



**Soil biology in agriculture**  
**Workshop Proceedings, Tamworth, 11-12 August 2004 - Readers' Note**

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## **Impact of management practices on activity of soil biota and productivity constraints in Vertosols of the northern grains region**

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### **Introduction**

The Vertosols represent the predominant soil type supporting the broadacre grain and cotton industries in the northern grains region. They are characterised by relatively high plant available water holding capacity (PAWC: >20mm plant available water/10cm profile depth—Webb et al 1997) and, in their virgin state, moderate to high levels of chemical fertility. However sustained cropping, initially using conventional tillage but in recent years employing reduced or zero tillage practices, has resulted in a decline in soil chemical fertility so that chemical fertiliser applications (especially nitrogen and phosphorus) are necessary to sustain productivity (Dalal, Probert, 1997).

Farming systems vary significantly across the region. In the higher/more reliable rainfall areas of the east, cropping is the dominant land use. Further west, mixed cropping and grazing enterprises are more common, with cropland rotated with grazed pasture phases. The dominant crops in the region are sorghum and cotton (summer) and wheat and chickpeas (winter), with barley, mungbeans and maize also important in some areas. This variety of crop options leads to many varied crop rotations across the region.

The predominant factor limiting crop productivity is water, with extremely variable annual rainfall contributing to a cropping sequence that becomes increasingly opportunistic in more western and northern areas (Freebairn et al 1997). Growing seasons are often characterised by long periods without rain, with crops solely reliant on subsoil moisture reserves. Fallowing is used to replenish soil water reserves between crops, with 12-18 month bare fallows common in the region. Long (12-15 month) fallows are also commonly used during the transition from summer to winter crops in the rotation (eg sorghum harvested in autumn, with wheat sown in winter the following year). The widespread adoption of conservation tillage and direct-drill methods have increased both the crop frequency and system productivity due to improved efficiency of water capture and use by crops.

Research has been conducted in the northern region on specific functional components of the soil biota, particularly on soil-borne diseases, plant-parasitic nematodes and on the importance of mycorrhizae in the cropping system (Wildermuth et al 1997). Less work has been done on various components of the detritus food web that can potentially moderate the impact of these functional components on crops in the farming system. This paper reports preliminary findings from a current research project that aims to quantify the impact of various management practices on soil biota in these land use systems, and to relate these impacts to crop productivity and system performance.

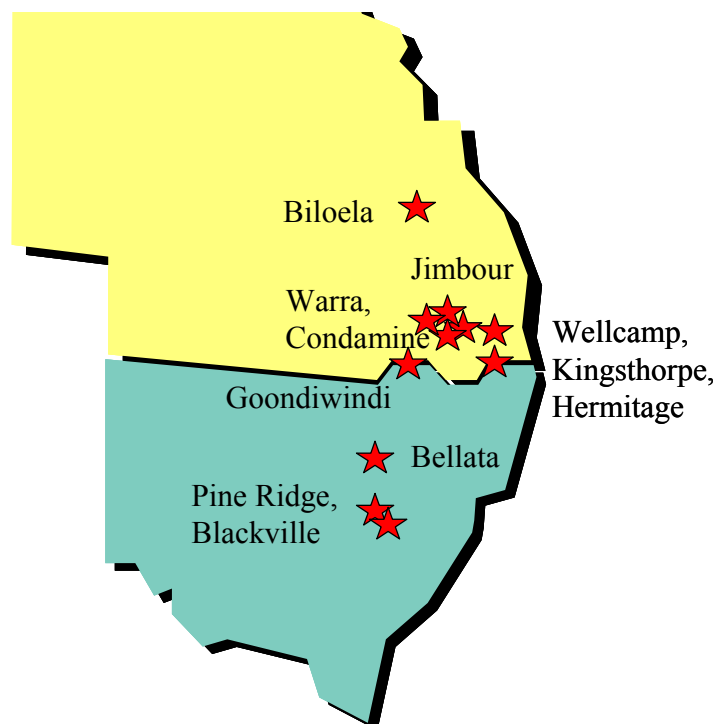
## Materials and methods

### **Experimental sites**

Experimental sites have been chosen for a variety of measurements of both soil biota and crop performance across the region, from Biloela in the north to Pine Ridge in the southern Liverpool Plains, and west to Goondiwindi (Figure 1). Given the predominance of chickpea, wheat and sorghum in the cropping system, sites that were growing these crops as part of the crop rotation were preferentially selected, while these species were also used in subsequent field and glasshouse studies.

The sites were chosen to represent long-term contrasts in management (ie paired sites in adjoining fields, or in controlled experiments), to represent contrasting farming systems (eg ley pasture phases versus continuous cropping) or as representative sites on which to benchmark system performance in a district or region. In addition to sampling existing sites and experiments, the project has also established a more controlled experiment to examine the impact of fallow length and crop rotation on soil biota and any biotic constraints to crop performance.

**Figure 1. Locations at which soils have been collected or research sites established in the northern soil biology project.**



### **Assessment of microbial diversity and chemical fertility in response to management and organic amendments**

Stratified soil samples were collected from the 0-5 cm, 5-15 cm and, in some cases, the 15-30 cm layers of the soil profile, with a number of representative samples collected from each plot (in the case of replicated field experiments) or at multiple sites (generally five) within a commercial field. While important differences in populations of soil biota can occur at greater depths in the soil profile (eg lesion nematodes, Peck et al 1993), it was assumed that the most significant impacts of management on soil biota would be detected in the upper layers.

Samples were homogenised and assessed either 'as is', or after amendment with a range of contrasting organic materials (feedlot manure, grass or legume plant material). In the case of the organic amendments, samples were moistened periodically for a period of six months to allow equilibration of the microbial communities before subsequent assessment. All samples were then separated into sub-samples for microbial or chemical analysis, or used in glasshouse bioassays.

Microbial analyses included determination of microbial biomass carbon, microbial activity using FDA, total DNA, nematode community analysis and mycorrhizal spore levels. A smaller subset of samples was assessed for root pathogens by the SARDI Root Disease Testing Service and for the composition of the microbial community using PL-FAME. Soil chemical properties included total and labile fractions of soil organic carbon, in addition to more traditional measurements of chemical fertility.

### ***Crop responses to soil pasteurisation and biocide applications***

Initial glasshouse bioassays were conducted using soil samples with or without pre-treatment of steam pasteurisation (60°C for 60 minutes), to assess potential biotic constraints in different regions and cropping systems. After pre-treatment, soils were potted with levels of nutrients designed to provide luxury levels of chemical fertility. Pots were placed on benches equipped with self-regulated watering systems and bioassay plants (sorghum cv MR Buster, wheat cv Hartog and chickpea cv Jimbour) were grown for eight weeks. Dry weights of tops and roots, root health, root length and mycorrhizal colonisation levels have been recorded.

In order to more closely examine the key components of soil biota associated with pasteurisation responses and suppression of lesion nematodes, a second series of experiments with treatments comprising a range of specific biocides (fungicides, nematicides and antibiotics, in addition to more general biocides like irradiation and pasteurisation) are being conducted.

To confirm the relevance of the glasshouse bioassays as predictors of biotic constraints in field-grown crops, field fumigation assays using a commercial fumigation rig to apply methyl bromide at rates of 1000 kg/ha are being undertaken on sites in the Jimbour-McAllister and Pine Ridge districts. Soil from these sites is simultaneously being used in additional glasshouse bioassays using both disturbed and undisturbed soil samples to assess response to pasteurisation, irradiation or more specific biocides.

### ***Assays for suppression of lesion nematode***

Soil from a subset of sites with contrasting managements (eg continuous crop v ley pastures) has been used to assess the impact of management on suppression of lesion nematodes in glasshouse experiments with wheat. Again, a range of biocides is also being used to help identify key components of soil biota responsible for reductions in pathogen reproduction and/or pathogenicity.

### ***Benchmarking the performance of field crops***

A number of crops, primarily in commercial fields, have been benchmarked against soil fertility and seasonal conditions using APSIM. Soils have been extensively characterised for PAWC, soil fertility at planting and seasonal rainfall and water use, with differences between simulated (potential) and actual yields being assessed as a potential measure of biotic constraints. Soil from these sites is also used in other assessments of biotic constraints (eg glasshouse bioassays).

## Results and discussion

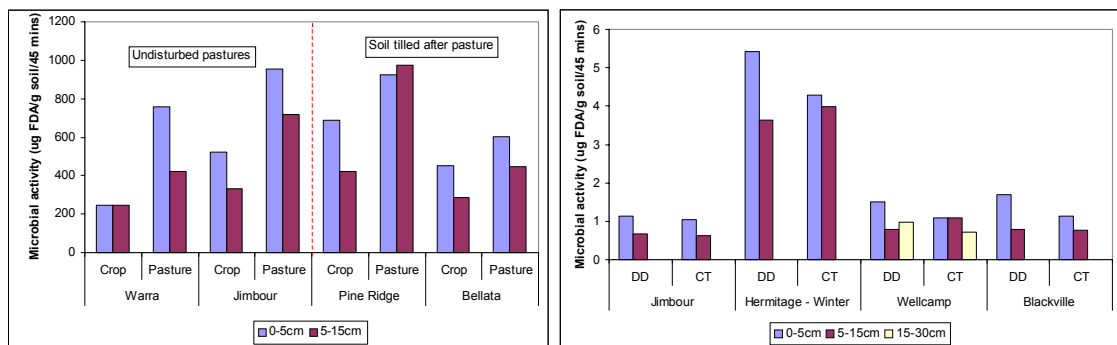
### **Impact of management and soil amendments on soil biota**

There have been some clear and consistent effects of management on soil biota (both amount and diversity) across the region.

- There were clear indications of a reduction in overall microbial activity resulting from continuous cropping compared with periodic pasture leys (Figure 2a), and with increasing length of bare fallow. The differences between crop and pasture ley soils were less pronounced once the pastures had been removed using conventional tillage.

**Figure 2a. Effect of crops and pasture leys on soil microbial activity at planting.**

**Figure 2b. Effect of tillage on soil microbial activity at planting**



- Changing from conventional tillage to direct-drill (Figure 2b) had only relatively small effects on microbial activity, with these primarily restricted to the top five cm of the soil profile, a layer that is dry for long periods in most years. These changes were minor compared with the increase arising from stubble retention (data not shown).
- Nematode faunal profiles were used to broadly characterise the biological status of these soils. The soil food web components identified in this analysis did not differ greatly in response to management. There was evidence of increasing food web structure being maintained under minimum/zero tillage, but stresses due to erratic rainfall and carbon inputs combined to minimise these differences. The predominance of fungal-feeding nematodes and the lack of bacterial-feeding enrichment opportunist species suggested slow residue decomposition and nutrient cycling dominated by fungi. Confirmation of this is being sought by other methods.
- PL-FAME analyses also suggested little change in dominance of the various functional groups within the microbial biomass in response to management, with the main impact on the overall level of activity.
- Adding organic amendments to soils increased soil microbial activity, but had only small effects on microbial diversity and crop performance.

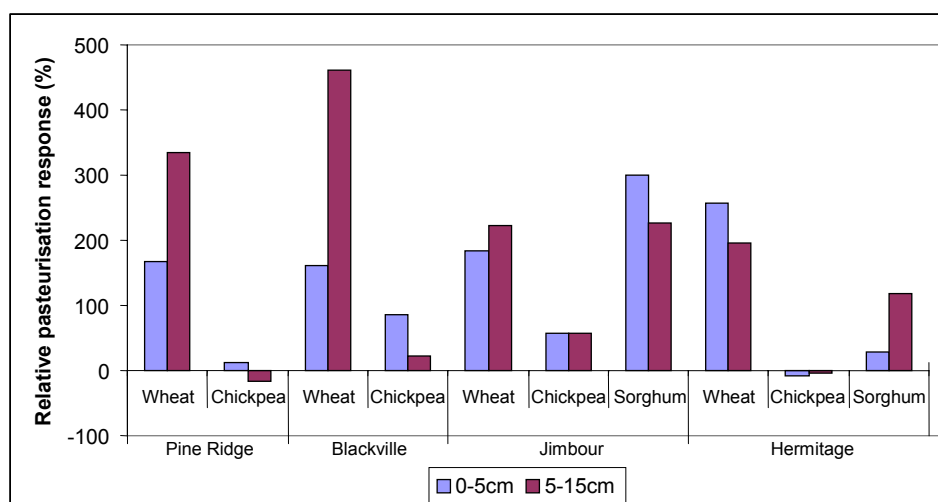
### **Plant response to elimination of biotic constraints**

Glasshouse bioassays comparing growth in pasteurised and unamended soil also showed consistent treatment responses, some of which are illustrated in Figure 3.

- Strong positive growth responses were recorded in response to soil pasteurisation in wheat and sorghum, but effects were much smaller in chickpea.
- Pasteurisation responses tended to be less when crops were grown in soil from the 0-5cm layer than from deeper layers of the profile.

- Pasteurisation responses were generally greater in continuously cropped soils compared with soils in pasture leys (data not shown).

Collectively, data suggest there may be widespread biotic constraints in the northern Vertosols that predominantly affect grain crops rather than grain legumes. There is also a fairly consistent trend for these constraints to be minimised in situations with greater levels of microbial activity.



**Figure 3.** Effects of soil pasteurisation on the relative growth response of sorghum, wheat and chickpea crops in soil from sites at Jimbour. Relative pasteurisation response was calculated as  $(\text{growth in pasteurised soil} / \text{growth in unpasteurised soil}) * 100$ .

Field fumigation studies are underway in an effort to confirm these results, and to determine the importance of these constraints under differing climatic conditions. However, benchmarking of a number of commercial and experimental crops using APSIM has so far suggested that a number of grain crops, especially wheat, have yielded poorly relative to the availability of water and nitrogen for the respective growing seasons (Figure 4). These results are consistent with the suggestion that there may be some significant biotic factors constraining yields of grain crops in this region. The ability of APSIM to predict yield in field-fumigated sites (analogous to potential yield in the absence of biotic constraints) is currently being assessed.

#### **Impacts of management on suppression of lesion nematode**

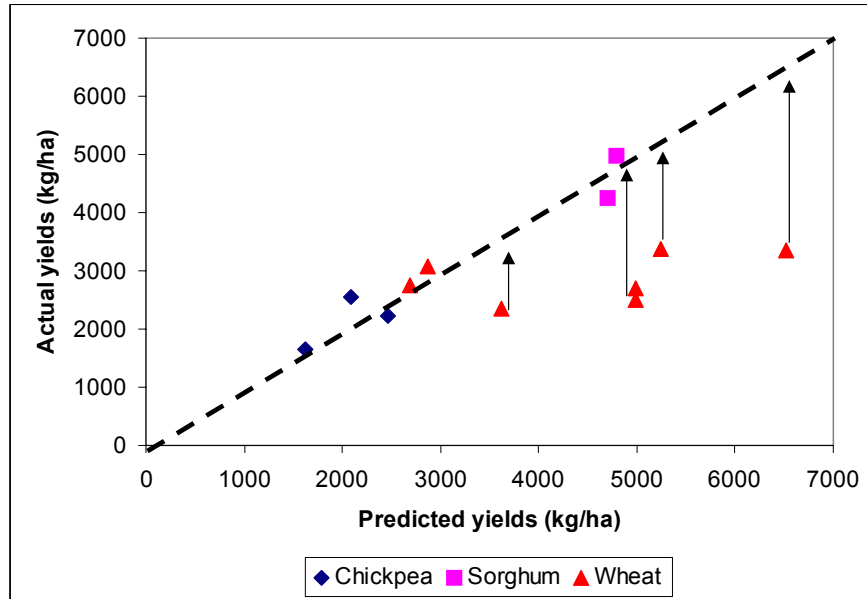
There has been clear evidence of suppression of lesion nematode multiplication in wheat grown in soils under a long-term pasture ley, relative to those with a long-term cropping history (Table 1).

**Table 1.** Effect of site (crop v pasture) and heat (heat v no heat) on multiplication of *Pratylenchus thornei* on wheat. All pots were inoculated so that the initial inoculum density was about 2.8 *P. thornei*/g soil.

Site	Final no. of <i>Pratylenchus</i> /g soil	
	Heated	Non-heated
Crop	23.0	5.4
Pasture	9.5	1.9
<b>LSD (P= 0.05)</b>	3.01	

This suppression seemed to consist of both a biotic component that could be eliminated by heating, and an abiotic component that was still evident after heating. The exact nature of this suppression is being investigated in further glasshouse studies, and a similar technique will be used to investigate potential for suppression of other pathogens such as crown rot in wheat.

**Figure 4. Actual commercial yields versus simulated yields predicted using APSIM for a number of commercial sites on the Darling Downs in 2002/03 and 2003 seasons. The arrows represent potential yield lost to unknown factors that could include soil biota.**



### Management implications, and where to from here

Data collected in the early stages of this project have suggested that detrimental soil biota may be having significant negative impacts on productivity of broadacre grain crops in the Vertosols of the northern region. There are some key components of the cropping system that are impacting on microbial activity and diversity (fallowing, crop rotation, use of pasture leys and tillage), and others are being investigated (eg use of inorganic fertilisers). However, the relationship between these management impacts and the incidence of particular soil-borne diseases is unclear at this time.

There is some concern at the obvious stratification of the majority of microbial activity in shallow surface layers that are dry for long periods of the growing season in this environment, especially under direct-drill systems. Similarly, management strategies that have been devised to conserve soil water (long fallows, skip row cropping) may be having negative impacts on soil biota due to a reduced frequency and non-uniform spatial distribution of organic matter inputs. The low frequency of pasture leys in the more reliable cropping areas is also contributing to the low levels of microbial activity. The implications of some of these management strategies are being investigated in more detail in the continuing experimental program.

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## Soil organic matter, biological activity, and productivity: myths and realities

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### What is soil organic matter?

Any discussion of soil organic matter (SOM) must first define just what is being talked about. This is because different people tend to have slightly different ideas of what SOM is and what it includes. Table 1 shows an example of these differences in the concept of SOM among 23 NSW Department of Primary Industries (NSWDPI) district agronomists in response to a recent survey. For research, extension and discussion of the roles, benefits, and properties of SOM to be coherent, there needs to be a clearer consensus on what is meant by SOM. Baldock and Nelson (2000) derived the following definition from several eminent sources: 'Soil organic matter is the sum of all natural and thermally altered biologically derived organic material found in the soil or on the soil surface irrespective of its source, whether it is living or dead, or stage of decomposition, but excluding the above-ground portion of living plants.' More simply put, soil organic matter is everything in the soil of biological origin, whether living or non-living.

