Chapter One

General Introduction

1.1 Introduction

Australian cotton has a worldwide reputation as a quality fibre. In the 2002-03 season, Australia produced around 3% of the world's cotton fibre and was the third largest exporter after Uzbekistan and the USA. This made up over 18% of the world export market in that year (Van der Sluijs et al. 2004). In Australia, cotton is grown on about 550,000 ha and the major soil types used in cotton production systems are cracking clay soils (black earths and grey brown clays) (McKenzie 1998; NLWRA 2002) or Vertosols (Isbell 1996a) as these are relatively fertile soils which have clay contents above 50%.

Despite this high chemical fertility, these Vertosols do have inherent problems. They are structurally unstable, tend to slake easily on wetting and are easily compacted. In addition, some of these Vertosols can be sodic at depth. A study by the Cooperative Research Centre for Soil and Land Management estimated in 1997 that in Australia, soil sodicity reduced farmers' incomes by $1.3 billion (Watson et al. 2000). Sodic soils are characterized by their poor structural stability in water. Structural stability in sodic soils, usually measured as dispersion, is said to be determined mainly by the levels of exchangeable sodium (Na) (Rengasamy and Olsson 1991; McKenzie 1998; Rengasamy and Churchman 1999). In Australia, the surface and/or the subsoils of the majority of cotton growing regions are sodic (Northcote and Skene 1972; McKenzie 1998) and dispersion directly affects the hydraulic conductivity, and consequently crop water uptake and water logging potential in Vertosols (So and Aylmore 1993). The dispersion of clay particles is also strongly dependent on the electrolyte concentration in the soil solution (Rengasamy et al. 1984; Quirk 2001). Micaceous minerals frequently occur in several sodic soils where large amounts of potassium (K^+) are usually available (Gupta and Abrol 1990). Traditionally, sodic soil amelioration and management has aimed to displace excess Na^+ from the cation exchange sites and to leach out the Na and thus reduce soil salinity and sodicity to levels that do not limit production and permit high levels of crop productivity.

Soil organic matter (SOM) is considered to be of central importance in maintaining soil quality. It plays an important role in influencing a soil's different physical, chemical and biological properties. These properties, along with soil organic carbon

(C), nitrogen (N) and phosphorus (P), are considered critical indicators for the health and quality of soil. Improved management of SOM in arable soils is essential to maintain the economic and environmental sustainability of agricultural lands and the urban and natural ecosystems with which they interact. It influences a wide range of physical, chemical and biological properties of soil and is considered the most important indicator of soil quality (Carter *et al.* 1999). With the exception of fertilizers, SOM provides the largest pool of macronutrients with > 95% of N and sulphur (S) and 20-75% of P found in SOM (Duxbury *et al.* 1989). Soil structural stability in cracking clay soils with low SOM content is positively related to the frequency of intensive wetting and drying cycles (Sarmah *et al.* 1996; Pillai and McGarry 1999). SOM and fertility commonly decline with cropping on Vertosols (Chan 1997; Dalal and Chan 2001). Cultivated Vertosols in New South Wales have reduced organic matter (up to 40% less), greater pH, less extractable P, and reduced structural stability and biological activity (Chan *et al.* 1988). P deficiency could partially be ameliorated by adding to the soil high quality organic materials which are locally available but are often ignored.

One management technique whereby soil organic matter content can be increased is the addition of organic waste materials from various sources (Reynolds 1930). Although application of organic amendments as a source of SOM and nutrients have a long history of use in agriculture, during the past century inorganic amendments have gained popularity because they are easier to manage, transport and apply (McLaughlin *et al.* 2002). It is also easier to synchronize both the release of nutrients and plant uptake by using inorganic fertilizer than is the case with organic amendments as the former is often released immediately into the soil solution. Although organic and inorganic fertilizer amendments are used mainly to increase the availability of nutrients to plants, they can also affect soil microorganisms (Marschner *et al.* 2003).

