii among different size classes of soil aggregates. *Applied and Environmental Microbiology*, **64**, 970-975.
MENDUM, T.A. & HIRSCH, P.R. 2002. Changes in the population structure of beta-group autotrophic ammonia oxidising bacteria in arable soils in response to agricultural practice. *Soil Biology & Biochemistry*, **34**, 1479-1485.
MENZIES, J.D. 1959. Occurrence and transfer of a biological factor in soil that suppresses potato scab. *Phytopathology*, **49**, 648-652.
MERRINGTON, G., ROGERS, S.L., & VAN ZWIETEN, L. 2002. The potential impact of long-term copper fungicide usage on soil microbial biomass and microbial activity in an avocado orchard. *Australian Journal of Soil Research*, **40**, 749-759.
MIKOLA, J., BARDGETT, R.D., & HEDLUND, K. 2002. Biodiversity, ecosystem functioning and soil decomposer food webs. *In*: LOREAU, M., NAEEM, S. and INCHAUSTI, P. eds. *Biodiversity and Ecosystem functioning: Synthesis and Perspectives.* Oxford: Oxford University Press. 169-180
MILLER, L.G., CONNELL, T.L., GUIDETTI, J.R., & OREMLAND, R.S. 1997. Bacterial oxidation of methyl bromide in fumigated agricultural soils. *Applied and Environmental Microbiology*, **63**, 4346-4354.
MILTNER, A., RICHNOW, H.H. & KOPINKE, F.D. 2005. Incorporation of carbon originating from CO2 into different compounds of soil microbial biomass and soil organic matter. *Isotopes in Environmental and Health Studies*, **41**, 135-140.

MUELLER, T., JENSEN, L.S., MAGID, J & NIELSEN N.E., 1997. Temporal variation of C and N turnover in soil after oilseed rape straw incorporation in the field: simulations with the soil-plant-atmosphere model DAISY. *Ecological Modelling*, **99**, 247-262.

MULDER, C., DE ZWART, D., VAN WIJNEN, H. J., SCHOUTEN, A.J., & BREURE, A.M. 2003. Observational and simulated evidence of ecological shifts within the soil nematode community of agroecosystems under conventional and organic farming. *Functional Ecology*, **17**, 516-525.

MULDER, C., DIJKSTRA, J.B. & SETÄLÄ, H. 2005.Nonparasitic nematoda provide evidence for a linear repsonse of functionally important soil biota to increasing livestock density. *Naturwissenschaften*, **92**, 314-318.

MULDOWNEY, J, CURRY, J.P. & O'KEEFFE, J. & SCHMIDT, O. 2003. Relationships between earthworm populations, grassland management and badger densities in County Kilkenny, Ireland. *Pedobiologia*, **47**, 913-919.

MURATA, T. & GOH, K.M. 1997. Effects of cropping systems on soil organic matter in a pair of conventional and biodynamic mixed cropping farms in Canterbury, New Zealand. *Biology and Fertility of Soils* , **25**, 372-381.

MURPHY, D.V., MACDONALD, A.J., STOCKDALE, E.A., GOULDING, K.W.T., FORTUNE, S., GAUNT, J.L., POULTON, P.R., WAKEFIELD, J.A., WEBSTER, C.P. & WILMER, W.S. 2000. Soluble organic nitrogen in agricultural soils. *Biology and Fertility of Soils*, **30**, 374-387.

MURPHY, D.V., STOCKDALE, E.A., BROOKES, P.C. & GOULDING, K.W.T. 2003. Impact of micro-organisms on chemical transformations in soil. *In* : L.K. ABBOTT & D.V. MURPHY ed. *Soil Biological Fertility – A key to sustainable land use in agriculture.* Dordrecht: Kluwer Academic, 139-152.

MURRAY, P.J. & HATCH, D.J. 1994. Sitona Weevils (Coleoptera, Curculionidae) As Agents for Rapid Transfer of Nitrogen from White Clover (Trifolium-Repens L) to Perennial Ryegrass (Lolium-Perenne L). *Annals of Applied Biology*, **125**, 29-33.

MYTTON, L.R., CRESSWELL, A. and COLBOURN, P. 1993. Improvement in soil structure associated with white clover. *Grass and Forage Science*, **48**, 84-90.

NASHÖLM, T., HUSS-DANELL, K. & HOGBERG, P. 2000. Uptake of organic nitrogen in the field by four agriculturally important plant species. *Ecology*, **81**, 1155-1161.

NAVNEET, B.K. & MEHROTRA, R.S. 1988. Phyllosphere microflora of wheat in relation to leaf leachates and resistance to helminthosporium blight. *Indian Journal of Phytopathology*, **41**, 398-405.

NEHER, D.A. 1999. Nematode communities in organically and conventionally managed agricultural soils. *Journal of Nematology*, **31**, 142-154.

NEHER, D.A. & OLSON, R.K. 1999. Nematode communities in soils of four farm cropping management systems. *Pedobiologia*, **43**, 430-438.

NEILSEN, K.E., LADEKARL, V.L. & NORNBERG, P. 1999. Dynamic soil processes on heathland due to changes in vegetation to oak and Sitka spruce. *Forest Ecology and Management*, **114**, 107-116.

NEILSON, R., ROBINSON, D., MARRIOTT, C.A., SCRIMGEOUR, C.M., HAMILTON, D., WISHART, J., BOAG, B. & HANDLEY, L.L. 2002. Above-ground grazing affects floristic composition and modifies soil trophic interactions. *Soil Biology & Biochemistry*, **34**, 1507-1512.

NI BHRIAIN, B., SKEFFINGTON, M.S., & GORMALLY, M. 2002. Conservation implications of land use practices on the plant and carabid beetle communities of two turloughs in Co. Galway, Ireland. Biological Conservation, **105**, 81-92.