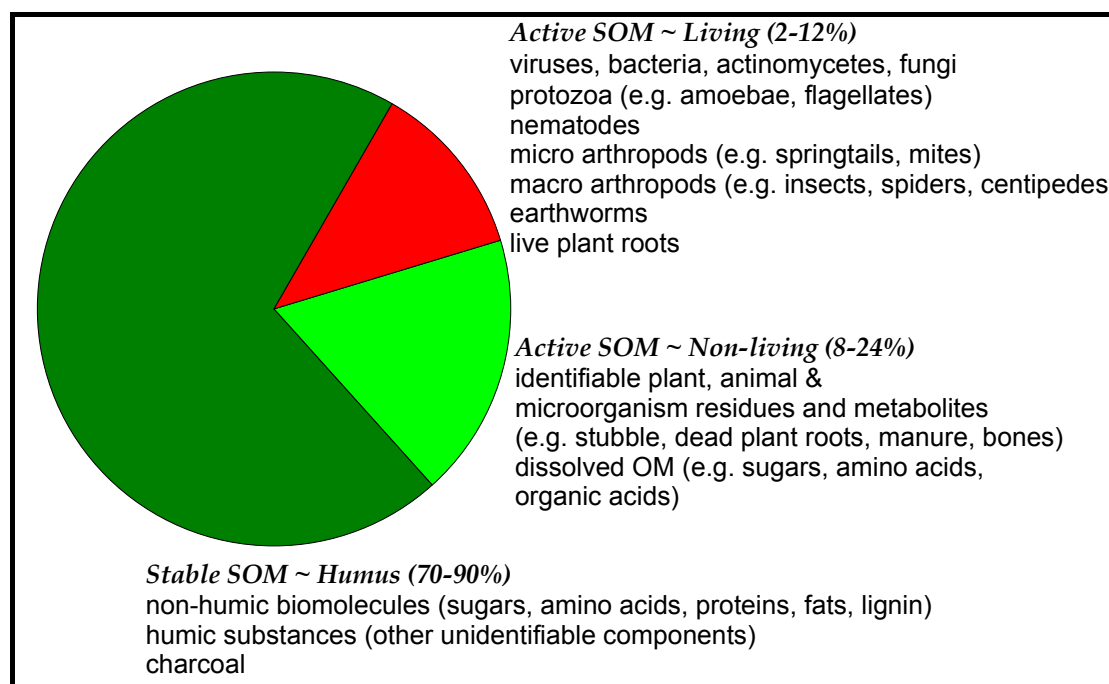
**Table 1. SOM components, along with the percentage of 23 NSWDPI district agronomists who would include each component in a description of SOM to a client.**

SOM component	% NSWDPI	OC?	SOM component	% NSWDPI	OC?
Humus	100	Y*	Earthworms	52	N
Live plant roots	57	Y/N	Dead animals	70	N
Dead plant roots	100	Y/N	Animal manure	87	Y/N
Viruses	35	Y	Leaf litter	96	Y/N
Bacteria	70	Y	Charcoal	52	Y/N
Fungi (including VAM)	70	Y	Standing stubble	43	N
Protozoa (eg amoebae, flagellates)	52	Y	Partly decomposed plant residues	100	Y
Nematodes	61	Y	Sugars	52	Y
Micro arthropods (eg springtails, mites)	26	Y/N	Amino Acids	57	Y
Macro arthropods (eg insects, spiders, centipedes)	22	Y/N	Organic acids (eg citrate, malate)	57	Y

\*The Y or N in the OC? column refers to whether the component is measured in an organic carbon soil test. Those marked Y/N are included either partially or occasionally, depending on the thoroughness of the pre-analysis processing, or the analysis method used.

A major reason for the differences in people's understanding of SOM is that soil sampling, sample processing, and laboratory analysis for SOM excludes many of the components listed in Table 1. Nevertheless, SOM is an diverse mixture of components (Figure 1) with proportions in any given soil sample differing enormously depending on climate, parent material, soil texture, vegetation, animals, microorganisms, topography

and land management. Because there is such a range of components encompassed in SOM, components are often grouped on the basis of their typical breakdown rates in soil and their biochemical makeup. The main groups are stable SOM and active SOM (Figure 1).



**Figure 1. Soil organic matter components and proportions (after Gregorich et al 1997).**

Stable components of SOM, known collectively as humus, are either chemically or physically stabilised. Chemically stabilised compounds are highly decomposed compounds of high molecular weight that few microbes can degrade. Physically stabilised compounds are those bound inside soil aggregates where microbes cannot reach. Carbon dating and isotope abundance techniques have shown that the residence time of humus in soils ranges from decades to centuries. Some compounds such as charcoal are practically inert. Chemical compounds within humus are a mixture of identifiable (non-humic substances) and more complex organic molecules (humic substances).

Modern spectroscopic techniques such as nuclear magnetic resonance spectroscopy (NMR), pyrolysis-gas chromatography-mass spectrometry (PyGCMS), and infrared spectroscopy (IR) characterise the chemical structure of SOM compounds non-destructively. These techniques have largely superseded the chemical extractions used until recently to subdivide humus into humic acid, fulvic acid and humin based on grouping humic substances according to their relative solubility in alkali and acid extractants. These terms can be misleading because humic and fulvic acids do not have a specific chemical formula, like sulfuric acid for example, but are instead groupings of many different chemical forms. The modern techniques mentioned above have shown these older subdivisions are flawed because new compounds are created during the extraction procedure, the subdivisions do not relate well to biological functions of organic matter in soils, and the properties of specific extracted compounds are altered compared with the same compounds in their native state in the soil.

Active or labile SOM (Figure 1) is so named because its components break down over periods ranging from days to years. A further subdivision is often used to discriminate between living and non-living components. Living components can be further subdivided using normal taxonomic classifications. Non-living components are typically subdivided based on their physical characteristics. The first subdivision typically occurs at the point of soil sampling, when choosing areas to be sampled (dead animals, animal manures, tree stumps, ash beds, etc. are avoided), when sampling (standing stubble and large leaf litter are physically removed from the surface), and when processing the samples (large leaf litter and plant roots are usually removed). Further subdivision can then be made by first dispersing the soil sample in water or another solution, then either floating off the 'light fraction' or washing the soil through a sieve retaining the 'macro organic fraction'. Organic compounds dissolved in soil water can also be separated from the bulk soil and further characterised.

### What are the roles and properties of soil organic matter?

Owing to the tremendous diversity of SOM components, its roles and properties in soil can be biological, chemical, physical or environmental. Table 2 lists these roles, along with their perceived importance to 23 surveyed NSW DPI district agronomists, with responses grouped according to the dominant agricultural industry in their district.

**Table 2. Properties of soil organic matter (from Baldock, Nelson 2000), as ranked by NSW DPI district agronomists in order of importance to agricultural industries.**

SOM property	Ranked according to dominant industry in district		
	>67% Cropping	Mixed	>67% Grazing
Water-holding capacity (direct – absorbs water)	1	2	3
Soil aggregate 'glue' (stabilises soil structure)	2	1	4
Water-holding capacity (indirect – enhances structure and pore geometry)	3	2	1
Nitrogen storage and supply	3	3	2
Nutrient cycling (from plant and animal residues to plant nutrients)	3	4	8
Cation exchange capacity	4	6	7
Phosphorus storage and supply	4	9	6
Food for microbes (reservoir of metabolic energy)	5	8	8
Sulfur storage and supply	6	10	5
pH-buffering capacity	7	5	10
Ecosystem resilience (eg resists loss of soil fertility induced by disturbance)	8	7	9
Low solubility (organic materials are not leached from soil)	9	11	14
Chelation helps reduce losses of micronutrient	10	14	13
Chelation reduces potential toxicity of metals	11	12	15
Degrades activity and persistence of pesticides	11	13	11
Dark colour (affects soil thermal properties)	12	15	12

Although all district agronomists rated soil structural and nitrogen supply properties highly, responses differed in their rankings according to what the principal agricultural industries of the district were. Interestingly, the group whose clients are mostly graziers felt nutrient cycling from animal and plant residues was less important than cropping-dominant or mixed groups.

The list given in Table 2 is noticeably biased towards the positive aspects or benefits of SOM. Some negative aspects include pathogenic organisms, allelopathic chemicals formed when some residues decay, hydrophobicity (water repellence), and the nitrogen tie-up by plant litter and stubble with high carbon:nitrogen ratio. There are also newly recognised roles and properties of SOM including carbon sequestration (mitigates greenhouse effect), and suppression of weeds (eg Kremer, Li 2003), nematodes (Dunn 1990) and pathogenic microbial organisms. Suppression may be either general suppression, derived from the overall diversity and activity of the soil biota, or specific suppression against single pathogens (Alabouvette et al 2004).

### **How important is soil organic matter to Australian agriculture?**

Nearly all agriculture, apart from soil-less hydroponics, requires some or all of the components of SOM to function. Since SOM is not a nutrient per se but a diverse collection of components, it is nearly impossible to set 'critical levels' below which the system will be affected. Estimates of the nutrient-supplying capacity have sometimes been used to indicate SOM levels required to produce sufficient nitrogen for a crop, but this source on its own is seldom sufficient to produce optimum yields. Other studies have found levels of SOM components below which deterioration in some physical properties has been linked to crop production (eg Bell et al 1998).

Almost all of the 23 district agronomists surveyed felt that SOM was important to very important to the main agricultural industries of their district, with some commenting that SOM was 'critical' or 'vital'. Feelings tended to be strongest in the group whose districts were dominated (>67%) by cropping. It was also this group where the farmers thought SOM important to the main agricultural industries of the district, according to their agronomists. District agronomists whose client base was dominated by grazing, or where cropping and grazing were of similar proportions, found SOM of lesser importance to their industries. Several agronomists commented that while many of their farmers had good knowledge of SOM and felt it was very important, they were still more focussed on other, more immediate, factors such as pH, soil depth, nitrogen or phosphorus fertiliser applications. While many had adopted minimal tillage operations and stubble retention, some industries were still intent on selling off plant residues, even when it had been demonstrated that the fertiliser value of the residues was greater the price received for the residues. The reason was purely economical – they needed a cash flow when produce price dropped below cost.

### **How can soil organic matter be managed?**

SOM is a dynamic, not a static resource. Many of the properties listed in Table 2 occur only as a result of SOM components being constantly broken down by biota to yield simpler compounds for other organisms, 'glues' for aggregate structure, and nutrients for biota and plants. In natural systems, an equilibrium has developed between the supply of raw materials for breakdown and the breakdown rates dictated by the environment. It is important to realise that once an area is converted from a natural system to a managed cropping and grazing system this equilibrium cannot be the same and should not be expected to be. Numerous studies have found that native SOM levels

rapidly decline by up to 60% within a few years of clearing and cultivation (Dalal, Chan 2001). The decline occurs because

- erosion removes SOM-rich topsoil
- cultivation aerates and breaks down aggregates exposing previously protected SOM to microbial activity
- cultivation dilutes SOM-rich topsoil with SOM-poor subsoil
- bare soils have increased periods of wet soil and increased temperature that benefit decomposition
- there are decreased levels and frequency of organic inputs to the soil
- increased quality of organic inputs allows faster breakdown (eg more nitrogen for microbes and less hard-to-digest lignins).

For land managers to manage their SOM requires continued efforts to counter these processes so that SOM decline may be halted and possibly even reversed. To build up SOM to its potential under a managed cropping or grazing system you need to add organic material, monitor SOM levels and reduce SOM losses.

#### **Add organic material**

- Grow healthy, productive crops and pastures (better use of fertilisers and crop rotations to increase plant biomass).
- Retain as much residue as possible through stubble, roots or even cover (green manure) crops or ley pastures.
- Apply animal manures, biosolids, etc.
- Locate and use off-farm sources of organic matter, such as food-processing wastes and composted products.

#### **Reduce losses**

- Reduce tillage operations by following minimum or no-tillage management systems.
- Retain crop residues by not burning or baling for fodder.
- Reduce periods of bare soil or fallowing by opportunity or response cropping.
- Control erosion.
- Grow plants more resistant to microbial breakdown.

#### **Monitor SOM levels**

- Keep in mind the likely effects of various operations on SOM and check periodically by soil testing.

The simplest check on SOM levels is a soil test for organic carbon (OC). While SOM can be measured directly, the method (called weight loss on ignition) can give errors in some soils (where some minerals also lose weight). OC is more often used as the method is simple and carbon is a relatively constant proportion of SOM, although OC to SOM conversion factors quoted in the literature vary from 1.724 (Van Bemmelen factor) to 2.0. Further conversions are sometimes used if the OC method used is 'Walkley-Black' as this method may not measure all carbon present. In both instances the conversion factors, if used at all, should really be specifically measured for that particular soil type and soil depth. The tediousness of doing this means that people usually report OC without conversion to SOM.

More component- or fraction-specific methods for monitoring SOM are available, but these tests are often not commercially available. One reason is that there are usually no universal interpretation criteria for the tests, and many are highly specific to recent, often transient conditions occurring in the soil. Doran (2000) found that while measurements of soil organisms were sensitive to management and correlated with beneficial soil functions, meaningful tests of soil organisms for use by land managers still required development.

### **What are some practicalities of SOM amendments?**

Soils depleted of SOM can be 'restocked' by adding more and losing less. While options for losing less are becoming more commonplace, particularly reduced tillage, large amounts of external organic amendments are required to make any short-term impact on soil SOM levels. For example, a hectare of soil to a depth of 10 cm weighs approximately 1000 tonnes. If OC is 1% (10 t/ha), then SOM is 1.7% (17 t/ha). To increase SOM to 3.5% (35 t/ha) (OC to 2.0%) would require an extra 18 t/ha of OM (10 t/ha OC). However, simply adding 18 t/ha of, say, animal manure would not give you the 1.8% increase in SOM because 80-90% of the original material is lost in the decomposition process over several years. Carbon is lost as CO<sub>2</sub> to the atmosphere. An alternative is to add material that is already largely decomposed, such as well-matured composts, considered to be equivalent to the more stable fractions of SOM. A review by Gibson et al (2002) found many cases of improvements in SOM, various soil properties and plant productivity as a result of adding such composted material, termed recycled organics. Where the balance of OM inputs exceeds losses over time, SOM should increase. While large additions of recycled organics or animal manures should increase SOM rapidly, improvements in cropping and pasture systems may take five years or more to register an increase in an OC soil test. Increases occur firstly in the smaller active SOM fraction with benefits to soil structure and microbial diversity, then later in the stable SOM.

Building SOM levels is unlikely to be the major aim of broadacre farm management, but should be an important consideration when planning management decisions that affect inputs and losses. For example, the use of organic amendments such as animal manures or recycled organics should be considered not just in a SOM-building sense but primarily as a means of supplying plant nutrients. Other important considerations of manures include cost, carbon:nitrogen ratio, weeds, diseases, contaminants, allelopathic chemicals, and application issues. Applying manures in excess of current plant requirements increases potential for environmental damage from runoff or leaching. Long-term trials (20-120 years) comparing manuring and inorganic fertiliser application (Edmeades 2003) have shown that manured soils had higher contents of SOM and numbers of microfauna than fertilised soils, and were more enriched in several plant nutrients. Manured soils also had lower bulk density and higher porosity, hydraulic conductivity and aggregate stability, relative to fertilised soils. However, on average, there was no difference in long-term effects on crop production between fertilised and manured plots. Why? High input agriculture often 'masks' the relationship between yield or crop health and soil quality parameters such as SOM (Gregorich et al 1997). But fertilisers and pesticides are significant farming costs. The benefits of extra SOM for nutrient supply or pest suppression should be valued as cost-saving or risk-reducing. A case study by Ringrose-Voase et al (1997) was able to link OC levels in several land units of the Wagga Wagga region to land values based on the relationships between OC and economical and productive potential.

### **What about applying microbial products to the soil?**

Our hypothetical soil of 1000 t/ha weight would conceivably contain about 1 t/ha of microbes. Commercially available microbial products claim not to increase this soil fraction directly, but rather to inoculate the SOM with specific organisms or groups of organisms that will rapidly build up in the soil to a point where they will influence soil properties. Favourable environmental conditions of moisture, aeration, temperature, pH, and energy source are required for inoculants to build up in soil; conditions under which native soil biota are also likely to be abundant and strongly competitive against the added organisms (Sullivan 2001). Until more widespread scientific testing of such products in the field is conducted, farmers should be equipped with robust methods of making their own comparisons between current and new technologies, rather than the ad hoc testing that predominates.

### **What about applying humic acids to the soil?**

Humic acid or humate products are generally extracts from Leonardite or lignite, a mineral similar to brown coal, although the extracted compounds differ from those extracted as humic acid from SOM (Ayuso et al 1997). As with microbial products, increasing SOM measurably with humic acid products is unlikely given the scale of addition advocated versus the background levels in SOM. However, there are many claims and some reports in the scientific literature that adding humic acid products to soils may stimulate plant growth and increase yield, possibly due to mechanisms such as delaying precipitation of phosphorus from mineral fertilisers in certain soil types (Delgado et al 2002). Whether such applications will work and, if so, are economical, will be affected by your particular farming system with its unique combination of soil type, climate, landscape, paddock history and economic situation. Determining this requires unbiased, scientifically rigorous, but not necessarily complex, testing and evaluation.

### **Acknowledgements**

A special thanks to the 23 NSW DPI district agronomists who replied to my survey. Thanks also to Trevor Gibson for reviewing this manuscript.

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## Impact of management practices on soil microbial functions in alkaline Mallee soils.

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### Introduction

The Mallee region covers an area of some 5 million ha of SA, Vic and NSW. It is characterised by dune/swale landforms that contain soils that are generally coarse in texture (5-15% clay). The soils are inherently of low fertility and low in organic matter (total carbon = 0.5%). Rainfall is low (250-350mm/year), variable and winter-dominant.

The marginal production characteristics of the Mallee have led to the development of low risk farming systems based on cereal/pasture or cereal/fallow rotations with low fertiliser inputs which have consisted largely of additions of phosphorus. This has resulted in low productivity both in terms of crops and pastures with cereal yields achieving around 50% of the water limited potential (average wheat yield 1.2 t/ha). Productivity can be limited by numerous factors including cereal root diseases and chemical subsoil constraints but overall it is limited in dry years by lack of rainfall and in better rainfall years by lack of fertility. The low productivity combined with other factors including heavy grazing, fallowing and endemic wind erosion have resulted in low returns of organic matter to the soils with subsequent limitations to microbial activity and microbial functions.

A project was initiated in 1997 (Mallee Sustainable Farming Project) to investigate the potential to significantly improve Mallee farming systems. The project was based on the hypothesis that productivity gains of up to 100% could be made by more efficient utilisation of the available rainfall with more intensive cropping and improved tillage and fertiliser strategies. This increase in cropping intensity, with increased productivity in association with limited grazing and zero tillage, would result in a substantial increase in the return of organic matter to the soil with an associated increase in microbial activity and function, which are essential for the long-term sustainability of any farming system. In these coarse-textured Mallee soils with low levels of organic carbon and limited opportunity for protection of organic matter it was hypothesised that an increased flow of microbially available carbon would quickly provide a significant improvement in microbial biomass, microbial activity and populations of microbial functional groups involved in nutrient cycling and other microbial functions.

This paper reports on results on the impact of agronomic management on soil microbial function from both the Mallee Sustainable Farming Project and an earlier experimental program at Avon SA.

### Methods and Materials

#### *Avon field site*

A long-term field trial has been maintained at Avon, South Australia (34° 14' S, 138° 18' E) since 1979. The climate is Mediterranean, characterised by hot dry summers and a winter-dominant, average annual rainfall of 350 mm. The soil is an alkaline calcareous sandy loam, classification Gc1.12 (Northcote et al 1975), or solonised brown soil (Stace et al 1968), or calcic xerosol (Dudal 1968). Soil chemical properties at the start of the

trial were pH(water) 8.2, organic carbon 1.6%, total nitrogen 0.14%, Colwell bicarbonate-extractable phosphorus 65mg/kg, and calcium carbonate 8%. Particle size distribution (%) is clay 12, silt 3, fine sand 34 and coarse sand 42.

#### **Waikerie field site**

A field trial was established at Waikerie, South Australia (34° 17' S, 140° 02' E) in 1998. The climate is Mediterranean, characterised by hot dry summers and a winter-dominant average annual rainfall of 260 mm. The soil is an alkaline calcareous loamy sand, classification Um5 (Northcote et al 1975), or gray-brown or red calcareous soil (Stace et al 1968), or alfisol (Dudal 1968). Soil chemical properties at the start of the trial were pH(water) 8.6, organic carbon 0.68%, total nitrogen 0.05%, Colwell bicarbonate-extractable phosphorus 12 mg/kg, and calcium carbonate 0.4%. Particle size distribution (%) is clay 6, silt 1, fine sand 43 and coarse sand 47.

#### **Analytical details**

Microbial biomass carbon, nitrogen and phosphorus levels were determined using chloroform-fumigation extraction methods (Joergensen 1995). *In situ* microbial activity measurements were performed using a portable infrared gas analyser (EGM-1 Environmental gas monitor, PP Systems, Hertfordshire, UK). Carbon and nitrogen mineralisation potentials were estimated using a 21-day laboratory incubation method (Gupta et al 1994) and the off-season nitrogen mineralisation values were calculated from the deep nitrogen measurements (mineral nitrogen measurements in one metre profile) at the harvest of a crop and seeding of next season's crop. Mineral nitrogen levels were measured on 2M KCl soil extracts (1:3 soil to extractant ratio) using the method described by Rayment and Higginson (1992). Soil enzyme activities were determined using the p-Nitrophenol and other colourimetric based methods originally described by Alef and Nannipieri (1995) with modifications to suit our soils.