Research in India and Canada suggests that applying feedlot manure at rates of 12-16 t/ha significantly enhanced K availability in Vertosols (Singh *et al.* 2002). At the same time laboratory studies have indicated that pelletised mixtures of sewage sludge and cotton gin trash had limited effects on soil physical properties of Vertosols, decreased pH and ESP, and increased organic C and $EC_{1.5}$ (Hulugalle 1996). Application of farm yard manure onto agricultural soil has been a global practice for thousands of years as a means of disposing of small quantities of waste and as a

fertilizer for plant growth (Unger and Stewart 1974) and can result in increased concentrations of soil nutrients (Chang *et al.* 1991; Eghball 2002). Residual effects of organic materials on soil properties can contribute to improvement in soil quality for several years after application ceases (Ginting *et al.* 2003). Adding organic amendments such as compost made from plant materials or municipal solid waste compost may help in the exchange of the adsorbed Na^+ by calcium (Ca^{2+}) and in the displacement of Na^+ (Somani 1990).

While the addition of organic matter is generally regarded as beneficial, it is also necessary to consider the possible negative effects of applying manures. Different types of organic matter act at different scales within the soil structure; sometimes they may enhance dispersion, but their general effect is to bind the soil particles together (Nelson and Oades 1998b). Organic anions enhance dispersion by increasing the negative charge on clay particles and by complexing Ca^{2+} and other polyvalent cations such as those of aluminium (Al), thereby reducing their activity in solution. On the other hand, large organic polyanions can bind clay particles together into stable macroaggregates (Greenland 1965a; 1965b).

But detailed information regarding the benefits of using different organic inputs to improve and maintain soil quality is not readily available for the cotton growers farming on Vertosols in Australia. It is also not known whether any economic benefits are derived by using these organic soil amendments, and how these benefits compare with those resulting from the application of cheaper inorganic amendments such as gypsum. In spite of the general paucity of information on the effects of organic amendments on soil properties of alkaline, swelling soils, in which approximately 75-80% of Australian cotton is grown, many commercial organisations have been promoting the sale and distribution of organic fertilizers and amendments based on data from acidic, non-swelling soils. This extrapolation may well be invalid, prompting many cotton growers to experiment with organic amendments such as feedlot manure, vermicompost and composted cotton gin trash.

1.2   Research objectives

With respect to the sustainability of natural resources and profitability of cotton production enterprises, the current project was initiated with the main objectives being to **identify locally available, economically viable organic amendments and their**

appropriate application rates which will improve soil quality in cotton farming systems, and to determine whether organic amendments are effective in ameliorating sodic Vertosols. The project was funded by the Cotton Research and Development Corporation, Australia, and the outcome of this project would help a manager to make informed decisions on which organic amendment would be most effective in maintaining soil quality and profitability on cotton farms.

The specific aims of this project are:

1. Changes in the physical, chemical and biological properties of Vertosol due to application of organic amendments.

2. Effect of organic amendments on cotton quality.

3. Short-term effects of different organic amendments on C-sequestration (labile C- fractions) and soil nutrient status.

4. Short-term effect of organic materials on soil physical properties.

5. Effect of different organic amendments on amelioration of sodic Vertosols.

6. Economics of using organic amendments in Australia.

1.3 Thesis structure

Chapter 2 provides relevant background research on using different organic amendments for soil quality improvement.

Chapter 3 describes the materials and methods that are common to the experiments described in Chapters 4-7.

Chapter 4 investigates the field application of organic amendments on Vertosol and cotton quality.

Chapter 5 identifies the potentially useful organic materials for Vertosol.

Chapter 6 investigates the appropriate rates of the potential organic amendments identified in the previous experiment.

Chapter 7 examines the effect of organic amendments for amelioration of sodic Vertosols.

Chapter 8 provides the economic analysis of using the organic amendments in cotton production systems in Australia.

Chapter Two

2.1 Introduction

The concept of soil quality integrates soil biological, chemical and physical attributes to assess a soil's capacity to function (Karlen *et al.* 1997). The functions of soil include the supply of nutrients for plant growth (chemical fertility), provide a buffer against pollutants such as pesticides and heavy metals, regulate exchange of gasses (e.g., O_2 and CO_2) through the soil (physical fertility) and provide a physical medium to support plant growth. Soil organic matter (SOM) plays an important role in influencing the soil's different physical, chemical and biological properties by influencing its aggregate stability, water holding capacity, cation exchange capacity, pH and as an energy and nutrient source for soil microbes (Krull *et al.* 2004). Therefore, a complete understanding of the concept and indicators of soil quality is necessary in order to appreciate the SOM as an important indicator of soil quality.