NICOL, G.W., GLOVER, L. A. & PROSSER, J.I. 2003. The impact of grassland management on archaeal community structure in upland pasture rhizosphere soil. *Environmental Microbiology*, **5**, 152-162.
NISHIO, T. & FUJIMOTO, T. 1989. Mineralisation of soil organic nitrogen in upland fields as determined by a $^{15}N\text{-}NH_4^+$ isotope dilution technique and absorption of nitrogen by maize. *Soil Biology & Biochemistry*, **21**, 661-665.
NOBBE, F. & RICHTER, L. 1902. Über den Einfluss des Nitratstickstoffs und der Humussubstanzen auf den Impfungserfolg bei Leguminosen. *Landwirtschaftlichen Versuchs-Stationen*, **56**, 441-448.
NORMANDER, B. & PROSSER, J.I. 2000. Bacterial origin and community composition in the barley phytosphere as a function of habitat and presowing conditions. *Applied and Environmental Microbiology*, **66**, 4372-4377.
ODUM, E.P. 1969. The strategy of ecosystem development. *Science*, **164**, 262-270.
OEHL, F., SIEVERDING, E., INEICHEN, K., MÄDER, P., BOLLER, T., & WIEMKEN, A. 2003. Impact of land use intensity on the species diversity of arbuscular mycorrhizal fungi in agroecosystems of Central Europe. *Applied and Environmental Microbiology*, **69**, 2816-2824.
OEHL, F, SIVERDING, E., MÄDER, P., DUBOIS, D., INEICHEN, K., BOLLER, T. & WIEMKEN, A. 2004. Impact of long-term conventional and organic farming on the diversity of arbuscular mycorrhizal fungi. *Oecologia*, **138**, 574-583.
OGILVY, S.E. 1996. LINK-IFS. An integrated approach to crop husbandry. *Aspects of Applied Biology*, **47**, 335-342
OKANO, Y., HRISTOVA, K.R., LEUTENEGGER, C.M., JACKSON, L.E., DENISON, R.F., GEBREYESUS, B., LEBAUER, D. & SCOW, K.M. 2004. Application of real-time PCR to study effects of ammonium on population size of ammonia-oxidizing bacteria in soil. *Applied & Environmental Microbiology*, **70**, 1008-1016.
OPPERMAN, M.H., WOOD, M., & HARRIS, P.J. 1989. Changes in Microbial-Populations Following the Application of Cattle Slurry to Soil at 2 Temperatures. *Soil Biology & Biochemistry*, **21**, 263-268.
OSLER, G.H.R., VAN VLIET, P.C.J., GAUCI, C.S., & ABBOTT, L.K. 2000. Changes in free living soil nematode and microarthropod communities under a canola-wheat-lupin rotation in Western Australia. *Australian Journal of Soil Research*, **38**, 47-59.
OSTLE, N., WHITELEY, A.S., BAILEY, M.J., SLEEP, D., INESON, P & MANEFIELD, M. 2003. Active microbial RNA turnover in a grassland soil estimated using a (CO_2)-^{13}C spike. *Soil Biology & Biochemistry*, **35**, 877-885.
PANDA, S. & SAHU, S.K. 2004. Recovery of acetylcholine esterase activity of Drawida willsi (Oligochaeta) following application of three pesticides to soil. *Chemosphere*, **55**, 283-290.
PANDEY, S. & SINGH, D.K. 2004. Total bacterial and fungal population after chlorpyrifos and quinalphos treatments in groundnut (*Arachis hypogaea L.*) soil. *Chemosphere*, **55**, 197-205.
PANKHURST, C.E., DOUBE, B.M., GUPTA, V.V.S.R. & GRACE, P.R. 1994. *Soil biota. Management in sustainable farming systems*. Melbourne: CSIRO.
PARFITT, R.L., YEATES, G.W., ROSS, D.J., MACKAY, A.D., & BUDDING, P.J. 2005. Relationships between soil biota, nitrogen and phosphorus availability, and pasture growth under organic and conventional management. *Applied Soil Ecology*, **28**, 1-13.
PARKIN, T.B. 1987. Soil microsites as a source of denitrification variability. *Soil Science Society of America Journal*, **51**, 1194-1199.

PATERSON, E. & SIM, A. 1999. Rhizodeposition and C-partitioning of Lolium perenne in axenic culture affected by nitrogen supply and defoliation. *Plant and Soil*, **216**, 155-164.

PEACOCK, A.D., MULLEN, M.D., RINGELBERG, D.B., TYLER, D.D., HEDRICK, D.B., GALE, P.M., & WHITE, D.C. 2001. Soil microbial community responses to dairy manure or ammonium nitrate applications. *Soil Biology & Biochemistry*, **33**, 1011-1019.

PENNINGTON, P.I. & ELLIS, R.C. 1993. Autotrophic and heterotrophic nitrification in acidic forest and native grassland soils. *Soil Biology & Biochemistry*, **25**, 1399-1408.

PETERSEN, H. 2002. Effects of non-inverting deep tillage vs. conventional ploughing on collembolan populations in an organic wheat field. *European Journal of Soil Biology*, **38**, 177-180.

PETERSEN, S.O., HENRIKSEN, K., MORTENSEN, G.K., KROGH, P.H., BRANDT, K., SØRENSEN, K., MADSEN, T., PETERSEN, J. & GRØN, C. 2003. Recycling of sewage sludge and household compost to arable land: fate and effects of organic contaminants, and impact on soil fertility, *Soil & Tillage Research*, **72**, 139-152.

PETERSEN, S.O., STAMATIADIS, S. & CHRISTOFIDES, C. 2004. Short-term nitrous oxide emissions from pasture soil as influenced by urea level and soil nitrate. *Plant and Soil*, **267**, 117-127

PORAZINSKA, D.L., BARDGETT, R.D., BLAAUW, M.B., HUNT, H.W., PARSONS, A.N., SEASTEDT, T.R., & WALL, D.H. 2003. Relationships at the aboveground-belowground interface: Plants, soil biota, and soil processes. *Ecological Monographs*, **73**, 377-395.

POWLSON D.S. 1980. The effects of grinding on microbial and non-microbial organic matter. *Journal of Soil Science*, **31**, 77-85.

POWLSON, D. S. & JOHNSTON, A. E. 1994. Long term field experiments: their importance in understanding sustainable land use. *In*: D.J. GREENLAND & I. SZABOLCS, eds. *Soil Resilience and Sustainable Land Use*. Wallingford: CAB International, 367-394.

POWLSON, D.S., BROOKES, P.C., & CHRISTENSEN, B.T. 1987. Measurement of Soil Microbial Biomass Provides An Early Indication of Changes in Total Soil Organic-Matter Due to Straw Incorporation. *Soil Biology & Biochemistry*, **19**, 159-164.

RASMUSSEN, P.E. & ROHDE, C.R. 1988. Stubble Burning Effects on Winter-Wheat Yield and Nitrogen-Utilization Under Semiarid Conditions. *Agronomy Journal*, **80**, 940-942.

RECOUS, S., MARY, B, & FAURIE, G. 1990. Microbial assimilation of ammonium and nitrate in soil. *Soil Biology & Biochemistry*, **22**, 597-602.

RECOUS, S., AITA, C. & MARY, B. 1999. In situ transformations in bare soil after addition of straw. *Soil Biology & Biochemistry*, **31**, 119-133

REGANOLD, J.P., PALMER, A.S., LOCKHART, J.C., & MACGREGOR, A.N. 1993. Soil Quality and Financial Performance of Biodynamic and Conventional Farms in New-Zealand. *Science*, **260**, 344-349.

REYNOLDS, J. 1994. Earthworms of the world. *Global Biodiversity*, **4**, 11-16.

RIBERA, D., NARBONNE, J.F., ARNAUD, C., & SAINT-DENIS, M. 2001. Biochemical responses of the earthworm *Eisenia fetida andrei* exposed to contaminated artificial soil, effects of carbaryl. *Soil Biology & Biochemistry*, **33**, 1123-1130.

ROMBKE, J., JANSCH, S. & DIDDEN, W. 2005. The use of earthworms in ecological soil classification and assessment concepts. *Ecotoxicology and Environmental Safety*, **62**, 249-265

ROSSI, J-P, LAVELLE, P. & ALBRECHT, A. 1997. Relationship between spatial pattern of the endogeic earthworm *Polypheretima elongata* and soil heterogeneity. *Soil Biology and Biochemistry*, **29**, 485-488.

ROVIRA, A.D., SMETTEN, K.R.J. & LEE, K.E. 1989. Effect of rotation and conservation tillage on earthworms in a red-brown earth under wheat. *Australian Journal of Agricultural Research*, **38**, 829-834.