## **Results and discussion**

#### **Management impacts on non-symbiotic N<sub>2</sub>-fixation.**

At an experimental site at Avon SA a nitrogen budget was calculated for 17 years of continuous wheat. The wheat crops were direct-drilled with no added nitrogen fertiliser and all stubble was retained on site. After accounting for nitrogen removed in the grain and the change in total soil nitrogen there was 334 kg of extra nitrogen/ha that was unaccounted for (Table 1). One possible explanation for the extra nitrogen is through an input from non-symbiotic N<sub>2</sub>-fixation (NSNF). An evaluation of potential NSNF for the Avon region has been made by Gupta et al (2002) who calculated an average nitrogen input of 10-15 kg/ha for the period from January to June based on rainfall/temperature relationships and assuming the requirement for available carbon would be adequately met from stubble remaining after harvest. Given that there is also the opportunity for some NSNF during spring, the modelled NSNF figures corresponded closely to the unaccounted nitrogen in the nitrogen budgets. The amount of unaccounted nitrogen is agronomically very significant in that it is equivalent to approximately 30-50 % of the nitrogen exported in wheat crops given expected district yields of 2-3 t/ha. It is probable that the carbon inputs from the management practices of intensive cereal cropping and stubble retention have directly contributed to an increase in NSNF at this site.

**Table 1. Nitrogen budget for 17 years of continuous wheat (1979-1996) with no added nitrogen fertiliser, Avon SA.**

Nitrogen removal in grain			Change in soil nitrogen			Unaccounted nitrogen	
Total wheat yield (t/ha)	Average N content (%)	Total N exported (kg)	Total soil N 1979 (%)	Total soil N 1996 (%)	Change in soil N (kg)	Total N (kg/ha)	N/ year (kg/ha)
19.7	2.0	394	0.14	0.135	-60	334	19.7

**Management impacts on nitrogen and phosphorus dynamics**

Following four years of treatments, the ‘improved systems’ with more intensive cropping and higher fertiliser inputs at Waikerie produced significant improvements in soil microbial measurements compared with the district practice of low fertiliser, wheat-pasture treatment (Table 2).

**Table 2. Microbial biomass carbon and nutrient levels in the surface soils (0-10 cm) of selected treatments at Waikerie core site (after four years).**

Cropping system	MB-C (kg C/ha)	MB-N (kg N/ha)	MB-P (kg P/ha)	Microbial activity <sup>§</sup> (g CO <sub>2</sub> /m <sup>2</sup> /hour)
Pasture-Wheat (DP)	265a <sup>#</sup>	26 a	16.0a	0.105
Pasture-Wheat (HI)	370b	43 cd	21.0b	0.185
Legume-Wheat (HI)	370b	46 d	13.0a	0.210
Canola-Wheat (HI)	357b	36 bc	16.5a	0.175

<sup>#</sup>Values in each row followed by the same letter are not significant at P<0.05.

<sup>§</sup> Average values from six *in situ* respiration measurements made in each experimental plot.

Microbial activity increased on average for all ‘improved systems’ by 81% over the district practice treatment and reflected the increase in plant biomass production with these treatments. Microbial biomass carbon increased 40% with higher fertiliser inputs and this increase was not influenced by rotation. Microbial biomass nitrogen was significantly influenced by rotation with an increase of 38% following continuous crop with no legume component and by 73% where there was a legume component in the rotation when compared with district practice.

The increase in microbial carbon following the ‘improved systems’ indicates that the carbon from the extra stubble and root residues is not just being respired as CO<sub>2</sub> but a proportion is being maintained within the microbial biomass. A significant increase was also seen in the populations of cellulolytic bacteria and fungi and nitrifying microorganisms, functional groups of microorganisms involved in carbon and nitrogen cycling. A key functional outcome of this is an improved net mineralisation of nitrogen between harvest and sowing (Table 3). Following a summer with a number of rainfall events (2000-01), up to an extra 30 kg/ha of nitrogen was mineralised under the ‘improved systems’ compared with the district practice. The higher rate of mineralisation under the ‘improved systems’ is also likely to provide benefits within crop, particularly during spring when extra nitrogen can be mineralised in response to rainfall events. This has the potential to provide better matching of nitrogen availability to plant requirement in the low and variable rainfall environment of the Mallee and to

allow the crop to optimise benefits from spring rains with less of the risks involved with higher fertiliser nitrogen levels.

**Table 3. Amount of nitrogen mineralised (kg N/ha) during the off-season (summer and spring) as influenced by cropping system type at Waikerie SA.**

Treatment 2000	Off-season mineralisation (Nov 2000-May 2001)	Treatment 2001	Off-season mineralisation (Nov 2001-June 2002)
Wheat-DP <sup>#</sup> (1)	10.3	Pasture-DP	18.5
Wheat-HI <sup>§</sup> (3)	14.0	Pasture-HI	37.0
Wheat-HI (8)	36.1	Peas-HI	35.0
Wheat-HI (9)	24.7	Canola-HI	23.1
Wheat-HI (10)	34.0	Wheat-HI	23.0
Canola-HI (11)	41.5	--	

# DP: district practice fertiliser rate comprising 10 kg P/ ha; 5 kg N/ha.

§ HI: high fertiliser rate comprising 15 kg P/ha; 27 kg N/ha (N excluded for pulse crops).

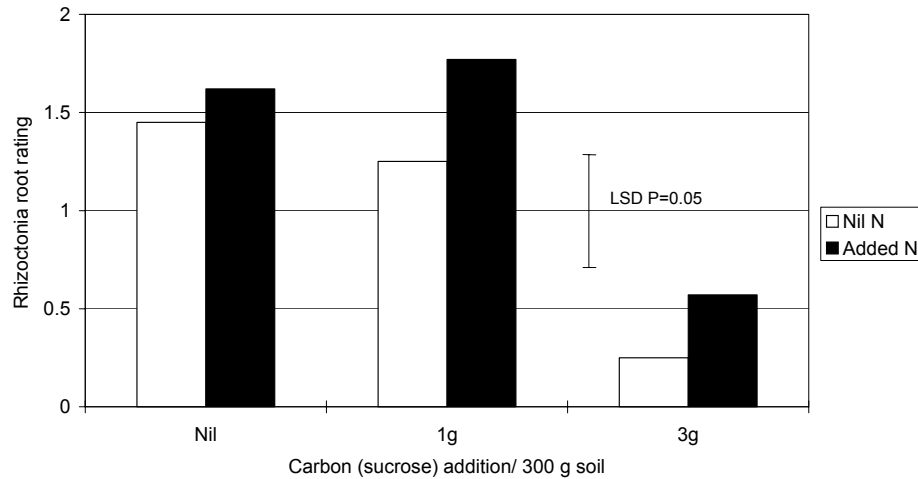
The greater reserves of nitrogen in the microbial biomass also provide a buffer against leaching losses that have been measured at up to 50kg N/ha in the Mallee following large summer rainfall events. Improved microbial biomass levels can contribute significantly to the long-term nutrient efficiency of these systems as indicated by an assessment of mineral nitrogen from a depth of 2-6 metres which showed up to 600 kg/ha of accumulated nitrogen following typical district practice management.

An assessment of the activity of phosphatase enzymes has shown an increase of up to 29% with the high microbial activity of the 'improved systems'. The dynamics of phosphorus in the soil make it difficult to quantify, directly, any improved phosphorus availability, but the higher enzyme activity coupled with increased available phosphorus levels provide strong evidence for this.

### **Management impacts on suppression of cereal root diseases**

The level of disease-suppressive activity in soils against fungal diseases is a function of the population, activity and composition of the microbial community. All soils have an inherent level of suppressive activity, but this level can be significantly modified by management practices used within a farming system. At Avon SA, disease suppression increased from a low to high level over a period of 5-10 years following a change in management practices to full stubble retention, limited grazing and higher nutrient inputs (Roget 1995). The increase in suppression provided complete control of the soil-borne diseases rhizoctonia (Roget 1995) and take-all (Roget 1997). Soils with high levels of disease suppression have also been identified in commercial farms across SA and Victoria (Roget et al 1999). The management factors consistently related to soils with improved disease suppression included intensive cropping, stubble retention, limited grazing, limited or no cultivation and above average yields (high water use efficiency). These management practices increase carbon inputs that result in changes to the composition and activity of the soil microbial community over time. These changes result in greater competition for soil resources that, along with predation and inhibition of pathogens, lead to increased suppression.

In the short-term (one to three years), the effectiveness of the disease-suppressive activity already developed in the soil can be influenced by the availability of mineral nitrogen, particularly during the summer and early autumn period. As the amount of available nitrogen (ie nitrate nitrogen) in the topsoil increases during this non-crop period, the disease suppression occurring in the following crop season decreases. The impact of higher soil nitrogen levels and its relationship to added carbon is shown in Figure 1. The underlying suppressive activity may not be lost, but may not be expressed effectively in the presence of high mineral nitrogen levels.



**Figure1. Impact of added carbon and nitrogen on the level of rhizoctonia root damage (suppression) on soil with a natural level of inoculum of *Rhizoctonia solani*. Nitrogen added as urea at 15% of carbon addition.**

Any factors that result in the accumulation of higher available nitrogen levels for longer periods of time will tend to curb the effectiveness of the suppressive activity of the soil. From the point of view of strong suppressive activity, a good farming system includes a productive intensive cropping system to provide export of nitrogen and a strong nitrogen sink through a supply of biologically available carbon (production and retention of residues) to maintain higher levels of microbial carbon turnover. This does not necessarily equate to a low fertility system but one in which the timing of nitrogen availability is more synchronised with the crop requirements. In such systems, early season nitrogen availability may depend heavily on fertiliser nitrogen but later crop requirements would be supplied by net nitrogen mineralisation through microbial turnover.

### Conclusions

Soil microbial functions, including non-symbiotic N-fixation, nitrogen and phosphorus availability and disease suppression can be significantly improved through changes in onfarm management in the Mallee. Management changes that increased productivity and resulted in the return to the soil of higher levels of microbially available carbon were the driver for greater soil microbial activity and improved microbial functions. The coarse-textured Mallee soils are likely to be particularly responsive to increased carbon inputs due to the low soil carbon levels and rapid turnover of added carbon as a result of the limited protection offered by these soils. The higher carbon inputs from the ‘improved systems’ will most likely determine the upper limit of the improvements in microbial function. At Waikerie these improvements have been monitored for five

years, and it is not clear at this stage if the maximum level of microbial function has already been obtained or if with time there is still further improvement to come.

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# Soil structure and soil biota: their interactions and implications on soil health

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## Introduction

Soil is a living system and has to be managed as such in agriculture to improve sustainability. The importance of harnessing the beneficial functions of soil biota in agriculture has been recognised (eg Lee, Pankhurst 1992), but to do this we need better knowledge of the factors controlling the distribution, abundance and activity of soil biota. These factors are largely controlled by the spatial heterogeneity of the soil environment which is determined by soil structure. Hitherto, this important aspect of soil attributes has been largely ignored (Young, Ritz 2000). Such knowledge is a pre-requisite if processes mediated by soil biota are to be harnessed for agricultural production. Soil has been largely studied as a homogeneous material, where soil chemists routinely grind up the soil prior to performing analyses and microbiologists carry out their study largely on laboratory culture media.

This paper reviews the interrelationships between soil structure and soil biota which affect soil functions. It outlines the importance of soil structure on abundance, diversity and activity of soil biota, looks at the effects of soil biota in modifying soil structure, and discusses the importance of soil biota interactions on soil health and the role of soil management practices in harnessing the beneficial functions of soil biota.

## Soil structure impacts on soil biota

### *Soil as a habitat for living organisms in agro-ecosystems*

Soil provides the habitats for a whole range of living organisms (Lee, Foster 1991). Soil biota (flora and fauna) have been traditionally classified in terms of their physical size, namely micro-, meso- and macro- (Lee, Foster 1991). The architecture and the physical environment of the ecological niches are controlled by soil structure. Structure can therefore influence the type of organisms (diversity), the population density that can exist in a particular soil (abundance), and the activity of organisms. All these factors affect the biological fertility of the soil.

Soil structure is often defined as the size, shape and arrangement of aggregates and the spaces or pores in between at a given time. Therefore soil structure can be described both in terms of the pore system as well as the arrangement of primary soil particles into hierarchical structural states (Kay 1990). However, from a functional point of view in terms of soil as a habitat and the activities of the inhabitants, it is more meaningful to focus on the pore space system which can be described in terms of total porosity, pore size distribution and continuity of the pore systems.

The concept of 'habitable pore space' suggests that there is a relationship between the size of organisms and the zones of soils they are physically able to inhabit (Young, Ritz 2000). As Figure 1 shows, soil contains pore spaces and particles (aggregates) that vary in size over seven orders of magnitude ( $<10^{-7}$  m to 1 m), so it provides a heterogeneous environment. Pore size distribution differs according to soil type, thus offering habitats for a diverse range of organisms. Clay soils provide better habitats for bacteria because of higher micro-porosity, so there are more micro-niches than sandy soils (Foster 1994).

Research has shown a clear correlation between soil pore volume classes and nematode and bacterial biomass in some cases (eg Hassink et al 1993).

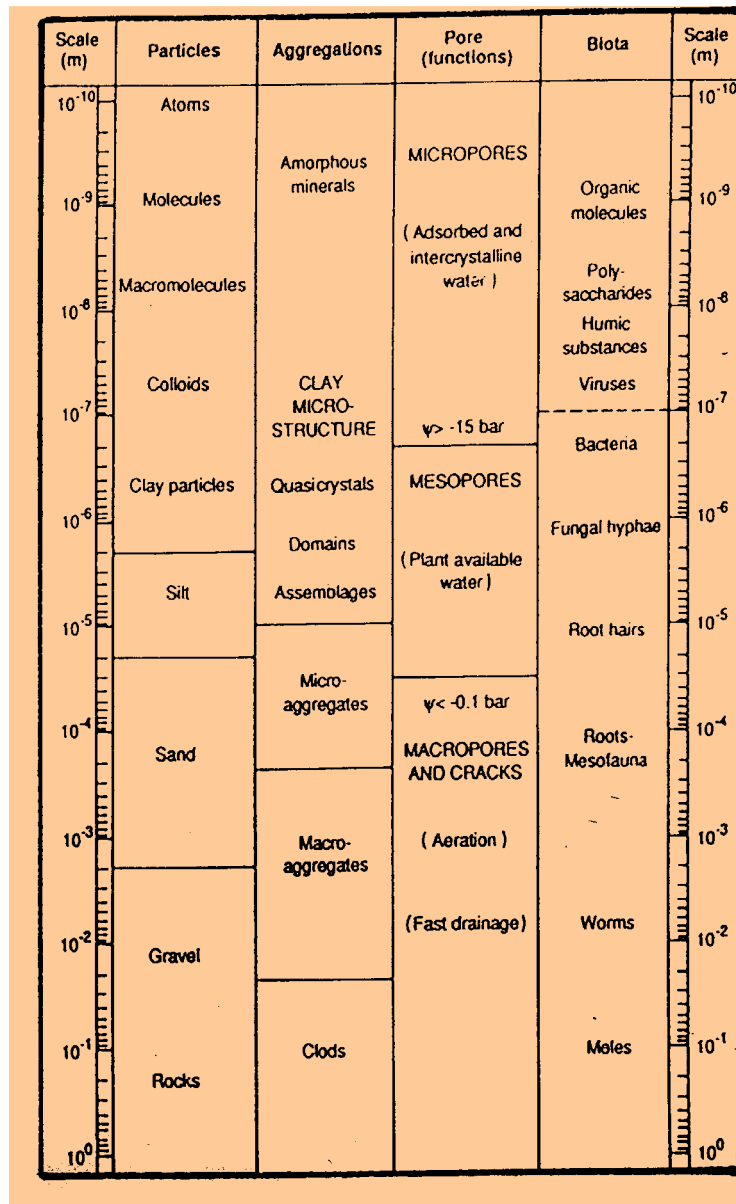


Figure 1. Soil biota in relation to different components of soil structure with different sizes and scales (Kay 1990).

**Soil structure**

The interaction of soil structure with soil water content determines a number of important physical properties and processes which in turn define the physical environment of the habitat (Smiles 1988).

**Soil water availability and permeability**

Soil water is held at different energy levels, depending on the pore size, so that water in smaller pores is held more strongly than that in larger pores. The relationship between soil water content and the energy with which the water is held by the soil (soil water potential) is referred to as ‘soil moisture characteristics’ and differs according to soil type. It defines the availability of water or, in other words, the moisture stress a



particular organism is subjected to at a particular soil water content. For a salt-free soil, the water potential represents the energy that an organism must expend to retain (extract) water against the attraction of the soil. It also controls the relative humidity of the soil atmosphere. The lower the water potential and the drier the soil, the greater the stress experienced by the organism. The moisture characteristics of sandy and clay soils are very different, so at the same water content, their water potentials which determine water availability will vary considerably. Ecologically, it is more meaningful to express soil water status in terms of potential rather than actual soil water content. The pore size distribution of a soil can be calculated from its moisture characteristics (Marshall, Holmes 1979). The relationship between permeability and soil water content is also determined by pore size distribution of a particular soil. As water can move much faster through larger pores than smaller pores, sandy soils have higher permeability than clay soil and the rate of water movement decreases rapidly as a soil becomes drier.

#### **Soil aeration**

The composition of soil atmosphere, and hence its aeration status, is governed by gaseous diffusion processes between the above ground atmosphere and the soil, which depend on the air-filled porosity of the soil (Smiles 1988). Air-filled porosity varies with the soil water content and the pore size distribution of the soil. Gaseous diffusion through the water phase is about four orders of magnitude less than through air and is therefore negligible. This means that at the field scale, soil is a heterogeneous environment with aerobic and anaerobic zones dispersed through out the soil volume.

#### **Soil strength**

While many of soil organisms live in and depend on pre-existing soil pores, some larger organisms can make new pores. Plant roots, earthworms and termites fall into this category. Their abilities to deform soil depend on the mechanical strength of the soil. Mechanical strength is a function of both soil structure and soil water content, and increases with bulk density (mass of dry soil per unit volume) and with decreasing soil water content.

#### **Thermal properties**

Varying soil temperature regimes within the soil environment, both diurnal and annual, directly affect biological activities. However, soil temperatures are modified by thermal conductivity and thermal capacity of soil, both of which are functions of soil structure (bulk density) and soil water content (Smiles 1988).

#### ***Soil structure and biological processes***

Soil is a heterogeneous environment, spatially and temporally. Many biological activities and processes in soils are modulated by soil structure, particularly via the interactions between soil structure and water.

#### **Activity and mobility**

Different soil organisms have different activity and survival ranges in terms of soil water potential. For instance, bacteria are most active in the soil moisture potential range of -0.01 and -0.03 MPa (Lavelle, Spain 2001). In the case of earthworms, activity in surface soil ceases when drier than -150 kPa (Baker et al 1993). Soil moisture characteristics control the mobility of many small soil animals such as nematodes, motile bacteria and aquatic phycomycetes which are restricted to existing water-filled soil pores. Existence and activities of these organisms depend on availability of sequences of water-filled pores of the right sizes to permit their passage (Papendick, Campbell 1985). This is pre-determined by soil structure (pore size distribution) at a particular soil water potential. Filamentous fungi are less restricted by such factors. Permeability affects the rate of soil water movement and solute supply, and controls

many biological activities such as wilting and germination of plants, and hatching of nematode cysts and spores (Smiles 1988).

#### **Nitrogen mineralisation and cycling**

Soil is a complex interconnected framework of pores and solids. At field scale and under particular soil water status, aerobic and anaerobic zones often exist in close proximity. This controls the distribution of the types of organisms and has direct impact on processes of nitrification, denitrification and decomposition processes and therefore nitrogen/carbon cycling. Spatial heterogeneity of soil structure, distribution of plant roots and other organic debris determine how soil functions and the degree of heterogeneity of microbial and nitrogen transformation throughout the profile (Young, Ritz 2000).