One management technique whereby SOM content can be increased is the addition of organic waste materials (Reynolds 1930). Many landholders have been investigating options of applying recycled wastes organic material to improve soil properties and with a view to partially or fully replacing inorganic fertilizers. Hence, the effects of different organic inputs in different types of soil have been considered particularly in relation to the Vertosols of cotton producing areas in Australia. Australia produces 3% of the world's cotton fibre and the major soil types used for growing cotton are clayey Vertosols (clay content > 50%). They are relatively fertile, but have low organic matter contents and can be sodic. Thus, it is also important to find out how different organic amendments affect the soil quality of a Vertosol as well as their effect on cotton production. The conclusions at the end of this review provide the basis for the approach taken in the experimental work.

2.2 Definition and concepts of soil quality

Soil is a dynamic, living natural system whose healthy condition is vital for both production of food and fibre (economic sustainability) and for environmental sustainability. The quality of soil can affect not only the productivity but also the quality of agricultural produce and the ecological sustainability of land (Acton and

Gregorich 1995), which has great consequences on plant, animal and human health. Soil influences environmental quality and the overall functioning of the biosphere by acting as a living filter, through which water is cycled and xenobiotics are altered (Scholes *et al.* 1994). Modern conservationists make the point that, 'mismanagement and neglect can ruin the fragile resource and become a threat to human survival' (Lal and Pierce 1991). This fact is supported by archaeological evidence showing that soil degradation which was primarily caused by the over-exploitation of the irrigation resources, was responsible for extinction and collapse of the Harappa civilization in India, Mesopotamia in Asia Minor, and the Mayan culture in Central America (Olson 1981).

Various researchers throughout the world have expressed differing and sometimes controversial views on soil quality. With changing social mores and better understanding of soil science, the concept of soil quality was constantly re-defined (Warkentin 1995). The more recent definitions of quality were mainly based on various functions that soil performs in ecosystems. Soil quality consequently becomes inseparable from the idea of ecosystem sustainability. Larson and Pierce (1991) expanded the idea of soil quality from the production or economic point of view to include ecological aspects and defined soil quality as "the capacity of a soil to function within its ecosystem boundaries and interact positively with the environment external to that ecosystem". A more detailed definition was developed by Soil Science Society of America (1995) as "the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain environmental quality and promote plant, animal and human health". This definition is similar to that of Doran *et al.* (1996) in which soil quality is defined as the "capacity of a soil to function, within ecosystem and land-use boundaries, to sustain biological productivity, maintain environmental quality, and promote plant, animal and human health".

2.2.1 Soil quality indicators

Soils are good biophysical integrators of the consequences of management activities across whole catchments (King and Pankhurst 1996). The soil quality concept has three aspects, chemical, physical and biological, which integrate in complex ways to affect a soil's capacity to function (Karlen *et al.* 1997). Due to its multifaceted character, direct measurement of soil quality is difficult, so various soil attributes are

used to serve as indicators of soil quality change. These attributes or indicators may be defined as measurable soil properties that influence the capacity of the soil to perform a specific function (Acton and Padbury 1993). Ideally, indicators should (i) encompass ecosystem processes, (ii) be sensitive to changes in climate and management practices, (iii) be easily measurable and reproducible, (iv) already be a component of existing soil databases, so any changes can be monitored and be accessible to many users and (v) be applicable to field conditions, and integrate soil physical, chemical and biological properties and processes (Doran and Parkin 1994).

Based on these important soil properties, Doran and Parkin (1994) have developed a list of basic soil parameters or indicators for screening for good soil quality and health which encompasses the physical, chemical and biological properties of soil. Physical indicators include soil texture, depth of soils, infiltration rate, bulk density and water holding capacity. Hseu et al. (1999) also included aggregate stability as an important physical indicator. Chemical indicators in the Doran and Parkin (1994) list are SOM or organic C and N, soil pH, electrical conductivity and extractable N, P and K. Hseu et al. (1999) also included available trace elements such as zinc (Zn), cadmium (Cd) and copper (Cu) as chemical indicators. Biological indicators included microbial biomass C and N, mineralizable N (anaerobic incubation) and soil respiration. Microorganisms play a particularly important role in the transformation of organic matter and 1-5% of C and N in soil are stored in living microbial tissue (Duxbury et al. 1989). Based on gross microbial activity in soil, microbial biomass C and basal respiration can be suggested as possible indicators of soil quality. Though difficult to measure, the degree of microbial diversity, in particular, is also thought to provide a measure of soil quality (Duxbury et al. 1989).