ROYCHOWDHURY, P., SUKUL, P., CHAKRAVARTY, A. & MUKHERJEE, D. 1999. Fluvalinate, a synthetic pyrethroid insecticide: Its effect on microbial dynamics and their activity. *Fresenius Environmental Bulletin*, **8**, 693-698.

RUMBERGER, A. & MARSCHNER, P. 2003. 2-Phenylethylisothiocyanate concentration and microbial community composition in the rhizosphere of canola. *Soil Biology & Biochemistry*, **35**, 445-452.

RYAN, M.H., CHILVERS, G.A. & DUMARESQ, D.C. 1994. Colonization of wheat by VA-mycorrhizal fungi was found to be higher on a farm managed in an organic manner than on a conventional neighbor. *Plant and Soil*, **160**, 33-40.

SARATHCHANDRA, S.U., GHANI, A., YEATES, G.W., BURCH, G. & COX, N.R. 2001. Effect of nitrogen and phosphate fertilisers on microbial and nematode diversity in pasture soils. *Soil Biology & Biochemistry*, **33**, 953-964.

SATTELMACHER, B., REINHARD, S. & POMIKALKO, A. 1991. Differences in Mycorrhizal Colonization of Rye (Secale-Cereale L) Grown in Conventional Or Organic (Biological-Dynamic) Farming System. *Journal of Agronomy and Crop Science-Zeitschrift fur Acker und Pflanzenbau*, **167**, 350-355.

SCHIMEL, D.S., 1986. Carbon and nitrogen turnover in adjacent grassland and cropland ecosystems. *Biogeochemistry*, **2**, 345-357.

SCHRÖDER, P., HÜBER, B., OLAZABAL, U., KAMMERER, A & MUNCH, J.C. 2002. Land use and sustainability: FAM Research Network on Agroecosystems. *Geoderma*, **105**, 155-166.

SCHRÖTER, D., BRUSSAARD, L., DE DEYN, G., POVEDA, K., BROWN, V.K., BERG, M.P., WARDLE, D.A., MOORE, J., & WALL, D.H. 2004. Trophic interactions in a changing world: modelling aboveground-belowground interactions. *Basic and Applied Ecology*, **5**, 515-528.

SCHUTTER, M.E. & DICK, R.P. 2002. Microbial community profiles and activities among aggregates of winter fallow and cover-cropped soil. *Soil Science Society of America Journal*, **66**, 142-153.

SCOTTISH EXECUTIVE 2003. Organic Action Plan for Scotland http://www.scotland.gov.uk/library5/agri/orap-00.asp Last modified: not given. Viewed 1 March 2006

SCOTTISH EXECUTIVE 2006. Soil condition and management plans as part of cross-compliance http://www.scotland.gov.uk/Resource/Doc/47121/0020243.pdf Last modified not given. Viewed 1 March 2006

SCULLION, J., EASON, W.R. & SCOTT, E.P. 1998. The effectivity of arbuscular mycorrhizal fungi from high input conventional and organic grassland and grass-arable rotations. *Plant and Soil*, **204**, 243-254.

SEASTEDT, T.R. 1984. The role of microarthropods in decomposition and mineralisation processes. *Annual Review of Entomology*, **29**, 25-46.

SEECH, A. G. & BEAUCHAMP, E. G. 1988. Denitrification in Soil Aggregates of Different Sizes. *Soil Science Society of America Journal*, **52**, 1616-1621.

SEGHERS, D., VERTHE, K., REHEUL, D., BULCKE, R., SICILIANO, S.D., VERSTRAETE, W. & TOP, E. M. 2003. Effect of long-term herbicide applications on the bacterial community structure and function in an agricultural soil. *FEMS Microbiology Ecology*, **46**, 139-146.

SEGHERS, D., WITTEBOLLE, L., TOP, E.M., VERSTRAETE, W. & SICILIANO, S.D. 2004. Impact of agricultural practices on the *Zea mays L*. endophytic community. *Applied and Environmental Microbiology*, **70**, 1475-1482.

SESSITSCH, A., WEILHARTER, A., GERZABEK, M.H., KIRCHMANN, H. & KANDELER, E. 2001. Microbial population structures in soil particle size fractions of a long-term fertilizer field experiment. *Applied and Environmental Microbiology*, **67**, 4215-4224.

SHANNON, D., SEN, A.M., & JOHNSON, D.B. 2002. A comparative study of the microbiology of soils managed under organic and conventional regimes. *Soil Use and Management*, **18**, 274-283.

SHIEL, R.S. 1986. Variations in amounts of carbon and nitrogen associated with particle-size fractions of soils from the Palace-Leas meadow hay plots. *Journal of Soil Science*, **37**, 249-257.

SIEPEL, H. 1995. Applications of microarthropod life history tactics in nature management and ecotoxicology. *Biology and Fertility of Soils*, **19**, 75-83.

SIMEK, M., BRUCEK, P., HYNST, J., UHLIROVA, E. & PETERSEN, S.O. 2006. Effects of excretal returns and soil compaction on nitrous oxide emissions from a cattle overwintering area. *Agriculture, Ecosystems & Environment*, **112**, 186-191.

SIX, J., CONANT, R.T., PAUL, E.A. & PAUSTIAN, K. 2002. Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. *Plant and Soil*, **241**, 155-176.

SMEDDING, F.W. & DE SNOO, G.R. 2003. A concept of food-web structure in organic arable farming systems. *Landscape and Urban Planning*, **65**, 219-236.

SMILEY, R.W. 1979. Wheat rhizosphere pseudomonads as antagonists of *Gaeumannomyces graminis*. *Soil Biology and Biochemistry*, **6**, 319-325.

SMIT, E., LEEFLANG, P., GOMMANS, S., VAN DEN BROEK, J., VAN MIL, S. & WERNARS, K. 2001. Diversity and seasonal fluctuations of the dominant members of the bacterial soil community in a wheat field as determined by cultivation and molecular methods. *Applied and Environmental Microbiology* , **67**, 2284-2291.

SMITH, B.J., KIRKEGAARD, J.A. & HOWE, G.N. 2004. Impacts of Brassica break-crops on soil biology and yield of following wheat crops. *Australian Journal of Agricultural Research*, **55**, 1-11.

SMITH, M.D., HARTNETT, D.C. & RICE, C.W. 2000. Effects of long-term fungicide applications on microbial properties in tallgrass prairie soil. *Soil Biology & Biochemistry*, **32**, 935-946.

SMITH, Z., MCCAIG, A.E., STEPHEN, J.R., EMBLEY, T.M. & PROSSER, J.I. 2001. Species diversity of uncultured and cultured populations of soil and marine ammonia oxidizing bacteria. *Microbial Ecology*, **42**, 228-237.

SPEDDING, T.A., HAMEL, C., MEHUYS, G.R., & MADRAMOOTOO, C.A. 2004. Soil microbial dynamics in maize-growing soil under different tillage and residue management systems. *Soil Biology & Biochemistry*, **36**, 499-512.