#### **Prey and predation relationship**

Organisms residing in pores of appropriate size are protected from predation by organisms of larger dimension since the latter are denied physical access to their prey. This has been demonstrated in the case of prey-predator relationship for *Pseudomonas fluorescens* and ciliate protozoan *Colpoda steinii* (Wright et al 1993). When the bacteria are predominantly located in the smaller pores (<6 µm), they better survive protozoa predation. Such interaction is influenced by pore size distribution and has impact on nutrient cycling, mineralisation processes and disease prevalence via its effect on composition of the food web (Lee, Pankhurst 1992).

#### **Substrate availability and carbon sequestration**

Substrates existing in pores of appropriate size are protected from breakdown by soil microorganisms because of their physical inaccessibility. As a result, even labile forms of organic carbon can be sequestered in soil. The effect of tillage and other forms of soil disturbance is to expose these substrates to the microbial population and results in the commonly observed accelerated soil organic carbon decline under conventional cultivation practices.

### **Soil biota impacts on soil structure**

It is well documented that soil biota can directly and indirectly influence soil structure. Directly, bacteria stabilise soil aggregates by their polysaccharides gel and, in the case of fungi, by the physical entanglement of their filamentous hyphae. Direct correlation between macroaggregate stability and hyphal length has been established (Tisdall, Oades 1980). Bacteria in colonies are often found within microaggregates and in association with dispersed clay particles (Foster 1994), and their carbohydrate gel surrounding them is important in microaggregate formation (Emerson et al 1986). Mesofauna such as protozoa and nematodes do not have direct effects on soil structure but can indirectly affect aggregate structure through their regulation of bacterial and fungal populations.

While the microbes largely survive in the pre-existing soil structure, larger soil biota are known to modify soil structure in creating aggregates and burrows. Earthworms, termites and ants have been called the soil 'ecosystem engineers' because of their ability to modify soil structure. Casting and burrowing activities of earthworms have been well documented since the seminal work of Darwin (1811) for *Lumbricus terrestris* (Lee, Foster 1991). Earthworms are by far the most effective in transporting of soil material within the soil profile (Table 2). However, given the large range of values, the ability of earthworms to move material varies with species, abundance, abiotic factors and management.

**Table 2 Soil turnover rates by different soil biota as recorded in the literature (modified from van Vliet, Hendrick 2003)**

Soil biota	Soil turnover rate (t/ha/yr)
Enchytraeids	0.75-21.8
Earthworms	40-1000
Termites	0.75-45
Ants	0.42-10

## Implications for soil health

Soil structure is defined as ‘the spatial heterogeneity of the different components or properties of soil’ (Dexter 1988). Ecologically, soil consists of very heterogeneous habitats characterised by hot spots of biological activities such as rhizospheres, drilospheres, and aggregatospheres (Lavelle, Spain 2001). A defining signature of a soil/plant/soil biota system may well be this heterogeneity (Young, Ritz 2000). Through soil structure, soil provides the physical space that accommodates many organisms in ecological niches. Through interactions with soil water, soil structure defines a range of environmental factors which are important for the growth and activities of its inhabitants. In agriculture, a good soil habitat is one that provides favourable environment for growth and development of plants and associated organisms important for ecosystem functioning. Soil structure determines the abundance, diversity and activity of the soil biota and therefore biological fertility of the soil.

On the other hand, soil biota can modify soil structure by stabilising as well as creating soil structure. Bioturbation, which results in biogenic mixing of soil materials, is an important process in soil formation. It assists interactions with inorganic fractions eg burial of litter, and is important for nutrient cycling and nutrient availability. The products of soil biota activities, eg aggregates, burrows and galleries, often have dominant effects on soil physical properties such as infiltration, aeration and soil strength (Lee, Foster 1991). Therefore, the interactions of soil structure and soil biota determine all the three aspects of soil health, physical, chemical and biological.

In natural ecosystems, many soil structural features are adaptive strategies by soil biota to increase the suitability of the soil habitat for survival (Wolters 2000). They exist in dynamic equilibria with soil biota activities, counteracting natural processes of soil chemical and physical degradation. For example, integrity of soil aggregates found under natural ecosystems depends on the continuous supply of carbohydrate gel from microbes which can be severely curtailed under annual cropping (Foster 1992). The modified soil structure found under ant mounds, (lowered bulk density and increased soil porosity) accelerates infiltration of water, alters temperature gradients and modifies pH, and is maintained by the continuous activities of the ants. These activities change the decomposition rates and functional structure of microbial communities in the soil (Wolters 2000). The transmitting burrows created by native anecic earthworms ensure that little surface runoff and soil loss occur, and are the preferred areas of root proliferation in the highly acidic subsoil (Chan 2004). Risk of runoff increased dramatically after three years of conventional tillage due to the disappearance of these burrows.

These examples illustrate the impact of agricultural management practices on soil structure - soil biota interactions. In agriculture, management practices can have direct

effects on soil biota (Clapperton et al 2003) and indirect effects due to changed soil structure. Tillage and associated field machinery traffic destroy habitats of earthworms and result in soil compaction that can render the soil too strong for these ecosystem engineers to survive. The undesirable effects of conventional management practices on soil structure have been recognised and are main impetus for conservation tillage systems. An essential feature of the latter is the harnessing of the beneficial functions of soil biota. Hitherto, attention has been focussed only on the effect of soil structure on plant growth. However, such understanding should be extended to the effect of soil structure on soil biota and their interactions. Ecological functioning by soil biota has to be studied in the context of structural heterogeneity. Better understanding of these interactions will help us develop management practices to manipulate soil biota (Elliott and Coleman 1988).

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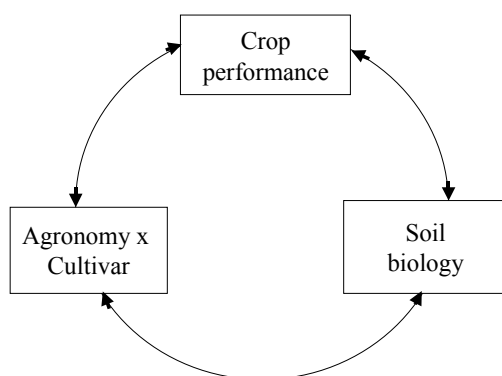
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# Impact of management practices on soil biota activity on acidic clay loams in NSW

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## Introduction

All management practices affect the numbers, diversity, and activity, both spatially and temporally, of various components of the soil biota. Apart from plant-associated organisms such as pathogens or symbionts, studies on the numbers and diversity of soil biota have, in themselves, rarely improved agricultural decision-making. What is important is how the interactions between management and soil biology affect crop performance (Figure 1). What the study of soil biology offers us is insight into how we might improve agronomic practices and cultivars: that is, how we might accelerate the adaptive management of Australian farms that has enabled rises in productivity to keep ahead of the steadily falling terms of trade over the last few decades.



**Figure 1. Interactions among management, crop performance, and soil biology that can accelerate adaptive management.**

On the acidic loams of southern NSW, there is clear evidence that changes in agronomic practice have improved the productivity and sustainability of farming systems by influencing (among other things) important components of soil biology. There is substantial evidence, from fumigation experiments and from as yet unexplained agronomic responses in field trials, that there is potential to capitalise further on interactions between crop management and soil biology. It is likely that studying the rhizosphere, the interfacial region between roots and soil, will foster the fastest progress.

There is equally clear evidence that management strategies designed to improve soil 'health' may not improve productivity and can even threaten other aspects of sustainability. Growers and agricultural extension specialists should be aware that the concepts of soil 'quality' and soil 'health' that have gained popularity among soil biologists and microbiologists in the last decade or two have received trenchant criticism (eg Sojka et al 2003). The problem is that these concepts are inherently subjective, without agreed meaning, as is evidenced by trying to endow them with units. In this paper I elaborate on these points, and especially emphasise the strong

interactions between cultivars, agronomic practice, and soil biology that offer promise of further substantial improvements in sustainable productivity.

### **Dryland farming systems on acidic loams: recent trends**

The dry-land farming system on acidic loams in southern NSW is predominately mixed farming enterprises based on cereal and sheep production, ranging in annual rainfall from 650 mm in the east to 400 mm in the west. The average farm size of around 1010 ha comprises 67% pasture and 33% crop, most of which (around 68%) is cereal, predominately wheat, the remainder being broadleaf break crops such as canola, lupin and field pea (Connell, Hooper 2002). These averages vary considerably within individual farms and paddocks, with flexibility to adjust to seasonal conditions and commodity prices. The large proportion of farm area devoted to legume-based pastures in sequence with a cropping phase is an important feature of these systems. The pasture phase results in a build-up of labile soil organic matter and biologically fixed nitrogen, as well as providing different opportunities to control problem weeds, although the acidifying effects of legume pastures must be controlled with lime. Further, when perennial species are used, especially lucerne, they enable the capture of water from the deep subsoil that may have drained beyond the reach of crop roots, thus improving the hydrologic performance of the system. The long-term productivity trends for wheat within these systems have been among the highest in Australia, especially in the 1990s (Angus 2001), and have been accompanied by several changes that improve physical resource sustainability, including increased adoption of minimum tillage (80% of crops in 2001 – Connell, Hooper 2002), increased use of lime (Angus 2001), and a move to more perennial pastures (ABS Agstats 2001). There has been recent concern about the risks of simplifying the rotations with shorter or less frequent pasture phases and continuous alternation of wheat with a broadleaf crop (eg wheat-canola-wheat-canola) (Wolfe, Cregan 2003). The static and declining canola yields in the area may indicate increasing disease problems associated with these intensive rotations, and despite impressive improvements, wheat yields in most shires remain below the water-limited potential. Future trends are likely to include an increase in the area of dual purpose winter wheat for grazing and grain production, larger farms and a continued focus on improved productivity to keep pace with the continuing 3-4% pa decline in the terms of trade.

### **Impacts of crop and pasture sequence on soil biology and crop production**

#### ***Legumes, nitrogen and organic matter***

The nitrogen-fixing activities of the legume-*Rhizobium* symbiosis in legume-based pastures and pulse crops represent one of the greatest benefits of their inclusion in the farming system. Because the amount of nitrogen fixed by legumes is related to legume biomass – 20 to 25 kg of shoot nitrogen fixed for every tonne of legume biomass produced (Peoples, Baldock 2001, Evans et al 2001), any management practices that enhance the growth of legumes will increase this nitrogen benefit within the system. Recent estimates suggest that wheat crops derive up to 40% of their nitrogen requirement from preceding legume crops (Evans et al 2001) and the impacts of various management strategies on the mineralisation and efficient use of this organic nitrogen source remain an active area of research (GRDC Project CSO000030). Importantly from a soil biology viewpoint, the frequency of highly productive legume-based pastures in the farming system of the area provides greater inputs of labile carbon and nitrogen to the system, than provided by crop residues, for sustaining microbial activity.

### **Break crops and soil biology**

The substantial productivity improvements in wheat in southern NSW in the last decades arise from the control of cereal root diseases, primarily take-all (*Ggt*), by the use of broadleaf break crops such as canola, lupin and field pea, and the spray-topping of grass hosts in pastures in the year prior to cropping. Reductions in other soil-borne diseases such as crown-rot (*Fusarium pseudograminearum*) and stubble-borne diseases such as yellow leaf spot (*Pyrenophora tritici-repentis*) have also contributed to these benefits. Break crops result in average wheat yield increases of 19%, and cereal crops freed from root disease respond more reliably to nitrogen fertiliser and use more of the subsoil water and nitrogen, which may otherwise be leached (Angus et al 2001). The combination of break crops, improved nitrogen management and liming has lifted productivity to a new level. Diseases common to wheat and canola (eg rhizoctonia, lesion nematodes *Pratylenchus neglectus*) or other inhibitory organisms may continue to reduce yield in the area, as suggested by significant responses to soil fumigation (see later).

Non-hosting of disease pathogens is not the only way in which break crops can influence the soil biology and crop production. Such crops can also influence the populations of specific rhizosphere organisms which may compete, antagonise or suppress pathogens, influence nutrient transformations or affect plant growth directly. We have conducted much research in southern NSW to investigate the impacts of canola on some of these other aspects of soil biology. A brief summary follows.

#### **Biofumigation**

When wheat grew better after canola than after other broadleaf break crops in southern Australia during the early 1990s it prompted speculation that chemicals called isothiocyanates (ITCs) released from canola roots may suppress disease organisms, a process termed biofumigation. Early pot studies showed that cereal pathogens such as take-all were sensitive to ITCs, but subsequent field studies found only limited benefits to following wheat crops from a biofumigation effect (Kirkegaard et al 2000, Smith et al 2004). Similar studies in northern NSW also failed to find direct biofumigation effects on crown rot, although the disease was lower after canola than after chickpea (Kirkegaard et al 2004). It now seems that any impacts of canola on root disease in addition to the non-host effect, relate to more general changes in rhizosphere bacteria (eg Rumberger, Marschner, 2003) or antagonistic organisms such as *Trichoderma* (Kirkegaard et al 2004) rather than direct killing of disease inoculum by ITCs.

#### **N mineralisation**

More evidence that break crops can significantly affect soil biology was that different amounts of mineral nitrogen accumulated in the summer fallow following different crops (Kirkegaard et al 1999). Surprisingly, more mineral nitrogen accumulated following canola (94 kg/ha) than following field pea (50 kg/ha) and the differences could not be explained by the amount, nitrogen content or carbon:nitrogen ratio of the crop residues. What caused this effect is uncertain, but populations of organisms associated with nitrogen-cycling such as free-living nitrogen-fixing bacteria, *Azospirillum* species and ammonium-oxidising bacteria were generally lower following canola, while total bacterial populations did not differ. The effect was shown to be transitory under laboratory conditions which accelerated mineralisation, but in the field it strongly influenced the growth of subsequent wheat crops.



### **Mycorrhizal fungi (AMF) and other beneficial organisms**

The realisation that canola may release fungicidal compounds from the roots prompted speculation that more frequent use of canola may influence the levels of beneficial organisms such as arbuscular mycorrhizal fungi (AMF) or *rhizobia*. Low colonisation of wheat crops following canola or fallow results in poor growth of wheat on low phosphorus soils in the northern wheat belt (Thompson et al 2001). Recent studies on wheat in southern Australia showed that lower AMF colonisation in wheat following brassicas (and fallow) did not reduce growth or yield in subsequent wheat crops despite strong phosphorus limitations on crop growth and yield (Ryan, Angus 2003). The authors hypothesised that AMF may even be parasitic on these crops prior to spring, utilising carbohydrates from the seedlings to support their growth. Surprisingly, it is possible that wheat in southern NSW may grow better after canola partly as a result of lower AMF colonisation. Smith et al (2004) found no evidence that canola influenced the nitrogen-fixing capacity of subsequent pea crops. Populations of the disease antagonist *Trichoderma* have increased more in wheat following canola than following chickpea or wheat (Kirkegaard et al 2004).

### **Tillage management and the soil biology: good and bad**

Conservation cropping systems involving direct-drilling and stubble retention have been developing for over 30 years in Australia, initially in response to concerns about fuel costs and more recently to reduce erosion, maintain organic matter and soil structure, and conserve water for crop use. During that period, the systems have been tuned to specific regions, especially by growers adapting sowing equipment to their circumstances (Cornish, Pratley 1987). In southern NSW, particularly in higher rainfall areas, adoption has been slower and the benefits to crop yields flowing from improved soil conditions more difficult to demonstrate (Kirkegaard 1995, 2001).

Research carried out over the last 15 years at a site near Harden in southern NSW helps illustrate the role of soil biology in the productivity and sustainability of conservation cropping. From the outset, a consistent problem with direct-drilled wheat in southern Australia was the reduced early vigour of crops compared with those sown into cultivated soil, a phenomenon which has not diminished over time (Kirkegaard et al 1995, Simpfendorfer 2002). The surprising results of Chan et al (1987) and later Kirkegaard et al (1995), showing that soil fumigation could overcome the early growth reductions pointed to the role of soil biological constraints. A subsequent investigation at 39 farm sites over three years in southern NSW (Simpfendorfer et al 2002) showed that the problem was widespread (62% of sites), was not related to any of the major soil-borne cereal disease organisms, nor to general changes in soil biology, but was strongly related to the inhibitory activity of *Pseudomonas* bacteria isolated from the rhizosphere of wheat seedlings at each site. The reduced early vigour at the Harden site reduced yield by 11% in wheat over that period and increased the amount of residual water and nitrogen left in the subsoil (Kirkegaard et al 2001).

During the course of our research to understand what reduced growth in direct-drilled wheat, the Harden site was used by many other researchers investigating the impact of management on other aspects of the soil biology. The widely promoted benefits to soil 'health' under direct-drill systems were evident there. Increases in soil organic matter (Pankhurst et al 2002), microbial biomass (Gupta et al 1994), populations of earthworms (Doube et al 1994), nematode diversity (Hodda et al 1997), faunal diversity (Longstaff et al 1997) and disease suppression (Pankhurst et al 2002) were all seen on

direct-drill/stubble retained treatment compared with late burn/single tine cultivation treatment, the most widely practised management in the area.

The dilemma for researchers and advisers highlighted by this example is that in spite of the apparent improvements in soil 'health', the growth and productivity of crops throughout the 15 year period were lower on the direct-drill/stubble retained treatment, and this was associated with more water and mineral nitrogen left in the subsoil, representing an increased risk of deep drainage/nitrogen leaching. This scenario may not be true for all farming systems, but it highlights the need to be pragmatic about the benefits of management to preserve various components of the soil biology for their own sake, and to keep in mind the other important aspects of the production system.

### **Plant root systems: an important part of soil biology**

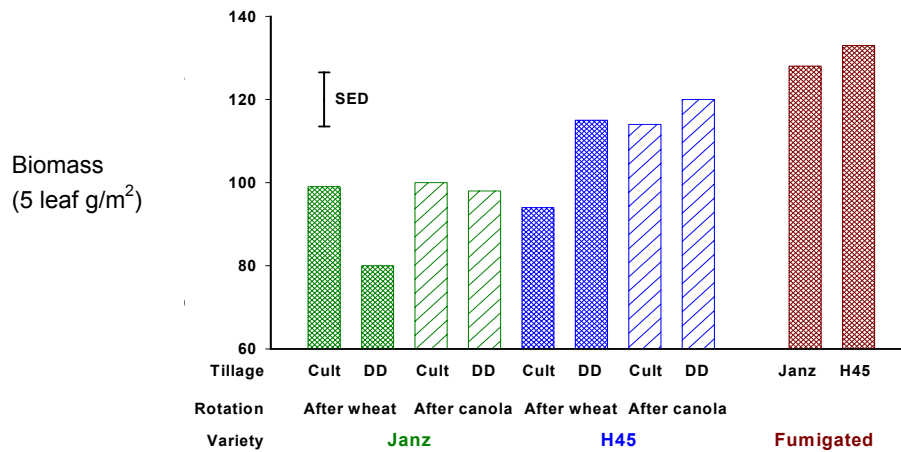
The most likely mechanism underlying the build-up of inhibitory bacteria on direct-drilled crops occurs in the rhizosphere, and is related to an interaction between soil hardness and an accumulation of pseudomonads on constrained, slow-growing roots (Watt et al 2003). Direct-drilled wheat roots grow at half the rate of roots in cultivated soil, contort around soil structure twice as often and have much shorter growth zones around the tips (Watt et al 2001 and unpublished). *Pseudomonas* bacteria accumulate on these slow-growing tips while the general bacterial population do not, likely related to growth and patterns of compounds released from roots (Watt et al 2003). If slow-growing roots are indeed more prone to being colonised by inhibitory pseudomonads, it could explain why management strategies such as early sowing into warmer soils and cultivation below the seed, both of which increase seedling root growth rates, may reduce the impact of direct-drilling on early growth. It could also mean that selecting wheat varieties with inherently fast rates of root growth may help reduce the impacts of hard soil on crop growth.

There are other examples of the ways in which plant roots can generate specific rhizosphere populations (excluding disease) which can have significant impacts on crop production. A recent example is that reported by Gupta et al (2003) in which specific populations of bacteria associated with some wheat varieties can persist in soil and influence the growth of subsequent wheat crops. Understanding the basis for these effects may provide opportunities to select or develop crop varieties that possess beneficial characteristics, and to exploit important interactions by matching the right variety to appropriate management. We recently investigated the interactions between tillage, rotation and wheat varieties in a field experiment in southern NSW, using fumigation as a tool to investigate the role of soil biology in the observed responses (Figure 2). The yield was drought affected, but the early biomass data demonstrated significant interactions between variety, tillage and rotation in the unfumigated treatments (on the left).