2.2.2 Significance of the soil microbial biomass and respiration

The microbial biomass denotes the living component of SOM (Jenkinson and Ladd 1981) apart from macro-, meso- and micro-fauna and plant roots. Microbial biomass in soil is composed of bacteria, fungi, algae and actinomycetes. The microbial biomass contributes to the maintenance of soil fertility and soil quality in both natural and managed ecosystems (Hart et al. 1986; Elliott et al. 1996).

Soil microbial biomass serves as: (i) a labile source or an immediate sink of available nutrients and organic substrates on a short term basis (Diaz-Ravina et al. 1995), (ii) a

driving force of nutrient transformation from stable organic forms to available mineral forms over longer periods (Smith and Paul 1990), (iii) an agent of polysaccharide secretion, a cementing agent of soil aggregates (Kennedy and Papendick 1995), and (iv) an agent releasing enzymes which are responsible for nutrient cycling (Saffigna *et al.* 1989; Carter 1991; Srivastava and Singh 1991).

Microbial biomass C (expressed as μg/g soil) comprises 1 to 5% of total soil organic C (w/w) and microbial biomass N ranges from 1 to 6% (w/w) of total soil organic N (Jenkinson and Ladd 1981; Wardle 1992). The size and activity of the microbial biomass is regulated by the quantity and quality of SOM and climatic conditions (Insam 1990), soil moisture content (Doran *et al.* 1990), soil temperature (Joergensen *et al.* 1990), soil pH (Roper and Gupta 1995), soil structure and texture (Ladd 1992) and soil and crop management practices (Amato and Ladd 1992; Srivastava and Lal 1994).

There is a strong link between soil microbial biomass, soil fertility and soil health (Hart *et al.* 1986; Insam *et al.* 1991; Srivastava and Singh 1991). The microbial biomass C and N have been used as sensitive indicators of changes in soil processes due to changes in management practices and cropping systems, because they have a much faster rate of turnover than the total SOM and more responsive or sensitive to change in soil physico-chemical conditions (Sparling 1992; Gupta *et al.* 1994).

Microbes use SOM as both a food (and energy) and nutrient source. The addition of high quality organic matter to agricultural soils should enhance soil microbial levels. Bolton *et al.* (1985) reported that addition of green manure to wheat based cropping systems over a 30 year period caused a significant increase in microbial biomass. Addition of farmyard manure (FYM) usually increases microbial biomass (McGill *et al.* 1986). Often, there is a strong correlation between soil organic C content and soil microbial biomass (Biederbeck *et al.* 1984; Dick 1992). Thus management practices that encourage incorporation of organic residues typically increase biological activity.

Increases in plant biomass production due to use of inorganic fertilizers e.g., superphosphate, increase the amount of organic residue return to the soil each year and also stimulates biological activity (Marshall 1977). Liang and Mackenzie (1996) conducted an experiment to study the effect of inorganic fertilization on organic C and microbial biomass N. They found that increasing fertilization rates above normal

(local area practice) increased microbial biomass - N immobilization with a subsequently greater amount of N released. Higher fertilization rates significantly increased both magnitude of soil microbial biomass N and fertilizer N recovery in the soil microbial biomass.

Cultivation and organic amendments influence microbial activity in soil. If the average C:N ratio for the total microbe population is 8:1, then soil with a C:N ratio of 24:1 would be ideal. At higher C:N ratios, N becomes limiting. A higher microbial biomass does not necessarily relate to greater soil fertility. Biederbeck *et al.* (1984) reported more biomass C and N under wheat with no fertilizer than the wheat receiving continuous N fertilizer. Tate *et al.* (1991) found more organic matter and a greater microbial biomass C on a low fertility pasture site compared to a high fertility site. Biederbeck *et al.* (1984) suggested that a larger but less active biomass was present with low turnover in the poorly fertilized system compared to low fertility soil. In addition, the forms of the organic C in the unfertilized soils could be of the less labile and intractable forms than those found on fertilized pastures especially if legumes were present. Legume residues decompose between 5 to 8 times faster than grass residues (King 1989). Crops grown on soils with greater quantities of available nutrients may need less extensive root systems resulting in less below-ground plant biomass, lower root exudates and consequently a decreased microbial biomass. Long-term fertilizer use can improve soil nutrient status but at the same time may cause soil acidification and perhaps decrease the microbial biomass while still improving the crop yield (Ladd *et al.* 1994).