SPEHN, E.M., HECTOR, A., JOSHI, J., SCHERER-LORENZEN, M., SCHMID, B., BAZELEY-WHITE, E., BEIERKUHNLEIN, C., CALDEIRA, M.C., DIEMER, M., DIMITRAKOPOULOS, P.G, FINN, J. A., FREITAS, H., GILLER, P.S., GOOD, J., HARRIS, R., HOGBERG, P., HUSS-DANELL, K., JUMPPONEN, A., KORICHEVA, J., LEADLEY, P.W., LOREAU, M., MINNS, A., MULDER, C.P.H., O'DONOVAN, G., OTWAY, S.J., PALMBOURG, C., PEREIRA, J.S. PFISTERER,

A. B., PRONZ, A., READ, D.J., SCHULZE, E.D., SIAMANTZIOURAS, A.S.D. TERRY, A.C., TROUMBIS, A.Y., TROUMBIS, A.Y. & LAWTON, J.H. 2005. Ecosystem effects of biodiversity manipulations in European grasslands. *Ecological Monographs*, **75**, 37-63.

STAHL, P.D. & PARKIN, T.B. 1996. Relationship of soil ergosterol concentration and fungal biomass. *Soil Biology and Biochemistry*, **28**, 847-855.

STEVENS, D.K. & BRADBURY, R.B. 2006. Effects of the Arable Stewardship Pilot Scheme on breeding birds at field and farm-scales. *Agriculture Ecosystems &* Environment, **112**, 283-290

STOCKDALE, E.A. & BROOKES, P.C. 2006. Detection and quantification of the soil microbial biomass - impacts on the management of agricultural soils. *Journal of Agricultural Science*, in press.

STURZ A.V. & CHRISTIE, B.R. 1998. The potential benefits from cultivar specific red clover - potato crop rotations. *Annals of Applied Biology*, **133**, 365-373.

SUN, H.Y., DENG, S.P. & RAUN, W.R. 2004. Bacterial community structure and diversity in a century-old manure-treated agroecosystem. *Applied and Environmental Microbiology*, **70**, 5868-5874.

SVENDSEN, T.S., HANSEN, P.E., SOMMER, C., MARTINUSSEN, T., GRONVOLD, J., & HOLTER, P. 2005. Life history characteristics of Lumbricus terrestris and effects of the veterinary antiparasitic compounds ivermectin and fenbendazole. *Soil Biology & Biochemistry*, **37**, 927-936.

SWIFT, M.J., HEAL, O.W. & ANDERSON, J.M. 1979. *Decomposition in terrestrial ecosystems*. Oxford: Blackwell Scientific.

SWIFT, M.J., IZAC, A-M. N. & VAN NOORDWIJK, M. 2004. Biodiversity and ecosystem services in agricultural landscapes – are we asking the right questions? *Agriculture, Ecosystems & Environment*, **104**, 113-134.

TARRANT, K.A., FIELD, S.A., LANGTON, S.D. & HART, A.D.M. 1997. Effects on earthworm populations of reducing pesticide use in arable crop rotations. *Soil Biology & Biochemistry*, **29**, 657-661.

TEMPLE, S.R., FRIEDMAN, D.B., SOMASCO, O., FERRIS, H., SCOW, K. & KLONSKY, K. 1994. An interdisciplinary experiment station based participatory comparison of alternative crop management systems for California's Sacramento valley. *American Journal of Experimental Agriculture*, **9**, 62-71.

THURSTON, J. M., WILLIAMS, E. D. & JOHNSTON, A. E. 1976. Modern developments in an experiment on permanent grassland started in 1856 – effects of fertiliser and lime on botanical composition and crop and soil analyses. *Annales Agronomiques*, **27**, 1043-1082.

TIEDJE, J. M., CHO, J. C., MURRAY, A., TREVES, D. XIA, B. & ZHOU, J. 2001. Soil teeming with life: New frontiers for soil science. *In:* R.M. REES, B. C. BALL, C. D. CAMPBELL & C. A. WATSON. Eds. *Sustainable Management of Soil Organic Matter*. Wallingford: CAB International. 393-412.

TIQUIA, S.M., LLOYD, J., HERMS, D. A., HOITINK, H. A. J., & MICHEL, F. C. 2002. Effects of mulching and fertilization on soil nutrients, microbial activity and rhizosphere bacterial community structure determined by analysis of TRFLPs of PCR-amplified 16S rRNA genes. *Applied Soil Ecology* , **21**, 31-48.

TISDALL, J. M. & OADES, J. M. 1982. Organic matter and water stable aggregates in soils Journal of Soil Science, **33**, 141-163.

TREWAVAS, A. 2004. A critical assessment of organic farming-and-food assertions with particular respect to the UK and the potential environmental benefits of no-till agriculture. *Crop Protection*, **23**, 757-781.

TUCKER, G. M. 1992. Effects of agricultural practices on food use by invertebrate feeding birds in winter. *Journal of Applied Ecology*, **29**, 779-790.

UNGER, P.W. & CASSEL, D. K. 1991. Tillage Implement Disturbance Effects on Soil Properties Related to Soil and Water Conservation - A Literature-Review. *Soil & Tillage Research*, **19**, 363-382.

UNIVERSITY OF NEWCASTLE 2006. Nafferton Ecological Farming Group. www.ncl.ac.uk/tcoa/producers/ Last updated, 19 April 2006. Viewed, 23 April 2006.

VAN BERGEN, P. F., NOTT, C. J., BULL, I. D., POULTON, P. R. & EVERSHED, R. P. 1998. Organic geochemical studies of soils from the Rothamsted Classical Experiments - IV. Preliminary results from a study of the effect of soil pH on organic matter decay. *Organic Geochemistry*, **29**, 1779-1795.

VAN DER PUTTEN, W. H., DE RUITER, P. C., BEZEMER, T. M., HARVEY, J. A., WASSEN, M. & WOLTERS, V. 2004. Trophic interactions in a changing world. *Basic and Applied Ecology*, **5**, 487-494.

VAN DIEPENINGEN, A. D., DE VOS O. J., KORTHALS, G. W., & VAN BRUGGEN, A. H. C. 2006. Effects of organic versus conventional management on chemical and biological parameters in agricultural soils. *Applied Soil Ecology,* **31**, 120-135.

VAN ELSAS, J. D., DUARTE, G. F., KEIJZER-WOLTERS, A. and SMIT, E. 2000. Analysis of the dynamics of fungal communities in soil via fungal-specific PCR of soil DNA followed by denaturing gradient gel electrophoresis. *Journal of Microbiological Methods*, **43**, 133-151.

VAN NOORDWIJK, M, DE RUITER, P. C., ZWART, K. B., BLOEM, J, MOORE, J. C., VAN FAASSEN, H. G., & BURGERS, S. L. G. E. 1993. Synlocation of biological activity, roots, cracks and recent organic inputs in a sugar beet field. *Geoderma*, **56**, 265-276.

VAN ZWIETEN, L., RUST, J., KINGSTON, T., MERRINGTON, G., & MORRIS, S. 2004. Influence of copper fungicide residues on occurrence of earthworms in avocado orchard soils. *Science of the Total Environment*, **329**, 29-41.

VINTEN, A. J. A., LEWIS, D. R., MCGECHAN, M., DUNCAN, A., AITKEN, M., HILL, C. & CRAWFORD, C. 2004a. Predicting the effect of livestock inputs of E-coli on microbiological compliance of bathing waters. *Water Research*, **38**, 3215-3224.

VINTEN, A. J. A., DOUGLAS, J. T., LEWIS, D.R., AITKEN, M.N & FENLON D.R. 2004b. Relative risk of surface water pollution by *E. coli* derived from faeces of grazing animals compared to slurry application. *Soil Use and Management*, **20**, 13-22.