Janz biomass was reduced under direct-drilling following wheat but not following canola, while H45 had a different response to these management practices. Under fumigation there was no management or variety effects observed (only means for each variety are shown) indicating the role of the soil biology in the responses in the unfumigated soil.

The results suggest there is considerable potential to exploit the interactions between variety and management within cropping systems although an understanding of the mechanisms involved will be necessary to provide a basis for varietal selections. In the case of direct-drilling it may be the rate of early root growth which benefits some varieties, while different root exudates may influence the bacterial populations in the rhizosphere of different wheat varieties. In addition, the results indicate that biological constraints may still exist for some varieties even with good rotation and tillage management (eg cf Janz/ DD/Canola with Janz fumigate). Fumigation eliminated these interactions (overall means shown on right).

**Figure 2. Interactions between tillage, rotation and variety on wheat biomass (left).**



### Organic farming

Organic/biodynamic farming places particular emphasis on creating a diverse and vigorous soil biological community. It is often assumed that the soil biology on organic farms will adapt to the elimination of soluble fertilisers and other inputs to play a larger role in plant nutrition and growth. In some of the most comprehensive comparisons of organic/biodynamic with conventional systems on both grain and dairy farms on acidic loams in south-eastern Australia, it was phosphorus deficiency which most limited the development of an effective soil biological community (Ryan 2003). There was no evidence that biodynamic preparations enhanced the biomass or functions of the soil biological community, and it was wrong to assume that inputs permitted on organic farms are friendly towards the soil biota. Furthermore, there was no evidence that organic management had a consistent positive benefit for grain nutritive value. While there may be economic, philosophical or personal reasons for adopting organic, biodynamic or similar farming systems, there is no clear evidence that a more effective soil biological community results, and on the generally phosphorus-deficient acidic loams, a real risk that overall system productivity and sustainability will be limited if phosphorus inputs are not matched with outputs.

### Conclusions

Management practices (including varietal selection) can substantially affect the sustainability and profitability of farming systems via interactions with important components of the soil biology. Recent research offers promise of further improvements in this regard. The vague term ‘soil health’ does not relate well to either the broader environmental performance or the productivity of farming systems. The capacity of systems to achieve and sustain well-established physiologically-based productivity potentials remain the most useful benchmarks.

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## **Soil biology and crop production in Western Australian farming systems**

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### **Introduction**

Agricultural management practices ultimately seek to optimise plant and animal productivity within the overriding constraints of both climate and the capacity of the soil (physical, chemical and biological attributes) to support plant growth (Abbott, Murphy 2003). While optimal physical and chemical conditions of the soil for plant growth are often well defined, we have a much poorer understanding of the control that biological factors, particularly non-pathogenic associations, have on plant growth. The objective of this paper is to examine the relative contribution of soil biological attributes to crop production in Western Australian farming systems. Once these key attributes have been identified, management practices can be selected that take into account the potential for enhanced soil biological fertility and improved yield.

### **Western Australian farming system**

The grain production zone (wheat belt) in Western Australia covers an area of more than seven million ha. Grain production is primarily restricted to areas where average annual rainfall is between 325 and 750 mm, the majority of which falls during the growing season (late autumn-late spring) in the south-west of Australia. Major soils in this region (Chromosols, Sodosols, Kandosols) are highly weathered with low surface clay and soil organic matter contents. The summer weather pattern is typified by hot dry conditions with infrequent storm events, largely restricting production to an annual winter cropping phase. Low winter rainfall and dry summers therefore constitute the primary constraint to organic matter production and accumulation. A lack of new plant residues and root exudates to provide a carbon food source in the soil, and problems associated with desiccation over summer as surface soil temperature peaks above 40°C, present significant challenges to the buildup of biological components in soil compared with temperate environments. However, this does not mean that soil biology is not important. Indeed, the Western Australian farming system is reliant on a cyclic pattern of biological activity which 'explodes into action' with rainfall and then slows at the onset of soil drying.

The relatively low growing-season rainfall and the inherently low capacity of major soil types in WA to retain water and plant nutrients are realised in poorer crop growth. Low potential yields have thus resulted in relatively low input systems, and these systems are therefore more reliant on biologically fixed nitrogen and organic matter decomposition to supply plant available nutrients and support crop production. In southern Australia for example, Angus (2001) calculated that, on average, 80% of crop uptake was supplied via biological processes, so the amount of nitrogen cycling through a WA soil during the growing season can be more than enough to satisfy crop nitrogen demand (43-122 kg/N ha, Murphy et al 1998), even where no fertiliser is applied. The exceptions to this are soils with a high leaching potential, which can result in the loss of both water and mobile nutrients below the rooting zone, and soils where microbial immobilisation of nitrogen out-competes plants for nitrogen availability (eg decomposing plant residues with high carbon:nitrogen ratio). Strategically timed or split

fertiliser applications (generally 20-80 kg N/ha) are therefore used to overcome the difficulties of matching biological nutrient supply with plant demand. Developing management strategies to improve asynchrony (microbial nutrient supply occurring when plant demand is low) and synlocation (plant-available nutrients being located in the soil matrix where there are no plant roots) is often difficult but essential for future sustainable production (Murphy et al 2004, Ridley et al 2004, Hoyle, Murphy this proceedings).

### Identifying soil constraints to crop production

From 1960 to 1990, the average wheat grain yield in 62 WA shires was 1.9 t/ha, with less than 5% of shires assessed in 1990 having reached 50% of their rainfall-limited yield potential (Hoyle, Anderson 1993). In our current research we have used the WA-Wheat model (Department of Agriculture), which has been developed as a front-end system for the APSIM model, to target districts that consistently under-perform. To do this, WA-Wheat was used to initialise (seeding date, varietal maturity, fertiliser application, actual rainfall, soil type) model simulations (1960-2001) on a shire basis for comparison against actual historical yields. Where potential yield is not achieved our approach has been to assume that this is the result of inappropriate management practices and/or soil physical, chemical or biological constraints to crop production (Figure 1).

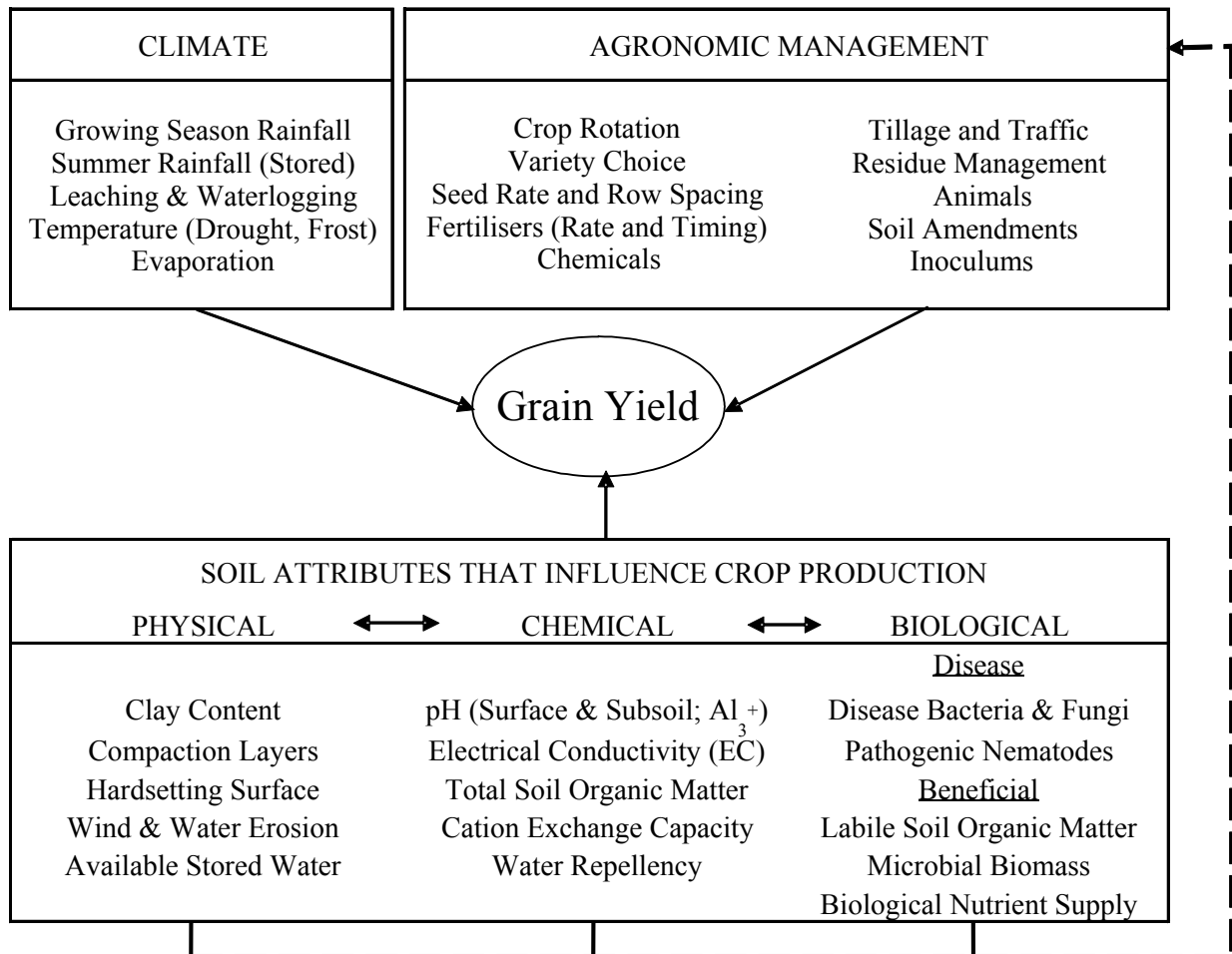


Figure 1. A conceptual model of climatic and agronomic factors along with key soil physical, chemical and biological constraints to yield production in Western Australian farming systems.

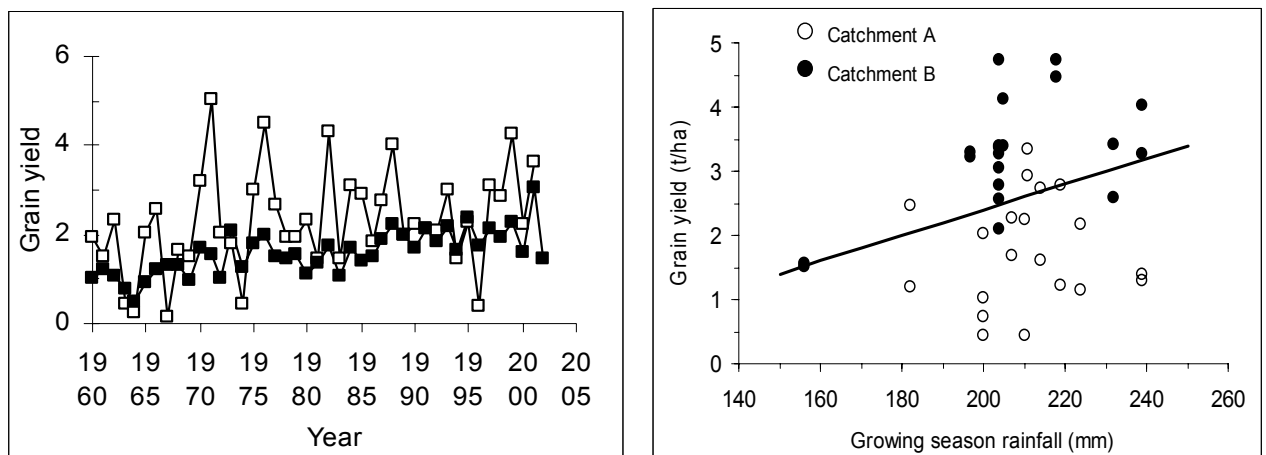


Once soil constraints are identified their economic importance can be assessed, so that the cost and practicality of removing the constraint versus potential yield benefit is known before implementing changes in agronomic practice. This approach focuses on discrete soil attributes that have a known direct impact on crop production, and can be measured and interpreted in the context of management solutions. This approach provides an economic evaluation of 'cause' and 'effect', enabling prioritisation of high return solutions to overcome major agronomic and soil limitations instead of placing effort in further detailed site characterisation which is not feasible over a large scale.

### Identifying soil constraints to crop production: a case study

Evaluation of the 'soil indicator' package described in Figure 1 was achieved by collecting climatic, agronomic and soil data from 40 paddocks on 20 farms in two adjoining catchment groups (named 'A' and 'B' for simplicity). Paddocks were located within a 10 x 20 km region and were chosen in consultation with growers to either compare high and low yielding areas, or encompass soils that consistently under/over performed against expected yields. Within each paddock three field replicates were established, and within each replicate area soil was collected in 0-5, 5-10, 10-30, 30-60 and 60-90 cm layers for laboratory analysis (in triplicate). Rainfall was recorded at each farm and agronomic data supplied through a one-on-one interview and questionnaire process with the principal grower in each farming unit. Grain yield cuts were taken by hand within a few days prior to machine harvest.

Using figures from the shire that includes A and B catchments, we compared the WA-Wheat model's predicted achievable grain yield against historical records (1960-2001) of actual average grain yield (Figure 2). In approximately 50% of years, we observed good agreement between actual and predicted yield, but in 20 of the 43 years there was a difference of greater than 0.8 t/ha in predicted yield compared with actual yield. Given the low average historical grain yield for wheat in this region (1.58 t/ha), this would represent a significant yield benefit if obtainable. Actual yield data from the 40 paddocks illustrate that on a site by site basis actual yield can vary considerably (mean = 2.5 t/ha, min = 0.44 t/ha, max = 4.74 t/ha) within a season (Figure 2) and can reach the same upper range as predicted by the model.



**Figure 2. Left: Actual (filled squares) and modelled (open squares) grain yield (t/ha) for the shire that contains catchments A and B. Right: Measured grain yield from the 40 paddocks plotted against growing season rainfall for each site. The solid line represents an achievable grain yield. Paddocks below this line are underperforming and those above the line are above reasonable expectation.**

The independent influence of rainfall, inorganic nitrogen fertiliser and soil constraints (as listed in Figure 1) on grain yield was determined using bivariate regression analysis (Table 1). In this regression analysis data for diseases (take-all, rhizoctonia) and pathogenic nematodes (*Pratylenchus neglectus*, *P. thornei*) were excluded as their occurrence was below detection limits or low in 38/40 paddocks. Biological nutrient supply was assessed solely as potentially mineralisable nitrogen in the regression analysis. Mycorrhizal bioassays were performed to determine their importance to plant nutrient supply. More than 30% of root length colonisation is required to obtain benefits of plant nutrient acquisition from mycorrhizal associations (Abbott, unpublished critical value). However, mycorrhizal root length colonisation in the plant bioassays performed was between 0-30% as the paddocks were sufficient in bicarbonate-extractable phosphorus.

**Table 1. Mean values for attributes determined in catchments A and B and results of bivariate regression analysis whereby climatic, agronomic and soil physical, chemical and biological attributes were assessed for their individual influence on wheat grain production across the 40 paddocks. Average grain yield was 1.76 and 3.24 t ha/ in catchments A and B respectively. All significant attributes have been presented; most non-significant attributes assessed have been removed. (The same letter denotes no significant difference between catchments for that attribute.)**

	Attribute	Catchment		Coefficient <sup>a</sup>	P-value <sup>b</sup>	Variability Explained <sup>c</sup>
		A	B			
Climate	Rainfall (mm)	211a	206a	-	ns	3.7
Agronomy	N fertiliser (kg N/ha)	20a	24a	0.02	0.055*	9.4
Physical	Clay content <sup>d</sup> (%)	11.0a	10.4a	0.08	0.062*	9.1
Chemical	Total carbon (t C/ha)	9.0a	10.8b	-	ns	0.2
	pH (CaCl <sub>2</sub> )	5.7a	5.6a	-	ns	0.4
	EC <sup>d</sup> (mS/m)	80a	63b	-	ns	0.1
Biological	Labile C (kg C/ha)	83a	118b	0.01	0.041**	10.5
	Microbial biomass C (kg C/ha)	107a	183b	0.01	0.001***	30.3
	PMN (kg N/ha)	7.0a	10.1b	0.14	0.003***	21.2

<sup>a</sup>The coefficient can be interpreted as t/ha grain yield change per unit change in attribute.

<sup>b</sup>\* = significant P<0.10; \*\* = significant P<0.05; \*\*\* = significant P<0.01; ns = not significant.

<sup>c</sup>The variability explained has a maximum of 100% and is not additive between individual attributes.

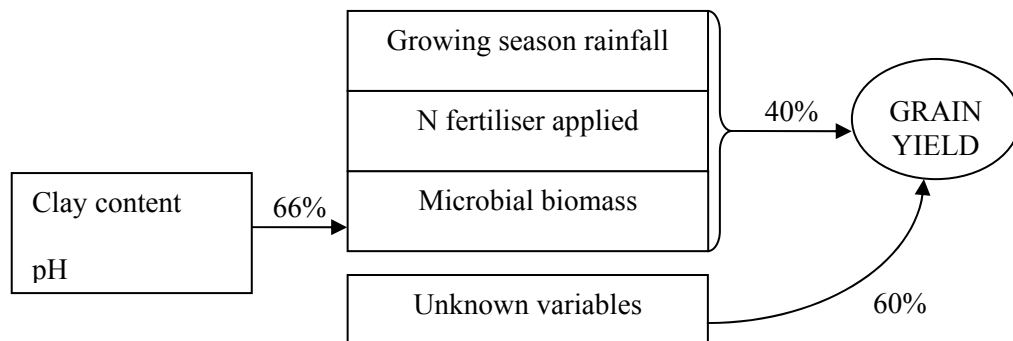
<sup>d</sup>clay and EC data were assessed using robust regression analysis due to unusual data points. EC = Electrical conductivity. .

Measured yields from catchment B were significantly higher than in catchment A, which is reflected in some, but not all of the soil attributes used in the regression analysis (Table 1). It is notable that the biological attributes explained the greatest amount of variability in yield between the 40 paddocks. For example nitrogen fertiliser and clay content each explained 9% of the variability. Potentially mineralisable nitrogen, an index of biological nitrogen supply, explained 21%. Microbial biomass explained 30% (Table 1). Growing season rainfall was not significantly related to grain yield, although we have already argued that rainfall is the primary driver of production in this environment. However, this was not surprising as we would only expect a strong relationship between growing season rainfall and yield if there were no other constraints to crop production. Over a 10 mm growing season rainfall gradient (200 to 210 mm), there was a grain yield variation from 0.5 to nearly 5.0 t/ha (Figure 2). Thus there was

certainly either poor agronomic management and/or the influence of soil constraints on crop production.

Combinations of significant factors that influenced grain yield were then determined using ordinary least square multiple regression analysis. Using a multiple regression model that included all nine parameters listed in Table 2 we were able to explain 42% of the yield variability (regression model not shown).

Several soil attributes were identified that did not have a significant direct influence on grain yield; but may have had an indirect influence through their effect on the size of the microbial biomass (Figure 3). In this case, 66% of the variability in microbial biomass could be explained by clay content (log transformed data), pH and labile carbon. In other words, providing an optimal physical and chemical soil matrix along with an available carbon (food) source was the primary basis for improving the mass of soil microorganisms in these soils. This is logical given microorganisms, like all other living organisms, function more effectively within an optimal environment and provided with a suitable food source. Removing attributes that were either directly related to microbial biomass, or those that were not significantly affecting grain yield from the initial model, resulted in the development of a simpler model to explain the variability in grain yield (Figure 3). This model, which consisted of growing season rainfall, nitrogen fertiliser and microbial biomass as the only three attributes used, still explained 40% of the variability in grain yield. This means that by removing six attributes from the initial model we only lost 2% of explained variability; but removed a considerable amount of the analytical measurements that would be required.