A disadvantage in using some organic amendments as fertilizers is contamination with toxic heavy metals. Heavy metals can have adverse effects on microbial biomass. Brookes and McGrath (1984) reported that soil amended with contaminated sludge over 30 years showed higher organic C and respiration but lower microbial biomass as compared to uncontaminated soil. Campbell *et al.* (1991) reported that the microbial biomass N was useful in predicting a change in soil quality than microbial biomass C. In contrast, Jordan and Kremer (1994) reported microbial biomass C to be a better indicator of soil quality.

Some authors prefer to use ratios of various microbial parameters as indicators. The respiratory quotient is expressed as soil respiration per unit of microbial biomass C. This measure is also termed the 'specific respiration rate' and usually expressed as µg

CO_2–C/hour/mg microbial biomass C. Anderson and Domsch (1990) suggested that respiratory quotient could be used as an indicator of soil development, substrate quality, ecosystem development and ecosystem stress. For example, Brookes and McGrath (1984) observed a higher respiratory quotient in soils containing heavy-metal contaminated sewage and sludge, compared with control soils containing no heavy metals.

Another ratio is the microbial quotient or the ratio of microbial biomass C (C_{mic}) to total soil organic C (C_{org}) (% C_{mic}/C_{org}) (Anderson and Domsch 1989; Sparling 1992). The microbial quotient has been used as an indicator of changing soil processes under different cropping sequences and management practices and may be more useful measure than either microbial biomass C or total C individually (Anderson and Domsch 1989; Sparling 1992) because as a ratio, it avoids the problems of working with absolute values and helps in comparing across soils with different organic matter content. One problem associated with this ratio is that both components have a common origin, and are not independent of each other. Also, changes in organic C will have more impact on the ratio than changes in microbial biomass since the former is quantitatively much more abundant.

Soil respiration or C mineralization is an index of total soil biological activity including soil microorganisms, macro-fauna and plant roots (if the latter two entities are left in soil during measurement) and it reflects the overall activity or rate at which energy is spent by the indigenous soil microbial pool (Anderson and Domsch 1990). As noted by Sparling (1997), microbial respiration is highly variable and can show wide variation depending on substrate variability in quantity and quality, moisture and temperature. Rate of soil respiration depends on the substrate availability, moisture and temperature of the soil (Orchard and Cook 1983; Brookes 1995). For example, rewetting of air-dried soil increases the respiration and mineralization of C and N from SOM (West *et al.* 1988; Van Gestel *et al.* 1992). Because of the great variability of microbial activity under natural conditions, for valid comparisons to be made between different management practices and for it to be useful as a soil quality indicator, respiration measurements must normally be carried out under controlled laboratory conditions (Anderson 1982; Brookes 1995), particularly with respect to temperature and moisture.

2.3   Soil quality and role of soil organic matter

2.3.1   What is soil organic matter?

Organic matter is widely regarded as a vital component of a healthy soil. It is an important part of soil physical, chemical and biological fertility. The term "soil organic matter" has been used in different ways to describe the organic constituents of soil. Baldock and Skjemstad (1999) defined SOM as "all organic materials found in soils irrespective of origin or state of decomposition". In a broader sense, SOM comprises all living soil organisms and all the remains of previous living organisms in their various degrees of decomposition. The living organisms can be animals, plants or microorganisms, and can range in size from small animals to single cell bacteria only a few microns long. Since SOM consists of C, hydrogen (H), oxygen (O), N, P and S, it is difficult to actually measure the SOM content and most analytical methods determine only the soil organic C content and estimate SOM through a conversion factor.

However, soil type, climate, management, mineral composition, topography, soil biota (the so-called soil forming components) and the interactions between each of these factors are the modifying processes that will affect the total amount of soil organic C present in a profile as well as the distribution of soil organic C with depth.