VITOUSEK, P. M. & HOOPER, D. U. 1993. Biological diversity and terrestrial ecosystem biogeochemistry. *In* : E. D. SCHULZE & A. D. MOONEY eds. *Biodiversity and ecosystem function.* Berlin: Springer Verlag, 3-14.

WALL, D. ed. 2004. *Sustaining Biodiversity and Ecosystem Services in Soils and Sediments.* SCOPE 64. Washington DC: Island Press. 15-43.

WANDER, M. M., HEDRICK, D. S., KAUFMAN, D., TRAINA, S. J., STINNER, B. R., KEHRMEYER, S. R., & WHITE, D. C. 1995. The Functional-Significance of the Microbial Biomass in Organic and Conventionally Managed Soils. *Plant and Soil*, **170**, 87-97.

WARDLE, D. A. 1992. A comparative assessment of factors which influence microbial biomass carbon and nitrogen levels in soil. *Biological Reviews of the Cambridge Philosophical Society*, **67**, 321-358.

WARDLE, D. A. 1995. Impacts of disturbance on detritus food webs in agro-ecosystems of contrasting tillage and weed management practices. *Advances in Ecological Research*, **26**, 105-185.
WARDLE, D. A. 2002. *Communities and ecosystems: linking the aboveground and below ground components.* Monographs in population biology 34. Princeton: Princeton University Press.
WARDLE, D. A. & GILLER, K. E. 1996. The quest for a contemporary ecological dimension to soil biology - Discussion. *Soil Biology & Biochemistry*, **28**, 1549-1554.
WARDLE, D. A., BONNER, K. I., BARKER, G. M., YEATES, G. W., NICHOLSON, K. S., BARDGETT, R. D., WATSON, R. N., & GHANI, A. 1999a. Plant removals in perennial grassland: Vegetation dynamics, decomposers, soil biodiversity, and ecosystem properties. *Ecological Monographs*, **69**, 535-568.
WARDLE, D. A., NICHOLSON, K. S., BONNER, K. I., & YEATES, G. W. 1999b. Effects of agricultural intensification on soil-associated arthropod population dynamics, community structure, diversity and temporal variability over a seven-year period. *Soil Biology & Biochemistry*, **31**, 1691-1706.
WATSON, C. A. and YOUNIE, D. 1995. Nitrogen balances in organically and conventionally managed beef production systems. *In*: G.E. POLLOT, ed. *Grassland into the 21st century: Challenges and Opportunities*. Reading: British Grassland Society, 197-199.
WATSON, C. A., RITZ, K., YOUNIE, D., & FRANKLIN, M. 1996. Nitrogen and soil biomass dynamics in ley/arable crop rotations. *Aspects of Applied Biology*, **47**, 43-50.
WATSON, C. A., YOUNIE, D., & ARMSTRONG, G. 1999. Designing crop rotations for organic farming: Importance of the ley-arable balance. *In*: J.E. OLESEN, R. ELTUN, M.J. GOODING, E.S. JENSEN, & U. KOPKE, eds. *Designing and Testing Crop Rotations for Organic Farmin, DARCOF Report No 1. Foulum:DARCOF,* 91-98.
WATSON, C. A., YOUNIE, D., STOCKDALE, E. A., & CORMACK, W. F. 2000. Yield and nutrient balances of stocked and stockless organic rotations. *Aspects of Applied Biology*, **62**, 261-268.
WATSON, C. A., ALROE, H. F., & KRISTENSEN, E. S. 2006. Research to support the development of organic food and farming. *In* : P. KRISTIANSEN, A. TAJI & J. REGANOLD, eds. *Organic Agriculture: A global perspective.* Victoria: CSIRO Publishing, 361-383.
WEINBERG, G. M. 1975. *An Introduction to General Systems Thinking*. New York: John Wiley & Sons.
Welsh Agri-Food Partnership, 1999. *The Welsh Organic Food Sector. A Strategic Action Plan.* Welsh Organic Industry Working Group.
WESTERGAARD, K., MULLER, A. K., CHRISTENSEN, S., BLOEM, J., & SORENSEN, S. J. 2001. Effects of tylosin as a disturbance on the soil microbial community. *Soil Biology & Biochemistry*, **33** , 2061-2071.
WHALEN, J. K. 2004. Spatial and temporal distribution of earthworm patches in corn field, hayfield and forest systems of southwestern Quebec, Canada. *Applied Soil Ecology*, **27**, 143-151.
WHALLEY, W. R., DUMITRU, E., & DEXTER, A. R. 1995. Biological effects of soil compaction. *Soil and Tillage Research*, **35**, 53-68.
WHITTAKER, R. H. 1972. Evolution and measurement of species diversity. *Taxonomy*, **21**, 213-251.

WIENS, J. A. 1992. Central concepts and issues of landscape ecology. In: Applying landscape ecology in biological conservation. Ed K.J. Gutzwiller. Springer-Verlag. New York pp 3-21.

WIENS, J. A. & MILNE, B. T. 1989. Scaling of 'landscapes' in landcsape ecology, or, landscape ecology from a beetle's perspective. *Landscape Ecology*, **3**, 87-96.

WILD, A. 1988. Plant nutrients in soil:phosphate. In: A WILD ed. *Russell's Soil Conditions and Plant Growth. 11th Edition.* Harlow: Longman. 695-742.

WILLIAMS, S. E., MARSH, H. & WINTER J. 2002. Spatial scale, species diversity and habitat structure: small mammals in Australian tropical rainforest. *Ecology*, **83**, 1317-1329.

WILSON, D. S. 1992. Complex interactions in metacommunities withi implications for biodiversity and higher levels of selection. *Ecology*, **73**, 1984-2000.

WILSON, R. J., ELLIS, S., BAKER, J. S., LINEHAM, M. E., WHITEHEAD, R. W. & THOMAS, C. D. 2002. Large-scale patterns of distribution and persistence at the range margins of a butterfly. *Ecology*, **83**, 3357-3368.

WILSON, S. C., ALCOCK, R. E., SEWART, A. P. & JONES, K. C. 1997. Persistence of organic contaminants in sewage sludge-amended soil: A field experiment. *Journal of Environmental Quality*, **26**, 1467-1477.

WINTER, J. P., VORONEY, R. P., & AINSWORTH, D. A. 1990. Soil Microarthropods in Long-Term No-Tillage and Conventional Tillage Corn Production. *Canadian Journal of Soil Science*, **70**, 641-653.

WOLTERS, V. 2000. Invertebrate control of soil organic matter stability. *Biology and Fertility of Soils*, **31**, 1-19.

WU, J. & DAVID, J. L. 2002. A spatially explicit hierarchical approach to modeling complex ecological systems: theory and applications. *Ecological Modelling*, **153**, 7-26

YEATES, G. W. 1982. Pungentus-Maorium (Nematoda, Dorylaimida) Population-Changes Under Pasture During 36 Months. *Pedobiologia*, **24**, 81-89.

YEATES, G. W. & KING, K. L. 1997. Soil nematodes as indicators of the effect of management on grasslands in the New England Tablelands (NSW): comparison of native and improved grasslands. *Pedobiologia*, **41**, 526-536.