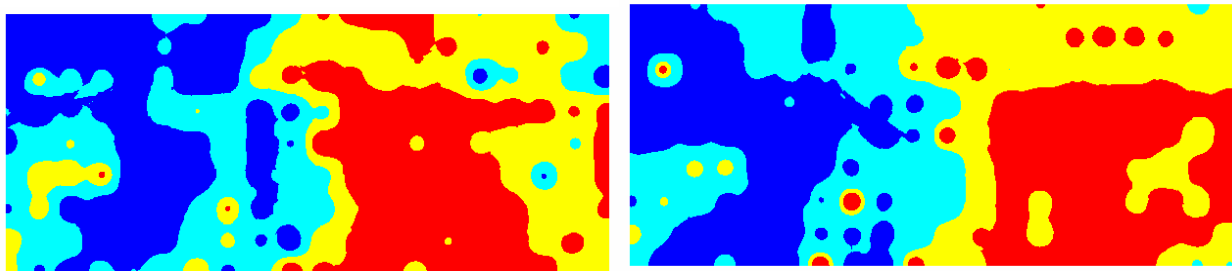


**Figure 3. A schematic representation of the multiple regression analysis models used to describe microbial biomass (explanatory variables:  $\ln(\text{Clay content}) \cdot \text{pH} \cdot \text{Labile Carbon}$ ) and grain yield (explanatory variables:  $\text{Growing season rainfall} \cdot \text{N fertiliser applied} \cdot \text{Microbial Biomass}$ ).**

Further analysis indicated that the influence of the microbial biomass on yield was predominately due to the strong relationship ( $r^2 = 0.77$ ) to potentially mineralisable nitrogen. Thus the model used to describe grain yield could alternatively be expressed as growing season rainfall, nitrogen fertiliser and potentially mineralisable nitrogen with a similar percentage of the grain yield still being explained (data not shown). This provides a simple water and nitrogen availability story as the key drivers of grain production in this environment, which is supported by the fact that water is essential for plant growth and that nitrogen is the primary nutrient limiting crop production throughout the world.

## Rapid prediction of potentially mineralisable nitrogen using mid infrared technology

Our current research has demonstrated that potentially mineralisable nitrogen (PMN) can be successfully predicted using mid infrared technology (Murphy et al 2004, Murphy, Milton this proceedings). The major advantage of mid infrared prediction over conventional laboratory analysis of PMN is that it enables rapid (two minute) and cost efficient analysis of a soil biological attribute that has a direct impact on yield production. For example, a one unit increase in PMN caused a 0.14 t/ha yield increase, Table 1. The accuracy of mid infrared to predict the within-paddock variability in PMN is illustrated in Figure 4.



**Figure 4. Spatial maps (10 ha) of potentially mineralisable nitrogen determined analytically using traditional biochemical analysis (left) and predicted using mid infrared technology (right) for the 0-10 cm layer of a Western Australian agricultural soil. Data categorised into four categories (Murphy et al 2004).**

Soil was collected using a 25m x 25m sampling grid (180 sampling points over 10 ha) from one of the 40 paddocks. Over this 10 ha area PMN ranged from 4-32 kg N/ha. PMN was determined using conventional biochemical laboratory analysis and also predicted (on the same soil samples) using a mid infrared calibration curve that was developed from an independent data set. There was good agreement between mid infrared predicted and measured PMN ( $r^2 = 0.70$ ) which is illustrated by the degree of similarity in the measured and mid infrared predicted spatial maps (Figure 4). While mid infrared is not 100% accurate at predicting PMN, it is of sufficient accuracy for categorising soils or zones within a paddock into poor, low, moderate and high biological soil nitrogen supply, which could be used to adjust for inorganic nitrogen fertiliser application rates.

## Management options to enhance soil biological fertility

Despite the identification of known soil constraints to grain yield, 60% of the variability in wheat grain yield is still not explained within these catchments. This highlights the complexity of soil-plant-microbe interactions and the difficulty in identifying drivers of grain yield within different environments. However, the fact that biological attributes had a greater quantifiable influence than chemical or physical attributes on yield variability in this case study provides justification to the development of agricultural farming systems that encourage soil biological fertility (Abbott, Murphy 2003). However, there are few, if any, quick fix solutions to improving soil biological fertility. Research trial data from WA (Table 2) demonstrates that it can take many years for differences in attributes of soil biological fertility to occur upon implementation of management practices. Soil biological attributes are generally highly variable spatially over small distances (see Case study 4, Table 2), with changes in the chemical and physical attributes of the soil often having a greater influence than imposed agronomic management practices on soil biological fertility. Therefore, it is difficult to measure

significant differences between treatments even when changes seem quite large (eg Case study 3 microbial biomass, Table 2).

**Table 2. Impact of agronomic management practices on microbial biomass, biological soil nitrogen supply (PMN) and diversity (catabolic diversity, range possible 0-24 with higher number indicating more diverse population) of microorganisms in four trials from WA that represent the major soil groups (Chromosols, Sodosols, Kandosols). Values with the same letter are not significantly different ( $P < 0.10$ ) within the same trial for the biological attribute specified.**

Case study #	Agronomic management	Microbial biomass	PMN	Catabolic diversity
		kg C/ha	kg N/ha	min = 1 max = 24
1	Harvest stubble burnt	98a	No data	14.5a
	Harvest stubble retained	153b	No data	15.5b
2	Continuous wheat rotation	308a	30a	15.8a
	Faba beans:Wheat:Canola:Wheat	317a	30a	16.4a
	Medic (grazed) : Wheat	421b	25a	18.0b
	Annual pasture - Ryegrass (grazed)	417b	45ab	16.5a
	Perennial pasture - Lucerne (grazed)	421b	67b	16.5a
3	Lupin - brown manure	140a	13a	15.9a
	Oat - brown manure	76a	14a	17.6b
	Mustard - brown manure	119a	15a	19.4c
4	Variability within 10 ha; n = 220 pts	22 to 1000	4 to 32	No data

1: Data collected after 17 years of imposed treatments, 0-5 cm, Chromosol, Merredin WA.

2. Data collect after 4 years of imposed treatments, 0-5 cm, Sodosol, Mindarabin WA.

3. Data collected after 2 months of imposed treatments, 0-10 cm, Kandosol, Meckering WA.

4. Minimum and maximum data from 220 composite bags of soil collected under a barley crop on a 25 m grid over 10 ha; 0-10 cm; Dangin WA.

Seasonal variability in the data collected is also a major issue in deciding when to sample soil for biological attributes. This is illustrated in Table 3 where it can be seen that the seasonal (sowing, tillering, flowering, harvest) differences in measured soil biological attributes are considerable.

**Table 3. Impact of season on the microbial biomass, potentially mineralisable nitrogen (PMN) and the actual daily rate of inorganic nitrogen release through microbial decomposition of soil organic matter and residues (gross nitrogen mineralisation). Six conventional farms were paired with two farms of each of the other farming systems listed. S = sowing, T = tillering, F = flowering, H = harvest.**

Farming system	Microbial biomass-N (kg N/ha)				PMN (kg N/ha)				Gross mineralisation (kg N/ha/day)			
	S	T	F	H	S	T	F	H	S	T	F	H
Conventional	52	42	20	12	42	34	40	36	7.1	6.6	1.2	1.0
Integrated	60	32	17	11	58	44	54	47	5.8	6.1	1.2	0.8
Organic	72	46	19	10	54	44	53	48	3.6	6.4	1.6	1.0
Bio-dynamic	72	37	26	11	54	46	58	54	5.3	5.5	1.3	1.1

However, it should be noted that PMN was more stable through the season than measurement of microbial biomass or microbial activity (gross nitrogen mineralisation), suggesting that it is an easier soil biological attribute to interpret between and within seasons. Data in Table 3 also illustrates that seasonal changes in the measured biological attributes are greater than measured differences between farming system type. Thus the capacity to alter soil biological fertility within a region is primarily constrained by water and temperature with agronomic practice as a secondary factor.

## **Conclusion**

Soil biological fertility was significantly correlated to grain production in WA. The benefit was predominately associated with the size of the microbial biomass, which was directly related to their capacity to decompose soil organic matter and fresh residues to release plant available nitrogen. These findings confirm our view that WA farming systems are highly reliant on biological nitrogen supply and that farming systems need to be modified where possible to fully benefit from water availability and microbial nutrient supply. However to achieve this, limitations associated with both the asynchrony and synlocation of water and nutrients need to be further addressed. This will require improved soil management to identify and remove soil constraints to plant growth and rooting depth, new plant breeding to improve plant root architecture in order to capture water and nutrients, a flexible fertiliser strategy (type, split applications, delivery), developing an economic role for deep rooted plants and improved plant residue management (carbon:nitrogen ratio of decomposing material, level and timing of incorporation) and identifying novel methods for manipulating microbial processes.

## **Acknowledgements**

This research is funded by the Grains Research and Development Corporation with support from The University of Western Australia. We also thank growers from the catchment groups for their participation in this research. This paper was reviewed by Matthew Braimbridge and Tamara Flavel.

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## Impact of fertilisers on soil biota

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University of Adelaide

### Introduction

#### **Fertilisers**

Fertilisers are an integral component of agricultural production systems and are applied with the main goal of maximising yields and economic returns. An underlying principle governing fertiliser application rate should be the replacement of nutrients removed in the harvested product. Fertilisers include mineral products such as urea, ammonium nitrate, sulfates and phosphates, and organic fertilisers such as animal manures and biosolids. Since most organic fertilisers are waste products, their application rate is often determined by availability rather than demand. Currently, the use of mineral fertilisers in broadacre Australian agriculture outweighs the use of organic fertilisers.

#### **Soil biota**

While the application of fertilisers usually aims at meeting plant demands, it can also have an impact on soil biota. Soil biota consist of the microflora (bacteria and fungi) and the soil fauna (protozoa and invertebrate groups such as nematodes, mites and earthworms). They influence the availability of nutrients for crop production via a range of activities such as the decomposition of crop residues, immobilisation (microbial uptake) of nutrients, mineralisation (transformation of organic nutrients into plant available inorganic forms), biological nitrogen fixation, and bioturbation. The soil fauna are crucial for the initial comminution of residues and mixing into the soil, while the microflora have a greater suite of enzymes for chemical breakdown of organic material (Paul, Clark 1996). Bacteria and fungi are often considered as a labile pool of nutrients (carbon, nitrogen, phosphorus and sulfur) called the soil microbial biomass that has a pivotal role in nutrient immobilisation and mineralisation. The release of nutrients from the microbial biomass is partly regulated through grazing by the soil fauna.

#### **Methods to determine fertiliser effect on soil biota**

The effect of fertilisers on soil biota can be measured either as changes in the amount of single organisms, organism groups or methodologically defined pools such as the microbial biomass, or as changes in biological activity such as soil respiration, enzyme activities and root colonisation rates. Variable effects of fertilisers on different organisms may change the composition of the microbial (or faunal) community without changing total amounts or activities. However, the majority of studies have focussed on the soil microbial biomass as the central pool in nutrient cycling.

#### **Approach of this review**

In this paper we summarise the current understanding of fertiliser effects on soil biota, based on the concept that fertilisers can affect soil biota through direct or indirect effects (Table 1). Direct effects via changes in nutrient availability are likely to become obvious in the first season after their application, or in the longer term if repeated additions are required to reach a threshold above which effects are seen. Indirect effects will usually take more than one season to establish, especially when changes in soil organic matter levels are involved. In the case of long-term data, it can be difficult to separate direct and indirect effects. The evidence from Australia is rather limited, and therefore the review includes literature from overseas, in an attempt to establish the



main principles of fertiliser effects on soil biota and to draw some conclusions applicable to agro-ecosystems in Australia.

**Table 1: Potential effects of fertilisers on soil biota**

<b>Direct effects</b>	<b>Time frame</b>
Increased amount and/or activity after removal of nutrient limitations	Short <sup>1</sup> - to long-term <sup>2</sup>
Decreased activity due to high nutrient availability	
Decreased amount and/or activity due to toxicity	
<b>Indirect effects</b>	
Change in pH	Long-term
Change in productivity, residue inputs and soil organic matter levels	

<sup>1</sup> One season

<sup>2</sup> More than one season

## **Direct effects of mineral fertilisation**

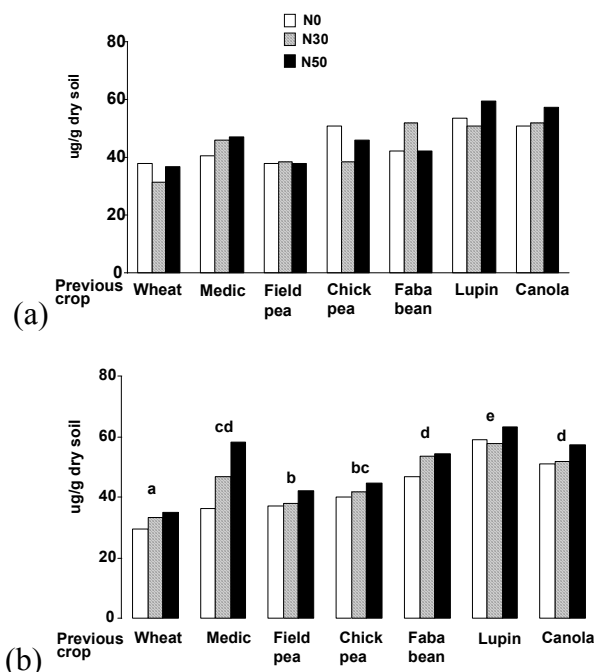
### ***Short-term stimulation of soil biota due to mineral nitrogen or phosphorus fertilisers***

Increased amounts or activity of soil biota as a direct result of mineral fertilisation have been reported in several studies. For example, Sarathchandra et al (2001) observed a short-term increase in gram-negative bacteria and rock-phosphate-dissolving bacteria two weeks after application of superphosphate or rock phosphate to a permanent pasture in New Zealand. Application of ammonium nitrate to pasture in the UK led to a short-term increase in the microbial processes of nitrification and ammonification, although the size of microbial biomass (carbon and nitrogen) did not change (Lovell, Hatch 1997). Indeed, a 'priming effect' of nitrogen addition on soil respiration is often observed (Kuzakov et al 2000), and there is recent evidence that nitrogen addition might specifically stimulate the decomposition of older, stabilised soil organic matter (Waldrop, Firestone 2004).

### ***No short-term effect or depression of soil biota due to mineral nitrogen fertilisers***

While such findings suggest a positive impact of mineral fertilisers on soil biota, probably via the removal of nutrient limitations, many more studies report no change or even a decrease in amounts or activity of soil biota following mineral fertiliser application. For example, a study in Western Australia (McNeill et al 2000) found little influence of fertiliser nitrogen, applied to wheat at sowing, on microbial nitrogen measured at tillering or anthesis (Figure 1a,b), compared with the effect of the previous crop on the microbial biomass (Figure 1b). Likewise, the addition of 120 kg N/ha as ammonium nitrate to a potato crop in the UK had no effect on soil respiration, dehydrogenase activity or microbial carbon and nitrogen throughout the growing season (Ritz et al 1992), although the addition of sucrose (as an energy/carbon source) in conjunction with nitrogen did increase all of these parameters. Nitrogen fertilisation did not affect the size and activity of the microbial biomass under permanent pasture, regardless of whether or not the soil had received nitrogen fertilisation for the previous 15 years (Hatch et al 2000). Similarly, addition of ammonium sulfate to soils under pasture in NZ had little effect on microbial use of added energy (glucose), although it appeared to actually decrease amounts of microbial phosphorus (Saggar et al 2000).

**Figure 1: Microbial N (ug/g) in 0-15 cm soil depth (Goomalling WA 1998) under a wheat crop after application of urea (0, 30 or 50 kg N/ha at sowing) and following different break crops (wheat, medic, field pea, chickpea, faba bean, lupin or canola): (a) at tillering in August and (b) at peak biomass of the crop in October. Different letters atop bars indicate significant differences between previous crop across N rates ( $p < 0.001$ ).**



### ***No short-term effect or depression of soil biota due to mineral phosphorus or sulfur fertilisers***

There is even less information available on the direct effects of mineral phosphorus and sulfur fertilisers on soil biota than there is for nitrogen fertilisers. A Canadian study did not find any immediate nor residual effect of sulfur fertilisers on the microbial biomass under a wheat-canola rotation, but observed a slight decrease in diversity as assessed by the utilisation of carbon substrates (Lupwayi et al 2001). A reduction in specific organisms such as arbuscular mycorrhizal (AM) fungi by phosphorus fertilisation appears to be fairly well established. In a comparison of Australian pastures under conventional and biodynamic management, Ryan et al (2000) noted a negative relationship between available phosphorus and colonisation rates of clover roots with AM fungi. However, AM colonisation rates of ryegrass were not affected by phosphorus addition (Ryan, Ash 1999). These findings agree with the variable effect of nitrogen, phosphorus, potassium (NPK) fertilisation on percent root colonisation by AM in different grassland species observed by Rillig et al (1998), suggesting that in the case of mycorrhizal symbioses, indirect effects through changes in plant growth and metabolism rather than direct effects on the fungi might prevail.

### ***Short-term toxicity effects of mineral fertilisers***

A decreased amount or activity of soil biota after mineral fertilisation could be due to the toxicity of metal contaminants contained in mineral fertilisers. In general, nitrogen and potassium fertilisers contain very low levels of contaminants, whereas phosphorus fertilisers often contain significant amounts of cadmium, mercury and lead (McLaughlin et al 2000). Metal contaminants are most prevalent in waste products from urban and industrial areas. Long-term chronic toxicity due to gradually accumulating metals appears to be far more common than immediate, acute toxicity (Giller et al 1998).

Therefore, toxicity effects on soil organisms will be discussed in the section on long-term effects.

## **Direct effects of organic fertilisation**

### ***Short-term stimulation of soil biota due to organic fertilisers***

The majority of studies report a positive direct effect of organic fertilisers on soil biota. For example, the application of farmyard or poultry manure to barley in the UK increased microbial nitrogen immediately and microbial carbon in the second year of the trial (Ritz et al 1997). Microbial carbon and gross nitrogen immobilisation were also increased under dung pats applied to pasture in the UK, which was attributed to a temporary increase in available soil carbon (Hatch et al 2000). In another study, the microbial biomass as well as soil respiration clearly increased after mixing of dung into the soil, but not under dung pads in the field (Lovell, Hatch 1997). Likewise, no change in microbial phosphorus under dung pads was observed in Victoria, except for an increase at the last sampling after 60 days (Aarons et al 2004).

### ***Role of carbon inputs with organic fertilisers***

The relatively greater influence of organic versus inorganic fertilisers was emphasised in a study where three months after application of 200 kg N/ha as ammonium nitrate to maize, soil respiration, acid phosphatase and dehydrogenase activity were higher than in the non-fertilised control. However, all these changes were even more pronounced when the same amount of nitrogen was added as dairy manure or composted sewage sludge (Marinari et al 2000). These differences are certainly related to carbon inputs with organic fertilisers. In contrast to plants, soil organisms are heterotrophic and need carbon as the most essential nutrient. A large fraction of soil organic carbon is poorly available to soil organisms. Additions of labile carbon have therefore also been observed to induce 'priming effects' on the decomposition of soil organic matter (Kuzyakov et al 2000).

## **Indirect effects of mineral and organic fertilisers**

### ***Absence of long-term effects of mineral phosphorus fertilisers***

Preliminary data from two trials in Australia indicate the absence of phosphorus fertiliser effects on microbial phosphorus in spite of significantly increased phosphorus availability after many years of phosphorus addition (Table 2). This agrees with findings from pastures in NZ, where no significant long-term effects of phosphorus additions on microbial phosphorus or earthworm abundance were noted (Sarathchandra et al 1993). The same was observed during six years of phosphorus fertiliser inputs to pastures in Victoria (Aarons 2001).

### ***Change in pH – a long-term effect of fertilisation***

While phosphorus fertilisation often does not have any effect on soil biota, mineral nitrogen fertilisation sometimes has a negative effect. For example, repeated additions of ammonium nitrate to various wheat rotations in South Australia decreased microbial carbon as well as pH (Ladd et al 1994). In a long-term trial in Sweden, Witter et al (1993) identified a change in soil pH due to repeated addition of ammonium sulfate as an indirect effect of fertilisation on soil biota. Urea application also decreased soil organic carbon, microbial carbon and pH under permanent pasture, while superphosphate applications affected none of these parameters (Sarathchandra et al 2001). While a decrease in pH appears to be the primary explanation for negative effects of mineral fertilisers on soil biota, it does not always correspond with a decrease in microbial biomass (Belay et al 2002, Moore et al 2000, Peacock et al 2001).