2.3.2   Effect of soil organic matter on soil properties

It is widely recognized that the SOM plays an important role in different soil physical, chemical and biological properties and these properties, along with soil organic C, N and P, are considered critical indicators for the health and quality of soil. In general, increase in SOM is seen as desirable by many farmers as higher levels are viewed as being directly related to better plant nutrition, ease of cultivation, penetration and seedbed preparation, greater aggregate stability, reduced bulk density, improved water holding capacity, enhanced porosity and earlier warming in spring (Lal 2002). Reeves (1997) noted that "soil organic carbon is the most often reported attribute from long-term agricultural studies and is chosen as the most important indicator of soil quality and agronomic sustainability because of its impact on other physical, chemical and biological indicators of soil quality". Janzen et al. (1992) pointed out that the relationship between soil quality indicators and soil functions does not always comply to a simple relationship increasing linearly with magnitude of the indicator.

However, irrespective of soil type it appears that if soil organic C contents are below 1%, it may not be possible to obtain potential yields (Kay and Angers 1999).

2.3.2.1 Soil organic matter as a source of nutrients

SOM is an important source of nutrients for plants. Most of the nutrients in SOM are derived from the mineralization of SOM and become available for plant uptake during decomposition and for this reason the particulate organic matter fraction (section 2.5) which can constitute from a few percent up to 25% of the total organic matter in a soil is often considered the most important proportion of SOM in providing nutrients to plants (Wolf and Snyder 2003). Losses of nutrients might occur via leaching or conversion to gaseous forms or are the result of immobilization. Some soil nutrients are used in the synthesis of new biomass, some are immobilized and another portion is mineralized and released as plant-available forms into soil mineral pool. With the exception of fertilizers, SOM provides the largest pool of macronutrients with > 95% of N and S and 20-75% of P found in SOM (Duxbury *et al.* 1989; Baldock and Nelson 1999).

The particulate organic matter pool is the most important fraction of SOM with respect to providing N and increasing availability of micronutrients for crop growth. These results are consistent with many other studies, where the 'active' fraction was implicated as being the most important component in providing nutrients and regulating nutrient supply (Loveland and Webb 2003).

2.3.2.2 Effect on physical properties

It is generally accepted that many soil physical properties are related to the organic matter content (McLaren and Smith 1996). The interrelationship between SOM and soil structure and other physical properties has been extensively studied and it is well established that addition of SOM can not only reduce bulk density and increase water holding capacity, but also effectively increase soil aggregate stability. Kay and Angers (1999) reported that a minimum of 2% soil organic C was necessary to maintain structural stability and also observed that if soil organic C content was between 1.2-2.5%, stability declined rapidly.

2.3.2.3 Effect on microbiological properties

Dick (1992), in a review, noted that there is generally a positive relationship between soil C content and soil microbial biomass, and concluded that any practice that

increases the amount and incorporation of organic residues into the soil increases biological activity. Since both organic and inorganic inputs, when coupled with appropriate management, increase the amount of residue returning to the soil, it follows that both can have beneficial effects on soil biological activity. It is also predictable that manures, when applied at the same nutrient inputs, will have a larger effect, relative to fertilizers, on soil biological activity.

2.3.2.4 Effect on sodicity

Sodic soils generally have low organic matter contents, due to their relatively low productivity and associated organic matter inputs and their relatively high losses of organic matter in erosion and leaching events (Nelson and Oades 1998a). In sodic soils, organic matter has been shown both to enhance dispersion (Gupta *et al.* 1984; Churchman and Oades 1993), and to reduce it (Loveland *et al.* 1987; Barzegar *et al.* 1997). The differences in the degree of sodicity and the chemical and physical nature of the organic matter may have caused these conflicting results. Generally, it acts as a coating and linking agent to soil minerals, and acts in promoting macroaggregate stability. Gupta *et al.* (1984) observed increased soil dispersion with addition of organic matter at high sodium adsorption ratio (SAR). Black and Abdul-Hakim (1985) reported that organic matter can prevent the loss of permeability of soil at low SAR. Humic/fulvic ratios increased with exchangeable sodium percentage (ESP) and C became more easily extractable at high ESP values (Laura 1976).