YEATES, G. W., BARDGETT, R. D., COOK, R., HOBBS, P. J., BOWLING, P. J., & POTTER, J. F. 1997. Faunal and microbial diversity in three Welsh grassland soils under conventional and organic management regimes. *Journal of Applied Ecology*, **34**, 453-470.

YEATES, G. W., WARDLE, D. A. & WATSON, R. T. 1999. Responses of nematode populations, community structure, diversity and temporal variability to agricultural intensification overa seven year period. *Soil Biology and Biochemistry*, **31**, 1229-1232.

YOUNG, I. M. & CRAWFORD, J. W. 2001. Protozoan life in a fractal world. *Protist*, **152**, 123-126.

YOUNG, I. M. & CRAWFORD, J. W. 2004. Interactions and self-organisation in the soil-microbe complex. *Science*, **304**, 1634-1637.

YOUNG, I. M. and RITZ, K. 1998. Can there be a contemporary ecological dimension to soil biology without a habitat? *Soil Biology and Biochemistry*, **30**, 1229-1232.

YOUNG, I. M. & RITZ, K. 2000. Tillage, habitat space and function of soil microbes. *Soil and Tillage Research*, **53**, 201-213.

YOUNG, I. M., ROBERTS, A., GRIFFITHS, B. S., & CAUL, S. 1994. Growth of a ciliate protozoan in model ballotini systems of different particle sizes. *Soil Biology and Biochemistry*, **26**, 1173-1178.

YOUNIE, D. & ARMSTRONG, G. 1995. Botanical and invertebrate diversity in organic and intensively fertilised grassland. *In:* J. ISART & J. J. LLERENA eds. *Proceedings of the first ENOF Workshop- Biodiversity and land use.* Barcelona: Multitext. 34-44.
ZALLER, J. G. & KOPKE, U. 2004. Effects of traditional and biodynamic farmyard manure amendment on yields, soil chemical, biochemical and biological properties in a long-term field experiment. *Biology and Fertility of Soils*, **40**, 222-229.

Glossary Terms

Abiotic	To describe the physical and chemical aspects of an organism’s environment.
Accreditation	Official recognition by organic certification bodies.
Acidobacteria	A newly devised division of Bacteria. As implied by their name they are acidophilic. Despite having been studied very little, this division is an important contributor to ecosystems, particularly where soil is concerned. Kingdom : Bacteria Phylum:Acidobacteria Order:Acidobacteriales Family: Acidobacteriaceae
ACOS	Advisory Committee on Organic Standards.
Actinomycete	A rod-shaped or filamentous bacterium belonging to a large group that includes some that cause diseases and some that are the sources of antibiotics. Order: Actinomycetales.
Aerobic/ Anaerobic	Living or taking place only in the presence of oxygen in the absence of oxygen.
Aggregate	1. Constituting or amounting to a whole; total. 2. In botany: crowded or massed into a dense cluster. 3. Composed of a mixture of minerals separable by mechanical means.
Aggregated/Aggregation	A total or collection of different things added together, or the process of adding them together.
Agro-ecosystems	Agroecology is a scientific discipline that uses ecological theory to study, design, manage and evaluate agricultural systems that are productive but also resource conserving.
Allelochemicals	A chemical produced by one plant that is toxic to another.
Allelopathic	The release into the environment by one plant of a substance that inhibits the germination or growth of other potential competitor plants of the same or another species.

Amoebae	A single-celled organism found in water and in damp soil on land, and as a parasite of other organisms. Genus: Amoeba.
Anecic	Species of earthworm named due to their habit in the soil. The other two categories are endogeic and epigeic.
Anthropocentric/Anthropogenic	Relating to or resulting from the influence humans have on the natural world.
Apterygota	A subclass of small, agile insects, distinguished from other insects by their lack of wings in the present and in their evolutionary history.
Arbuscular/Arbuscular Mycorrhizal Fungi	A **mycorrhiza** (Greek for "fungus roots") is a distinct type of root symbiosis in which fungus colonize the roots of a host plant. An **Arbuscular mycorrhiza** is a type of mycorrhiza in which the fungus penetrates the roots of a vascular plant. They are characterized by the formation of unique structures such as vesicles and arbuscules.
Archaea/Archael	Members of one of two distinct groups of the most primitive living single-celled organisms, similar in size to bacteria but very different in molecular organization.
Arthropods	An invertebrate animal that has jointed limbs, a segmented body, and an exoskeleton made of chitin. Insects, arachnids, centipedes, and crustaceans are arthropods. Phylum: Arthropoda.
Assimilate	Absorb, in particular nutrients or carbon.
Asymptotic	The term asymptotic means approaching a value or curve arbitrarily closely
Autotrophic	Used to describe organisms, especially green plants, that are capable of making nutrients from inorganic materials.
Azoxystrobin	Asystemic, broad-spectrum fungicide with activity against the four major groups of plant pathogenic fungi including *Ascomcetes* (eg powdery mildews), *Basidiomycetes* (eg rusts), *Deutoromycetes* (eg rice blast) and *Oomycetes* (eg downy mildew).
Below-ground biodiversity	Range of organisms that live in the soil or other niches associated with the soil e.g. within plant roots.

Binary fission	The reproduction of a cell or a one-celled organism by division into two nearly equal parts.
Bioassay	A technique for determining the concentration or potency of a substance such as a drug by measuring its effect on a living organism.
Biocontrol	Control of pests by disrupting their ecological status, as through the use of organisms that are natural predators, parasites, or pathogens. Also called Biological Control.
Biodiversity	The range of organisms present in a given ecological community or system.
Biodynamic	Biodynamic farming is a system of organic farming developed by the Austrian scientist and philosopher Rudolf Steiner in the early part of the 20th Century. Biodynamic farming takes into consideration both the biological cycles and the metaphysical or spiritual aspects of the farm.
Biofilms	A collection of microorganisms surrounded by the slime they secrete, attached to either an inert or living surface.
Biofumigations	The use of plants containing biologically active compounds as rotation crops or green manures to suppress soil-borne pests and diseases in agricultural production systems.
Biomass	The mass of living organisms within a given environment, measured in terms of weight per unit of area.
Biota	The total complement of animals and plants in a particular area.
Bioturbation	The stirring or mixing of sediment or soil by organisms, especially by burrowing or boring.
Bioturbator	An organism that stirs or mixes sediment or soil, especially by burrowing or boring.
Butyrate	A salt or ester of butyric acid.
Butyrate Oxidisers	A substance which enables the combination of Butyrate with oxygen.
CAP	Common Agricultural Policy.