**Table 2: Effect of mineral phosphorus fertilisation on available and microbial phosphorus (Bünemann et al, unpublished)**

Site	Land use	P inputs	Available P <sup>1</sup>	Microbial P <sup>2</sup>	Organic C <sup>3</sup>
		kg/ha/yr	mg P/kg soil	mg P/kg soil	g/kg
Walpeup, Vic	Wheat-fallow rotation <sup>4</sup>	0	4.4 ± 0.2	1.0 ± 0.1	0.63 ± 0.07
		135	18.0 ± 0.8	1.5 ± 0.8	0.60 ± 0.07
Otterbourne, NSW	Permanent pasture	0	1.4 ± 0.1	3.9 ± 0.2	1.99 ± 0.11
		125	4.0 ± 0.1	4.4 ± 0.7	2.18 ± 0.08
		250	9.1 ± 0.1	3.0 ± 0.9	2.17 ± 0.02

1 P extractable with anion-exchange resin membranes

2 P-flush after hexanol fumigation

3 Organic C predicted by mid-infrared spectroscopy (MIR)

4 Permanent phosphate trial (McClelland 1968)

### ***Change in soil organic matter - a long-term effect of fertilisation***

Graham et al (2002) investigated the amounts of microbial carbon and nitrogen under sugarcane after 59 years of differential crop residue management and NPK fertilisation, showing that the microbial biomass was directly influenced by residue management and indirectly by NPK fertilisation through increased residue inputs. In fact, many long-term experiments with different forms and rates of fertilisation have rendered strikingly good relationships between the size of the microbial biomass and soil organic matter contents (Houot, Chaussod 1995, Leita et al 1999, Moore et al 2000, Witter et al 1993). Usually, the treatments with mineral fertilisation are at the lower end of the correlation, as they increase soil organic and thus microbial carbon to a lesser degree than organic amendments such as farmyard manure and compost (Leita et al 1999). The example provided by Houot and Chaussod (1995) is especially interesting as it shows how the excellent correlation found after more than one hundred years of constant management practices remained disturbed two years after a change in crop rotation and crop residue management. The time required to reach a new equilibrium is a factor that may confound the results from many short-term studies.

Nevertheless, most studies that show a positive effect of mineral or organic fertilisation on soil organisms also record an increase in soil organic matter. Indirect effects of fertilisers on soil biota through an increase in plant productivity and returned crop residues may also explain results from the Mallee region in South Australia. Gupta et al (2004) showed that after four years of a low input system with fertiliser rates of 10 kg P/ha and 5 kg N/ha, microbial biomass size and activity were lower than for a high input system with rates of 15 kg P/ha, 27 kg N/ha and 1.5% zinc (Table 3). On the other hand, the absence of phosphorus fertiliser effects on microbial phosphorus shown in Table 2 may be due to the fact that no significant changes in soil organic carbon content were observed.

### ***Long-term toxicity effects of fertilisers***

The accumulation of toxic metals in soil due to repeated fertiliser applications is of concern in the long-term. It has been shown that specific organisms such as nitrogen fixing rhizobia are far more sensitive to metal toxicity than clover as their host plant. This resulted in nitrogen deficiency of clover due to ineffective rhizobia in sludge-amended soils (Giller et al 1998). Such observations warrant strict regulations of fertiliser quality and applied quantity, especially of waste products such as sewage

sludge and biosolids, in order to minimise contamination of agricultural land with toxic metals.

**Table 3: Microbial biomass carbon (MB-C), nitrogen (MB-N), phosphorus (MB-P), and respiration as a measure of microbial activity in the surface soils (0-10 cm) of selected treatments at the Mallee Sustainable Farming Systems Waikerie core site after 4 years. DP is district practice and HI is high input as referred to in the text.**

Cropping system	MB-C(kg C/ha)	MB-N (kg N/ha)	MB-P (kg P/ha)	Respiration (g CO <sub>2</sub> /m <sup>2</sup> /hour)
Pasture-Wheat (DP)	265	26	16.0	0.105
Pasture-Wheat (HI)	370	43	21.0	0.185
Legume-Wheat (HI)	370	46	13.0	0.210
Canola-Wheat (HI)	357	36	16.5	0.175

From Gupta et al 2004.

## Conclusions

The availability of carbon substrates is more important for soil biota than that of nutrients such as nitrogen, phosphorus, potassium and sulphur. Therefore, organic fertilisers usually have greater impact on soil biota than mineral fertilisers. Direct effects of mineral fertilisers on soil biota seem to be variable but perhaps less important than indirect effects. The main indirect effects are a depression of soil biota due to a decrease in soil pH, and an increase in biological activity with increasing plant productivity, crop residue inputs and soil organic matter levels. As Australian soils are generally low in organic matter and nutrient contents, any increase in soil organic matter is desirable in view of the important role of soil biota in nutrient cycling.

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## **Impacts of pesticides on soil biota**

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### **Introduction**

There is significant uncertainty in the prediction of impacts of external stressors such as pesticide application on landscape effects. For example, a pesticide may be toxic to a soil organism under laboratory studies, but does field application result in long-term population changes, and if so what is the significance of these population changes? If an applied pesticide removes a particular species from a site, and the species returns to the site from neighbouring populations, has there been any harm to soil health?

Recent developments in aquatic ecotoxicology have combined ecotoxicological principles (eg direct toxicity measurements to individual species), with ecological knowledge to enhance the realism of risk assessment (Liess 2004). In other words, an understanding of the entire system is required to gain an accurate picture of potential impacts. Many studies often evaluate short-term effects on individual indicators, so cannot realistically predict the true risk of a chemical or management practice on soil biota.

The use of pesticides has long been a feature of conventional agricultural practice and their use has made it possible to increase crop yields and food production (Lee 1985). However, many of these chemicals have toxic effects that are not confined to their target species, and their application may have impacts on organisms that benefit the wider agroecosystem. The paradigm of Newton's Third Law, although referring to motion and energy, is quite applicable to the management of soils: 'To every action there is an equal and opposite reaction.' Every management decision a farmer implements will impact on the health of the soil, resulting in either a beneficial or detrimental outcome.

This paper discusses the potential impacts of several pesticides on soil biota and highlights implications for the management of these soils.

### **What are soil biota and how can impacts be measured?**

There are many excellent reviews on soil biota (see Pankhurst 1997, Rovira 1994) and their interactions in farming systems. Soils contain microorganisms including bacteria, fungi, yeasts; photosynthetic organisms including algae, and macroorganisms such as protozoa, nematodes, mites, springtails, spiders, insects and earthworms. The functions of this complex array of biota, often referred to as the 'soil food web,' are diverse, and include residue decomposition, nutrient storage and release, soil structure and stability, resistance against disease and degradation or immobilisation of pesticides and other pollutants. In the literature, soil biota are measured or studied in many different ways, including direct observation under microscopes and other direct counting mechanisms, analysis of intra- and extracellular enzymes, respiration and biomass carbon, fatty acid profiles, and DNA fingerprinting. Unlike the field of aquatic ecotoxicology where many standard tests are applied to measure impacts of stressors, there are no standard reporting mechanisms for measuring soil biota. This renders the direct comparison between different pesticides and their potential impacts impossible.



### **What processes can affect soil biota?**

Many factors can have negative impacts upon soil health. These factors include loss of organic carbon (Islam and Weil 2000), compaction (Singleton and Addison 1999), disruption of soil macroaggregates (Islam and Weil 2000), pesticides (Mitra and Raghu 1998) and pesticide breakdown products (Cernakova and Zemanovicova 1998), inorganic pollution arising through fertilisers, fungicides and sludge application (Merry et al 1986, Gong et al 1997), the use of fertilisers (Stamatiadis et al 1999) and non-pesticide organic pollution including surfactants (Wilke 1997). Other causes of reduced soil health can arise through water and wind erosion (Garcia et al 1997), grazing (Anon 2001), loss of organic matter due to fire, deforestation and tillage (Islam and Weil 2000).

### **How can pesticides influence soil biota?**

Pesticides can impact on soils through direct or indirect mechanisms. For example, the use of herbicide may not directly influence soil biota, but the process of removing vegetative material from the soil can influence availability of rhizosphere exudates and organic material to biota, and can reduce soil stability, leading to erosion and compaction. Direct impacts of pesticides can occur when the chemical reaches the soil, either due to direct targeted deposition such as the case of soil fumigants, or pre-emergent herbicides, or through indirect deposition from spray and spray drift, dripping from plant material, and contaminated plant material falling to the soil.

There are several factors influencing the amount of pesticide reaching the soil, and potentially impacting on soil biota. During application, pesticide may degrade in the spray mix and/or undergo photodegradation (ie sunlight). The pesticide may drift off target and affect non-target sites. These processes reduce the amount of active chemical impinging on the soil surface.

On the soil, pesticides dissipate through several processes, including volatilisation, leaching and surface run-off. Degradation can occur through biological and non-biological processes. The biological availability of the pesticide may be reduced by its binding to components in soil, in particular, organic matter and clay.

### **Effects of selected pesticides on soil biota**

Table 1 summarises the effects of a few selected pesticides on soil biota. It can be observed that negative impacts range from negligible to very significant, while some pesticides even appear beneficial to soil biota!

Although the list of chemicals mentioned is not exhaustive, it covers a range of herbicides, fungicides, insecticides, veterinary care products and soil fumigants. Often data is not available on the effects of pesticides on soil biota, as registration requirements do not require extensive soil biota tests to be undertaken.

Table 1. Impacts of selected pesticides on soil biota

Active chemical	Effects	Reference
<b>Herbicides</b>		
glyphosate	Bacterial populations reduced. Fungi and actinomycetes increase, with 9-19% increase in microbial activity. Increased rate of glyphosate degradation over time.	Araujo et al 2003
glyphosate	Short-term changes to community structure. Increased microbial activity and no long-term changes to community structure	Busse et al 2001
glyphosate and paraquat	Activation of urease and invertase soil enzymes, but suppression of phosphatase enzyme	Sannino and Gianfreda 2001
pendimethalin	Soil nematodes and other invertebrates reduced, plant-rhizobium symbiosis affected.	Strandberg and Scott-Fordsmand 2004
atrazine and metolachlor	Altered community structure of several groups of bacteria and actinomycetes.	Seghers et al 2003
atrazine	Significant activation of soil urease activity, and suppression of invertase enzyme.	Sannino and Gianfreda 2001
butachlor	Significant effects on earthworms and soil health	Panda and Sahu 2004
isoproturon	Affects earthworms at very high soil concentrations (not agricultural rates) with LC50 for <i>Eisenia fetida</i> >1000mg/kg	Mosleh et al 2003
oxyfluorfen	Stimulates microbial populations, and increases availability of phosphorus in rhizosphere soil.	Das et al 2003
<b>Insecticides/ nematicides</b>		
chlorpyrifos	Reduced bacterial numbers, but significantly increased fungal numbers.	Pandey and Singh 2004
carbofuran	Significant impacts on acetylcholinesterase activity in earthworms.	Panda and Sahu 2004
dimethoate	Short-term reduction in microarthropod numbers (Collembola), but recovery in numbers after time.	Martikainen et al 1998
malathion	Short-term impacts on earthworm population.	Panda and Sahu 1999
DDT and arsenic contamination	Changes in microbial properties.	Edvantoro et al 2003
DDT	Reduced bacterial and soil algal populations, but may have increased fungal counts.	Megharaj et al 2000
arsenic	Reduced performance of soil functions resulting in reduction of DDT degradation.	Van Zwieten et al 2003
<b>Fungicides</b>		
copper	Earthworm populations avoid soils with concentrations as low as 34mg/kg. Lack of breakdown of organic	Van Zwieten et al 2004

	carbon. Potential long-term implications reported.	
copper	Increased respiration indicating microbial stress. Significantly reduced microbial biomass.	Merrington et al 2002
copper	Reduced performance of soil functions resulting in reduction of DDT degradation.	Gaw et al 2003
metalaxyl	Reduced enzyme activity, in particular dehydrogenase. Toxic to nitrogen fixers.	Monkiedje et al 2002
benomyl	Suppression of respiration, stimulation of dehydrogenase, effects were less noticeable with organic matter addition	Chen et al 2001
benomyl	Significant long-term impacts on mycorrhizal colonization (80% reduction), reduction in fungal to bacterial ratios and nematode numbers.	Smith et al 2000
captan	Suppression of respiration and dehydrogenase, but increases in ammonium nitrogen.	Chen et al 2001
chlorothalonil	Suppression of respiration, stimulation of dehydrogenase.	Chen et al 2001
<b>Antimicrobials</b>		
tylosin, oxytetracycline, sulfachloropyridazine	Tylosin and sulfachloropyridazine significantly impact on gram positive bacteria, while oxytetracycline inhibits general microbial respiration at levels as low as 1mg/kg in soil.	Vaclavik et al 2004
tylosin	Long-term changes to microbial community structure, and short-term reduction in total microbial numbers.	Westergaard et al 2001
<b>Fumigants</b>		
propargyl bromide and 1,3-dichloropropene	Significant effects on respiration and dehydrogenase activity.	Dungan et al 2003

Few studies demonstrate long-term impacts of the pesticide application, and even less discuss measured or observed changes to soil processes. Very significant effects were demonstrated with copper-based fungicides, where Van Zwieten et al (2004) demonstrated reductions in earthworm populations in soil. The authors also mentioned a buildup of organic material on the soil surface, most likely as a result of the lack of bioturbation (mixing of the soil by biota). Likewise, Gaw et al (2004) described the lack of pesticide residue breakdown in soils where copper residues were co-contaminating. Merrington et al (2002) further demonstrated significant impacts on soil microbial processes such as respiration and biomass carbon, and showed conclusively that copper residues resulted in stressed microbes. These impacts are unlikely to change in the near future, as copper accumulates in surface soils and is not prone to dissipative mechanisms such as biodegradation. Also having very significant effects are the soil fumigants, whose purpose in soil is to eliminate biology and any competition for soil resources by the crop. The long-term effects of fumigants however was reduced by the addition of commercially available composted steer manure, where normal biological activity was observed 8-12 weeks following application of high rates of the fumigant (Dungan et al 2003). In the absence of the organic amendment, little recuperation of soil function was detected even after 12 weeks.

## **Impacts of biological pesticides on soil biota**

Microorganisms have been knowingly used to control plant diseases for over 100 years (Winding et al 2004). However, risks of using biological control agents are often forgotten. While the microbes selected may naturally occur in the environment, there are concerns that altering the proportion of these microbes will result in environmental impacts on non-target species including mycorrhizal and saprophytic fungi, soil bacteria, plants, insects, aquatic and terrestrial animals and humans (Brimner and Boland 2003). The authors argue dry-spored biocontrol agents could potentially become a problem (eg allergen to humans) as these spores are more suited to air transport than wet-based spores and therefore more likely to be spread widely.

Recently, there has been evidence that significant non-target effects may occur with bacterial biocontrol agents (Winding et al 2004), however effects were generally observed as short-term and they did not impact on soil health.

## **Implications for farm management**

There is clear evidence that soil biota are impacted by pesticides. Some impacts include the short-term stimulation of enzymatic activity and bacterial numbers, while other impacts include the long-term elimination of earthworm populations. While not an exhaustive list, some important issues that need to be considered are outlined below.

### ***Nutrient management***

Altering microbial populations in soil can affect the availability of nutrients. For example, Taiwo and Oso (1997) report the reduction in phosphorus availability to plants following application of pesticides including pyrethrin, atrazine and metolachlor. Oxyfluorfen has been shown to increase phosphorus availability in rhizosphere soil (Das et al 2003).

### ***Arbuscular mycorrhizal associations***

Fungicides such as benomyl have been shown to have very significant effects on AM fungi (Smith et al 2000), possibly altering the uptake of nutrients by the plant, and affecting natural disease control mechanisms.

### ***Breakdown of organic material***

Residues of copper (Van Zwieten et al 2004) and other pesticides such as malathion (Panda and Sahu 1999) reduce earthworm populations, thus reducing bioturbation and breakdown and incorporation of organic material in soil.

### ***Breakdown of soil contaminants***

Metal based pesticides, in particular copper (Gaw et al 2004) and arsenic (Van Zwieten et al 2003) have been shown to increase the persistence of recalcitrant contaminants such as DDT.

While uncertainty still exists on the implications on many of the effects described in the literature, what can Australian farmers realise from the current state of knowledge? If we think about the paradigm of Newton's Third Law 'To every action there is an equal and opposite reaction' it is almost certain that the application of a pesticide will have some impact on soil biota. Whether this impact is significant for the future health of the agro-ecosystem is still unknown in most cases. The achievement of sustainable agriculture was 'let down' in the 20th century when research focused strongly on soil chemical and physical factors, and neglected biological factors (Sherwood and Uphoff 2000). Concern about soil health is motivated by present and future interest in

both agricultural productivity and profitability. Research on the implications of agricultural management on soil biology is in its infancy, and many important findings will become evident in the coming years.

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# **Can we manipulate resource availability to drive changes in microbial carbon assimilation and nitrogen cycling?**

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## **Introduction**

Decomposition of soil organic matter (SOM) is controlled by microorganisms and leads to the release of nitrogen and other nutrients. However, soil microbial activity is often limited by the absence of readily available carbon-based substrates. This paper discusses the effect of stubble management (stubble burning vs. stubble retention) and temperature on microbial carbon respiration and nitrogen cycling. We then examine the application of labile carbon substrates under controlled conditions on microbial process rates.

Nitrogen supply in dryland agricultural systems is derived predominantly from microbial decomposition of SOM and crop residues as a response to water-limited yield potentials and low inorganic fertiliser inputs (eg less than 50 kg N/ha for wheat production in Western Australia). In the typically low input agricultural systems of WA, up to 80% of soil nitrogen supply in wheat is therefore obtained from microbial transformation of SOM (Angus 2001), compared with 50% in temperate soils (Jenkinson 2001). In many farming systems the amounts of nitrogen cycling through soils during a year are more than enough, even where no fertiliser is applied, to satisfy crop nitrogen demand (see Table 1, Murphy et al, this proceedings). In WA, the role of soil microorganisms can therefore be highly significant in determining the amount and timing of biologically derived nitrogen. Thus an economic benefit can be associated with management of soils for optimal biological fertility influencing potential yield. It is therefore important to identify and understand the primary factors influencing microbial mass and activity in this environment and therefore the potential limitations to gross nitrogen transformation rates (ie total supply of inorganic nitrogen from microbes).

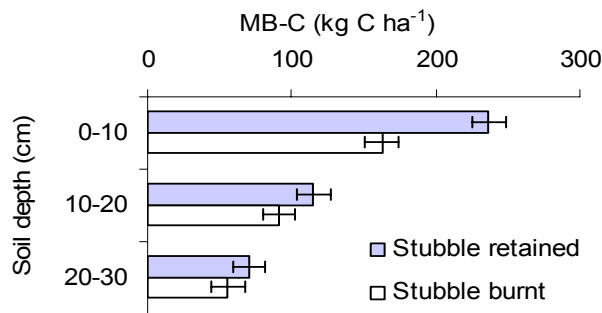
## **Effect of residue management on carbon and nitrogen cycling**

Resource availability influences both carbon and nitrogen dynamics in soil. By limiting readily available carbon, reduced residue inputs can also influence population density and immobilisation of nitrogen which is essential for maintaining microbial growth and activity. Burning stubble for example, has been associated with depletion of SOM levels and declines in microbial biomass and/or activity (Powlson 1987).

Soil nitrogen released from crop residues and soil organic matter results primarily from the activity of microorganisms. Therefore any change in their mass (microbial biomass), or their activity ( $\text{CO}_2\text{-C}$  evolution) can result in changes to the rate of biological soil nitrogen supply. Plant residues are a primary form of organic matter utilised by microorganisms for microbial growth and activity. Since microorganisms are usually starved in soil because they lack available carbon food sources, retention of crop residues provides a practical means of increasing the size of the microbial population. This section investigates the effect of retaining or burning stubble on microbial carbon and nitrogen process rates in a low rainfall (< 325 mm annual rainfall) environment on a red-brown earth (Red Chromosol). Additional trial data and agronomic responses are presented in the poster abstract (Hoyle, Murphy this proceedings).

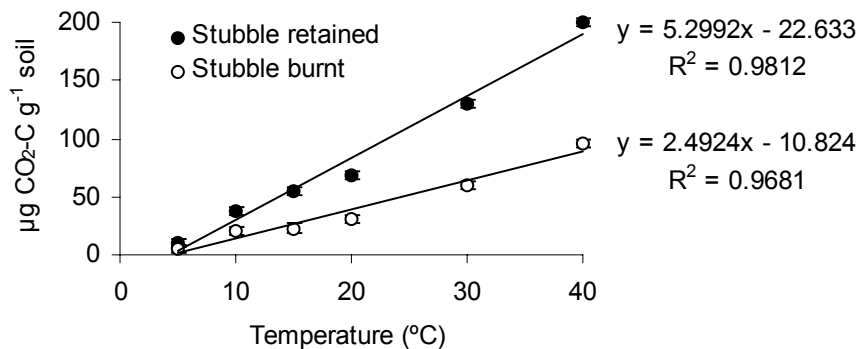


Stubble retention increased microbial biomass in the surface layer (0-5 cm) of the soil by up to 45% compared with burnt treatments, though no significant difference in biomass was observed below 5 cm (Figure 1). Microbial biomass carbon measured to 30 cm depth in this trial was 423 kg C/ha in stubble retained treatments (ie total mass of microorganisms was equivalent to 18 sheep/ha) and 310 kg C/ha in burnt treatments (equivalent to 13 sheep/ha). Microbial biomass nitrogen was measured at 89 and 67 kg N/ha for stubble retained and burnt treatments respectively (data not presented). This means that there is equivalent to 192 kg/ha (stubble retained) and 146 kg/ha (stubble burnt) of urea contained within the soil microorganisms, a significant source of potentially plant available nitrogen.



**Figure 1. The effect of long-term (17 year) stubble management (retained versus burnt) on microbial biomass carbon (MB-C) at different soil depths (Hoyle et al unpublished).**

Cumulative activity (measured as CO<sub>2</sub>-C respiration) was up to 219% greater in stubble retained treatments compared with burnt stubble treatments (Figure 2) after seven days, demonstrating that changes in microbial processes were attributable to the quantity and quality of organic carbon. The average daily rate for CO<sub>2</sub>-C evolution measured was 1.27 μg CO<sub>2</sub>-C/g/soil °C for stubble retained and 0.63 μg CO<sub>2</sub>-C/g/soil °C for burnt treatments (data not presented). These results also illustrate that stubble retention promoted both a larger but also more active microbial community compared with burning.



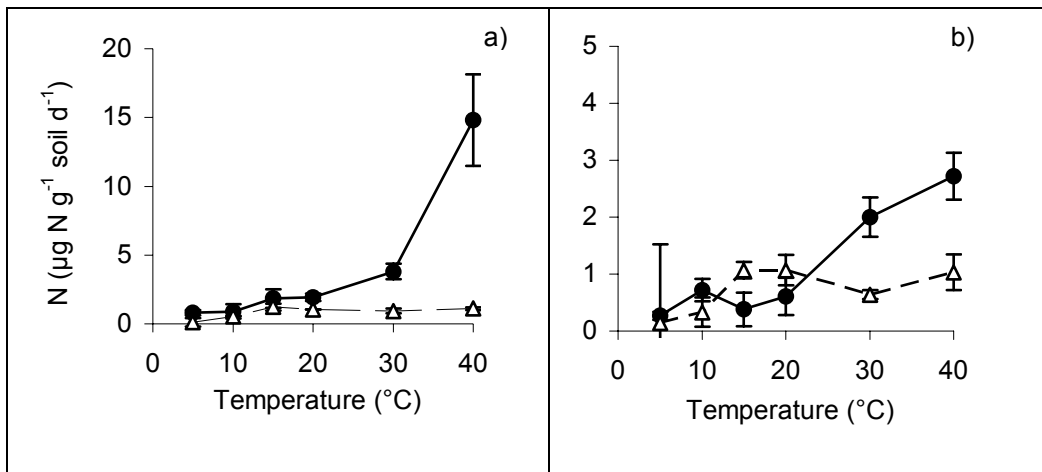
**Figure 2. The effect of long-term stubble management (retained versus burnt) and incubation temperature on microbial activity (cumulative CO<sub>2</sub>-C respired) after seven days incubation (Hoyle et al unpublished).**

### Effect of temperature on nitrogen cycling

The influence of environmental regulators such as temperature and moisture must also be considered as they control microbial activity and regulate substrate availability. For

example, substrate availability in semi-arid soils may be constrained by slower diffusion of nutrients and dissolved OM in soils, with poor connectivity between water films. Few studies have so far examined the effects of high soil temperatures, as typically experienced in Mediterranean or semi-arid environments, on carbon and nitrogen cycling. In particular there is a lack of data on the kinetics of gross mineralisation, immobilisation and nitrification rates at high temperature ( $> 20^{\circ}\text{C}$ ). This section investigates the effect of stubble retention vs. burning on short-term gross nitrogen transformation rates at constant temperatures of 5, 10, 15, 20, 30 and  $40^{\circ}\text{C}$  using  $^{15}\text{N}$  isotopic pool dilution (Murphy et al 2003) on a red-brown earth (Red Chromosol).

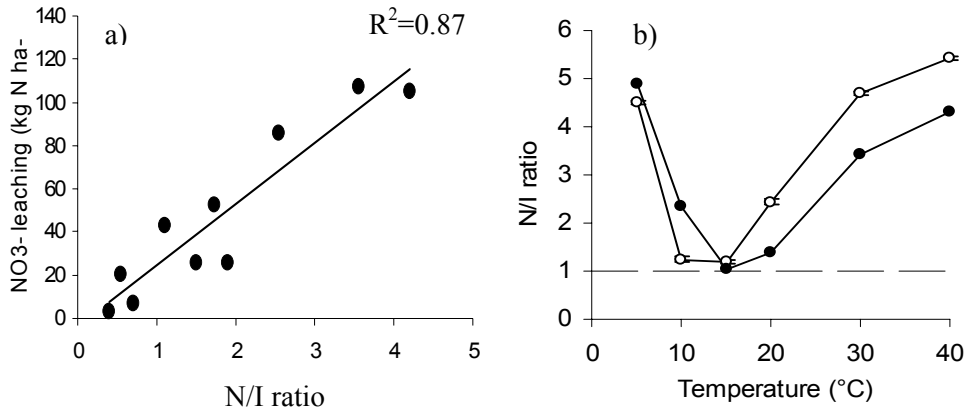
Increases in the inorganic nitrogen concentration measured in our study (data not presented), suggest both mineralisation and nitrification processes remain active at higher temperature. Gross nitrogen mineralisation increased linearly between 5 and  $40^{\circ}\text{C}$  in burnt stubble treatments and between 5 and  $30^{\circ}\text{C}$  in stubble retained treatments (Figures 3a, 3b). In stubble retained treatments, a rapid increase in gross mineralisation rates measured at  $40^{\circ}\text{C}$  resulted in significantly ( $P < 0.001$ ,  $\text{LSD} = 2.921$ ) higher mineralisation rates than either those observed in burnt stubble treatments ( $14.8$  and  $2.7 \mu\text{g N/g soil}$  respectively), or at other incubation temperatures ( $0.8$ - $3.8 \mu\text{g N/g soil}$ ). Microbial immobilisation reached a plateau at  $15^{\circ}\text{C}$  (Figures 3a, 3b), while mineralisation continued to increase. This indicates a separation in the mineralisation (microbial nitrogen supply) immobilisation (microbial demand) turnover that was more apparent at high temperature and in stubble retained treatments. Therefore below  $20^{\circ}\text{C}$ , as is commonly experienced in temperate systems, microbial supply and demand are relatively well matched, so farming systems are less likely to lose nitrogen at these temperatures. However, in warmer climates this means that if water is not a limiting factor at high temperature (eg summer rainfall events), mineralisation can occur, but microbial immobilisation is constrained, leading to an accumulation of inorganic nitrogen susceptible to leaching.



**Figure 3.** Effect of temperature on gross nitrogen mineralisation rate (●) and immobilisation (Δ) in (a) stubble retained and (b) burnt treatments (Hoyle et al unpublished). Capped bars may be smaller than symbols and represent the 95% confidence interval.

Previous research has demonstrated that below a carbon:nitrogen ratio of approximately 22, nitrogen is in excess of microbial demand and released, resulting in an oversupply of nitrogen for microbial use. This means there is more nitrogen available for nitrification and thus the nitrification: immobilisation (N/I) ratio increases. This relationship can be used to determine the risk of increasing nitrate losses within a soil,

and the impact of changing management practice on nitrogen retention vs. loss pathways. Stockdale et al (2002) demonstrate this relationship in temperate grassland soils with greater loss of nitrate associated with increasing N/I ratio (Figure 4a). In our study, soil temperature had a greater influence on the N/I ratio than stubble management treatments, indicating potential losses of nitrate are greater in soils at low (5°C) and high (30 and 40°C) temperature (Figure 4b). This illustrates that environmental variables often override management practices in terms of microbial activity.



**Figure 4. (a) Relationship between N/I ratio and nitrate leaching losses for a temperate grassland (Stockdale et al 2002). (b) Effect of temperature on the N/I ratio in stubble retained (○) and stubble burnt (●) treatments in WA (Hoyle et al unpublished). The dashed line represents a ratio of 1 (ie gross nitrification=microbial immobilisation).**

Studies of the processes controlling soil nitrogen supply confirm that there is a strong influence of soil temperature, water potential and residue quality on gross nitrogen fluxes in soil. Therefore, by changing either the physical status of soil (ie soil pore structure, size and connectivity, water holding capacity), or increasing food supply (ie through increased organic matter returns or application of carbon substrates), we can alter the rate of decomposition and shift the dominance of rates for mineralisation, immobilisation, and nitrification. Since mineralisation and immobilisation are carbon driven and nitrification is nitrogen driven, we can potentially alter the N/I ratio in favour of nitrogen retention by microbes instead of nitrogen loss via leaching. This data illustrates that the relative dominance of these microbial nitrogen pathways defines the amount of plant available nutrients, as well as the capacity of a soil to leach nitrate.

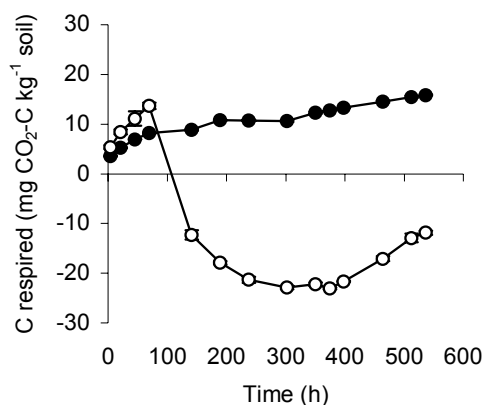
### **What is the capacity to manipulate microbial carbon and nitrogen processes?**

The microbial assimilation of nitrogen is closely linked to the availability of organic carbon to sustain growth and energy requirements. Therefore changes in the mass of microorganisms and their activity are commonly reflected in changes to the soil supply rate of both carbon and nitrogen. Labile carbon substrates may therefore be used in theory to manipulate immobilisation (consumption of ammonium ( $\text{NH}_4$ ) by microbes) in favour of nitrification (loss process), and hence alter the timing of inorganic nitrogen release. The intent is to drive microbial processes toward immobilisation and away from nitrification (which causes nitrate losses).

Although significant quantities of SOM are usually present, it is largely resistant to decomposition and does not provide sufficient maintenance energy for the soil microbial biomass. De Nobili et al (2001) showed that additions of labile carbon substrates (11.3-34 mg C/kg soil;  $\approx$  8 and 24 kg C/ha) such as glucose, amino acids or root exudates to temperate soils caused more CO<sub>2</sub>-C to be evolved than was contained in the original substrate (ie theoretically 'feeding the microbes'). They proposed that this increased CO<sub>2</sub> production may result from an evolutionary strategy used by soil microorganisms to become 'metabolically alert' in response to a forthcoming food event. In this trigger molecule theory, De Nobili et al (2001) hypothesised that detection of these molecules would indicate the pending arrival of fresh substrate and cause the microbial biomass to activate itself in anticipation of a more significant forthcoming food event. Thus the microbial biomass, by investing more energy than was contained in the original substrate, may gain a later benefit. Such a microbial response can in theory also be used to manipulate the dominant nitrogen transformation pathway in favour of nitrogen microbial retention instead of nitrogen loss. However, suitable products or management strategies are yet to be well defined.

This section investigates the effect of glucose-carbon to determine whether microorganisms can be manipulated in such a way on low fertility coarse textured soil from WA. In our experiment, the addition of glucose-carbon to non-cellulose (labile carbon only) and cellulose amended (labile carbon + stable carbon applied) soils resulted in significantly greater ( $P < 0.05$ ) evolution of cumulative CO<sub>2</sub>-C compared with the control (data not presented). However, the cumulative amounts of respired carbon after addition of glucose-carbon to these soils indicated no additional release of CO<sub>2</sub>-C from the microorganisms following application of glucose to either non-cellulose amended or cellulose amended soils (Figure 5, Hoyle et al, submitted). In cellulose amended soils, CO<sub>2</sub>-C evolution was often lower than in control soils suggesting the addition of glucose inhibited cellulase activity, possibly by end-product inhibition (Figure 5).

An alternative interpretation of our findings and those of De Nobili et al (2001), suggest any delayed CO<sub>2</sub>-C evolution is therefore more likely an indirect result of the applied glucose. For example CO<sub>2</sub>-C evolution may result from differences in basal CO<sub>2</sub>-C evolution measured between glucose-amended and non-amended soil, due to the activation of different microbial populations on addition of glucose-carbon, or an accelerated turnover of microbial biomass-carbon, or the formation of a glucose-derived metabolite more readily decomposed by a greater diversity of microorganisms. Therefore although the 'trigger molecule' concept proposed by De Nobili et al (2001) was not evident in soils tested in WA, the application of a labile carbon substrate (in this case, glucose-carbon) resulted in changes to microbial activity and carbon cycling. The implications of these findings to WA farming practices are currently under investigation.

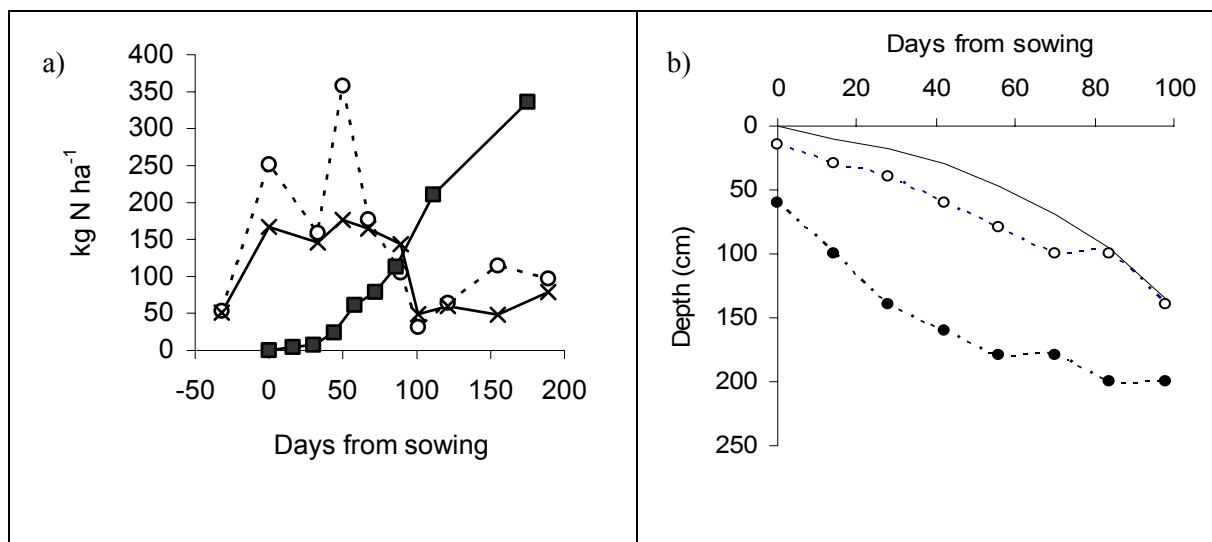


**Figure 5.** Cumulative CO<sub>2</sub>-C evolved (treatment minus control) after addition of a single glucose-carbon solution at 30 mg C/kg soil ( $\approx$  21 kg C/ha) at 0 hours to cellulose amended ( $\circ$ ) and non-cellulose amended soil ( $\bullet$ ) in a coarse textured soil from WA (data from Hoyle et al submitted). Capped bars representing standard errors (n=3) are plotted for each sampling point, and may be smaller than symbols.

### Asynchrony of N cycling

Carbon substrates, plant residues or soil amendments that change the number of biologically active days, or alter activity and function may enhance the timing of release and/or plant nutrient uptake. Plant residues are a primary form of organic matter utilised by microorganisms for microbial growth and activity. Since microorganisms are usually starved in soil because they lack available carbon food sources, crop residues provide a practical means of increasing the size and/or activity of the microbial population. However, nutrients released from residue decomposition may be underutilised by a growing crop, as supply and demand are often not in synchrony (Murphy et al 2004, Ridley et al 2004).

Differences in plant residues and management influence organic matter quality, decomposition rate, and hence the timing and amount of nutrient release. For example, greater contact between plant residue and soil microorganisms, as may occur in a green manure (incorporated) phase can cause faster residue decomposition and hence greater nutrient release than the same residue retained with little or no disturbance (brown manure, desiccation only) as is demonstrated in Figure 6a. Asynchrony is illustrated in this example with high inorganic nitrogen levels early in the season when crop demand is low, demonstrating the potential for high nitrogen loss. Rainfall patterns (amount and distribution) will also significantly influence the release and location (synlocation) of nitrogen within the soil profile (Figure 6b) and hence the capacity of a growing crop to capture nitrogen. In the example demonstrated (Figure 6b), high rainfall is associated with greater distribution of inorganic nitrogen to depth. In this case, the wheat crop gains access to only 19% of the total nitrogen available within the soil profile 40 days after sowing in a high rainfall (leaching) year, compared with 50% under a non-leaching year. Similar improvements in nitrogen capture may also be achieved through management strategies such as deep ripping to improve root exploration (see Murphy et al 2004). These results illustrate that there is potential to improve the synchrony between plant growth and nitrogen uptake in WA farming systems, thereby reducing potential nitrogen losses.



**Figure 6.** (a) Effect of field pea residue management (green manure (○) vs. brown manure (x) crop) on nitrogen release (0-50 cm) and synchrony with nitrogen uptake (■) in a subsequent wheat crop at East Beverley in 2002. Asynchrony is illustrated here with high inorganic nitrogen levels early in the season when crop demand is low. (b) Location of inorganic nitrogen in relation to the rooting zone (solid line) of a wheat crop using actual rainfall (216 mm) from East Beverley in 2002 (○) or simulating a high rainfall (444 mm) season (●). Data simulated using SYN (Hoyle and Murphy, unpublished).

## Conclusion

Stubble retention has been shown to increase the amount of microorganisms in soil compared with stubble burning, resulting in greater soil nitrogen supply. Residue incorporation can also be an effective means of increasing biological soil nitrogen supply and potential grain production. However, potential yield depends on the ability of the subsequent grain crop to utilise this inorganic nitrogen, and is therefore influenced significantly by both rainfall pattern and intensity as this will alter the location of inorganic nitrogen in the soil profile compared with plant root development. Management strategies to optimise the synchrony between nutrient release and crop demand must therefore be considered.

## Acknowledgements

This work was funded by the Grains Research and Development Corporation, with grant support from the Department of Agriculture and the University of Western Australia.

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