Carabid	A carnivorous beetle that lives in the soil and feeds on other insects. Family: Carabidae.
Carbamates	Any salt or ester of carbamic acid, used especially as a pesticide.
Carbon Sequestration	uptake and storage of carbon, especially by trees and plants that absorb carbon dioxide and release oxygen.
Certification	Inspection process for organic farming systems.
Chenopodiaceae	A flowering plant family, the Goosefoot family.
Chloropicin	A pesticide which controls: Cockroaches, Fungi, Fusarium, Mites, Nematides, Phytophthora, Pythium, Silverfish, Verticillium, Wireworms, Wood Infesting Insects and Wood Rot/Decay.
Chlorothalonil	A broad spectrum, multi-site fungicide which has particular strengths against *Septoria tritici*.
Chlorpyrifos	An organophosphate (OP) insecticides it is used to kill insect pests by disrupting their nervous system.
Ciliates	A simple microscopic organism with projecting threads that thrash to help it to move along. Phylum: Ciliophora.
Collembola	Common name Springtail. They are among the most abundant of all soil-dwelling arthropods. They live in a variety of habitats where they feed as scavengers on decaying vegetation and soil fungi..
Commensal	Of, relating to, or characterized by a symbiotic relationship in which one species is benefited while the other is unaffected.
Comminution	To divide something, especially property, into small parts.
Community	All the plants and animals that live in the same area and interact with one another.
Compaction	The pressing together of particles to make a denser mass, or the compressed state of the resulting mass.
Competitive exclusion	The concept that two or more species with identical requirements cannot coexist on the same limited

	resources because one will compete more successfully than the other.
Consumer	In an ecological community or food chain, an organism that feeds on other organisms, or on material derived from them.
Conventional	Agricultural production methods that have become the 'norm' over the last 50 + years that are specialised and reliant on high agrochemical inputs to achieve high yields.
Cortical	The tissue in plant stems and roots between the outer layer (epidermis) and the central core (stele).
Crenarchaeota	A major group of Archaea (a unicellular (prokaryotic) organism similar to bacteria in some ways and to Eukaryotes (Organisms with complex cells where the genetic material is in a membrane bound nuclei) in others). The group contains many extreme thermophilic organisms.
Cultivars	A variety of a cultivated plant that is developed by breeding and has a designated name.
Decomposers/Decomposition	An organism, especially a bacterium or fungus, that causes organic matter to rot or decay.
Defoliation	To strip trees and plants of their leaves, for example by using chemicals or through pollution or attack by pests, or to lose leaves in any of these ways.
Denitrifier/Nitrifier	To convert nitrates into nitrites, ammonia, and nitrogen.
Desiccation	To remove the moisture from something or become free of moisture.
Detritivores	An organism that feeds on decaying animal or plant material.
Detritus	Organic debris formed by the decomposition of plants and animals.
Dimethoate	A crystalline compound used as an insecticide.
Diplopods	A millipede that has two pairs of legs on each body segment. Class: Diplopoda.
EA	Environment Agency.

Ecophysiology	The study of the interrelationship between an organism's physical functioning and its environment.
Ecosystem Engineers	Organisms that create, modify and maintain habitats.
Efflux	Something that flows out of something else.
Empirical evidence	Evidence that relys on or is derived from observation or experiment. Verifiable or provable by means of observation or experiment.
Enchytraeids	Common name Pot Worm. They are very small white worms that can reach densities of 250,000 individuals per square meter. The highest populations are found in acid soils. They feed on bacteria and fungi.
Encystment	To enclose or be enclosed in a cyst.
Endogeic	See Anecic.
Endophytic Bacteria	A unicellular prokaryotic organism that is growing within another plant.
Enmeshment	To entangle somebody or something in something from which it is difficult to be extricated or separated
Ephemeral	a plant or insect that lives for only a short period of time.
Epidermal	The outer layer of cells of invertebrates that secretes the protective waxy cuticle.
Epigeic	See Anecic.
Epoxiconazole	A broad-spectrum fungicide with preventative and curative action used principally on wheat and winter barley.
Epoxiconazole/fenpropimorph/ kresoxim-methyl	A broad-spectrum fungicide with preventative and curative action used principally on wheat and barley.
Equilibria/Equilibrium	A state or situation in which opposing forces or factors balance each other out and stability is attained.
Ergosterol	A crystalline steroid alcohol that is found mainly in yeast and moulds and is converted to vitamin D2 by ultraviolet light.

Eukaryotic	Any organism with one or more cells that have visible nuclei and organelles.
Evapotranspiration	The return of moisture to the air through both evaporation from the soil and transpiration by plants.
Excreta	Any waste matter discharged from the body, for example faeces, or urine.
Extensive	Agricultural term relating to a farming practice in which a large area of land is cultivated using little labour and expense, resulting in a relatively small crop/return.
Exudation	The release of a substance through pores or a surface cut, for example the release of sweat from the body or resin from a tree.
Fauna	1. All the animal life in a particular region. 2. A living organism characterized by voluntary movement.
Fermentation	Typically refers to the conversion of sugar to alcohol using yeast. In its strictest sense, fermentation (formerly called zymosis) is the anaerobic metabolic breakdown of a nutrient molecule, such as glucose, without net oxidation. It is also used much more broadly to refer to the bulk growth of microorganisms on a growth medium.
Flagellates	A usually non-photosynthetic free-living protozoan with whip-like appendages.
Flora	1. All the plant life in a particular region. 2. A living organism lacking the power of locomotion.
Fodder	A coarse food for livestock composed of entire plants or the leaves and stalks of a cereal crop.
Forbs	A broad-leaved, herbaceous, non-woody flowering plant that is not a grass.
Formulants	Any added material in a pesticide formulation other than the biologically *active ingredient(s).*
Functional Group	A group of organisms that perform a similar activity or role.
Herbicide	A chemical substance used to destroy or inhibit the growth of plants, especially weeds.

Heterotrophic	An organism that cannot synthesize its own food it obtains its energy from carbohydrates and other organic material. All animals and most bacteria and fungi are heterotrophic.
Horizon	A layer of soil in the soil profile.
Humus	Partially decomposed organic matter; the organic component of soil.
Hybrid	Something of mixed origin or composition.
Hyperparasitism	A condition in which a secondary parasite develops within a previously existing parasite.
Hyphal	Threadlike filaments forming the mycelium, the vegetative part, of a fungus.
IFOAM	International Federation of Organic Agricultural Movements.
Intensive	agricultural term relating to a form of agriculture in which scientific and technological methods, for example the use of chemicals that boost growth or crop yields, are used to increase productivity.
Isopods	Common name for crustaceans belonging to the order Isopoda. Any of various small terrestrial or aquatic crustaceans with seven pairs of legs adapted for crawling.
Legume	a plant that has pods as fruits and roots that bear nodules containing nitrogen-fixing bacteria.
Macro-	Prefix (Greek) meaning large.
Macroporosity	Porosity means the property of being porous; being able to absorb fluids.
Meso-	Prefix (Greek) meaning 'in the middle'.
Meta-analysis	The process or technique of synthesizing research results by using various statistical methods to retrieve, select, and combine results from previous separate but related studies.
Metabolic	Of, relating to, or resulting from metabolism.

Metabolism	The chemical processes occurring within a living cell or organism that are necessary for the maintenance of life. In metabolism some substances are broken down to yield energy while other substances are synthesized.
Methanogens	Anaerobic unicellular organisms originally thought to be bacteria but now recognized as belonging to the archaea. They produce Methane as a metabolic by-product. Methanogens play an important role in the degradation of complex organic compounds.
Methanotrophs	Bacteria that are able to grow using methane as their only source of carbon and energy. Occur mostly in soils, and are especially common near environments where methane is produced. They are of special interest to researchers studying global warming.
Methylotrophs	A diverse group of microorganisms that can utilize reduced one-carbon compounds, such as methanol, as the carbon sources for their growth.
Micro-	Prefix (Greek) meaning small; minute.
Microbiota	The combined micro-flora and micro-fauna of an organism; or, the micro-flora or micro-fauna considered separately.
Microbial Inoculant	When a pathogenic micro-organism is used as a safe and alternative method in controlling insect pests, a form of biological control.
Microcosms	A small, representative system having analogies to a larger system in constitution, configuration, or development.
Micro-habitat	A very small, specialized habitat, such as a clump of grass or a space between rocks.
Microorganisms	An organism of microscopic or submicroscopic size, especially a bacterium or protozoan.
Mineralisation	The process where a substance is converted from an organic substance to an inorganic substance.
Monogastrics	An organism which has only one stomach, and is the alternate gastric complex to a four-chambered stomach known as a ruminant. Examples of monogastric animals include rabbits, humans, and pigs.

Mucilage	A thick gluey substance produced by most plants and some microorganisms.
Multi-nucleated cytoplasm	Having two or more nuclei. The protoplasm outside the nucleus of a cell.
Multi-trophic	When more than two trophic level are involved.
Municipal Waste	Refuse/rubbish of, in, or belonging to a city.
Muriate of potash	Potash products are sold as Muriate of Potash. See Potash.
Mutualistic	An association between organisms of two different species in which each member benefits.
Mycelial morphology	The study of the structure and form of the vegetative part of a fungus, consisting of a mass of branching, threadlike hyphae (long, thread like filaments).
Mycorrhizal	The symbiotic association of the mycelium of a fungus with the roots of certain plants, such as conifers, beeches, or orchids.
Nematicides	Chemical method of nematode management.
Nematode	Any of several worms of the phylum Nematoda, having unsegmented, cylindrical bodies, often narrowing at each end, and including parasitic forms such as the hookworm and pinworm.
Nodulation	The formation or presence of nodules.
Nodules	In Botany: A small knotlike outgrowth, as those found on the roots of many leguminous plants. In Mineralogy: A small rounded lump of a mineral or mixture of minerals, usually harder than the surrounding rock or sediment.
Nucleic Acid	Any of a group of complex compounds found in all living cells and viruses, composed of purines, pyrimidines, carbohydrates, and phosphoric acid. Nucleic acids in the form of DNA and RNA control cellular function and heredity.
Oligochaeta	Common name: "few-bristled" worm. Well-segmented Annelids (worms). Includes earthworms, tubificids, pot worms, ice worms, blackworms (Lumbriculidae) and many interstitial marine worms.

Omnivorous	Feeding on both plants and animals.
Organic Wastes	Sewage sludge, animal manure and abattoir waste.
Organophosphates	Any of several organic compounds containing phosphorus, some of which are used as fertilizers and pesticides. Or an insecticide that interferes with an insect's nervous system.
Oribatid	A group of mites that live in the soil.
Pathogens	A disease-causing agent. Microorganisms, viruses, and toxins are examples of pathogens.
Permeability	In soil science, it is a measure of the infiltration rate of precipitation into the soil.
Perturbation	A secondary influence on a system that causes it to deviate slightly. In terms of soils it relates to changes in the nature of alluvial (sediment) deposits over time.
Pesticide	A chemical used to kill pests, especially insects.
Phenology	The scientific study of periodic biological phenomena, such as the flowering of plants, in relation to climatic conditions.
Phylum	A primary division of a kingdom, as of the animal kingdom, the ranking next above a class in size.
Phytophthora	Destructive parasitic fungi causing brown rot in plants.
Poaching	To become muddy or broken up from being trampled.
Pore	A space in rock, soil, or unconsolidated sediment that is not occupied by mineral matter and that allows the passage or absorption of fluids.
Porosity	The property of being porous; being able to absorb fluids.
Potash	Any of several compounds containing potassium, especially soluble compounds such as potassium oxide, potassium chloride, and various potassium sulfates, used chiefly in fertilizers. Potash products are sold as Muriate of Potash.

Precipitation	Meteorology: 1. Any form of water, such as rain, snow, sleet, or hail, that falls to the earth's surface. Or 2. The quantity of such water falling in a specific area within a specific period. Chemistry*:* The process of separating a substance from a solution as a solid.
Prokaryotic	An unicellular organism having cells lacking membrane-bound nuclei; bacteria are the prime example.
Propagule	Any of various usually vegetative portions of a plant, such as a bud or other offshoot, that aid in dispersal of the species and from which a new individual may develop.
Prostigmatid	A group of mites that live in the soil.
Proteobacteria	A major group of bacteria. They include a wide variety of pathogens.
Protozoa	Any of a large group of single-celled, usually microscopic, eukaryotic organisms, such as amoebas, ciliates, flagellates, and sporozoans.
Pseudomonads	Any of various gram-negative, rod-shaped, mostly aerobic flagellated bacteria of the phylum Pseudomonad, commonly found in soil, water, and decaying matter and including some plant and animal pathogens.
Pythium	Common name: cottony blight or grease spot, a fungal disease of turfgrasses.
Resilience	The ability to recover quickly from setbacks.
Resistance	The ability to remain unaltered by the damaging effect of something, for example an organism's ability not to succumb to disease or infection.
Rhizobia	Any of various nitrogen-fixing bacteria of the genus *Rhizobium* that form nodules on the roots of leguminous plants, such as clover and beans. Can fix atmospheric oxygen.
Rhizobium	See Rhizobia.
Rhizoplane	Surface part of a plant's root. The part of a plant's root that lies at the surface of the soil, where many microorganisms adhere to it.

Rhizosphere	The soil zone that surrounds and is influenced by the roots of plants.
Ribotype	A subtype of a bacterial strain more detailed than the species or serotype level, determination of a ribotype is based on analysis of patterns formed by DNA fragments.
Rivet hypothesis	The ‘rivet’ or 'rivet popper' hypothesis suggests ecosystems are like aeroplane wings where flight (ecosystem functioning) may or may not be compromised depending upon which rivets (species) are lost.
RNA	One of a group of molecules similar in structure to a single strand of DNA. The function of RNA is to carry the information from DNA in the cell's nucleus into the body of the cell, to use the genetic code to assemble proteins, and to comprise part of the ribosomes that serve as the platform on which protein synthesis takes place.
Salinisation	The accumulation of soluble mineral salts near the surface of soil.
Saprophytic	Feeding or growing upon decaying animal or vegetable matter.
Sclerotium	Compact usually dark-colored mass of hardened mycelium constituting a vegetative food-storage body in various true fungi; detaches when mature and can give rise to new growth. Plural: Sclerotia.
Secondary metabolites	Chemical compounds in organisms that are not directly involved in the normal growth, development or reproduction of organisms.
Senescent	Growing old or ageing.
Surfactants	A surface active agent or a substance capable of reducing the surface tension of a liquid in which it is dissolved.
Sward	Land covered with grassy turf. A lawn or a meadow.
Symbiosis	An interaction between two organisms living together in more or less intimate association or even the merging of two dissimilar organisms. The term host is usually used

	for the larger of the two members of a symbiosis. The smaller member is called the symbiont.
Testate amoebae	Protozoa: Rhizopoda. Unicellular shelled animals.
Tillage	The cultivation of soil for growing crops.
Trichoderma	Fungi which are present in nearly all soils and other diverse habitats. They are favoured by the presence of high levels of plant roots, which they colonize readily.
Trophic Level	The position that an organism occupies in a food chain - what it eats, and what eats it.
UKAS	United Kingdom Accreditation Service.
Xenobiotics	A chemical which is found in an organism but which is not normally produced or expected to be present in it e.g. antibiotics.