Exchangeable Na, in the presence of low electrolyte concentration, increases clay dispersion by increasing net repulsive forces at the size scale of the clay particle (Shainberg *et al.* 1989). Organic matter helps to prevent clay dispersion by stabilizing structure at range of size scales. Roots and fungal hyphae help to prevent the breakdown of larger aggregates, whereas at smaller scales, polymers such as polysaccharides, aliphatic materials, and organic anions bind clay particles together through a variety of bonding mechanisms (Oades 1993). In 1998, Nelson *et al.* reported that the decomposition of organic matter might reduce the clay dispersion in sodic soils by altering electrolyte composition.

2.3.3 Organic amendments as replacements for inorganic fertilizer

Currently world-wide concern regarding the impact of modern farming practices on soil and water quality and much recent research has focused on management options

for reducing nutrient runoff and leaching (Sharpley *et al.* 1994; Gillingham and Thorrold 2000) and improving soil quality (Dick 1992; Liebig and Doran 1999). Alternative management systems, such as organic and biodynamic farming, are being promoted on the basis that they are more environmentally benign and specifically enhance soil and water quality relative to conventional practices (Conacher and Conacher 1998). These systems place much emphasis on the importance of SOM and biological activity and promote the use of 'natural' sources of nutrients (i.e. unprocessed raw materials) and manures. At the farm system level there is evidence showing that, relative to conventional management practices, organic and biodynamic farming can increase SOM and related soil chemical, biological and physical properties (Reganold 1995; Liebig and Doran 1999).

Increase in SOM following the conversion of conventional to organic farming practices could be attributed to either the use of manures or a change in cropping rotation, including green manure crops. There is an abundance of evidence showing that the application of fertilizers to nutrient deficient soils generally increases SOM content (Paustian *et al.* 1997). The primary reason for this is that fertilizers increase crop production, thereby increasing the amount of plant residue, including roots, returned to the soil. A consequence of the increase in SOM is that some soil chemical, biological and physical properties are also improved (Perrot *et al.* 1992; Fraser *et al.* 1994).

It is likely that manures will have greater effect on SOM and related soil properties than fertilizers, when applied at equivalent nutrient inputs as they supply additional exogenous source of organic matter. However, this may not always be the case as organic matter accumulation depends on the net incorporation of organic matter in the soil, which in turn depends on the cropping and management system employed (Dick 1992; Paustian *et al.* 1997). According to a review by Edmeades (2003) on the long-term effects of manures and fertilizers on soil productivity and quality, the manured soils have a lower bulk density and increased porosity and aggregate stability, but such changes in the physical properties may not always be beneficial. For example, if the permeability of water through the soil profile is enhanced, then greater leaching of nutrients may result, although this could simultaneously reduce surface runoff (Johnston 1997).

Inorganic fertilizers have variable effects on soil microbial populations. Superphosphate would seem to have few deleterious effects on soil biota in the long-term (Fraser *et al.* 1994). In their review of management practices and soil biota, Roper and Gupta (1995) stated that microbial biomass is reduced if fertilizer application causes change from the normal pH, but there can be an increase in microbial biomass if environmental conditions (e.g. pH) remain unchanged. Where manure and inorganic N fertilizers were applied together, the depressive effect on soil microbial biomass of the inorganic fertilizer would seem to be offset ((in Hunt 1998).

2.4 Different organic amendments and their effect on soil quality

The characteristics of different organic amendments are presented in Table 2.1.

Table 2.1 Characteristics of different organic amendments

Parameter	[A]Biosolids	[B]Poultry litter	[C]Cattle feedlot manure	[D]Composted cotton gin trash
pH	6.7	8.1	7	-
$EC_{1:5}$ (dS/m)	0.4	6.8	12.4	-
Dry matter (g/100 g)	17.7	75	-	-
C (g/100 g)	-	36	-	10
N (g/100 g)	3.7	2.6	2.2	1.2
P (g/100 g)	3.4	1.8	0.8	-
K (g/100 g)	0.3	1	2.3	1.3
Ca (g/100 g)	1.2	2.5	-	3.3
Na (g/100 g)	0.1	0.3	0.6	0.036
Cu (mg/kg)	100-500	0	-	-
Zn (mg/kg)	200-2500	-	-	-

Sources:
[A] NSW EPA (1997)
[B] Griffiths (2004)
[C] McGahan and Tucker (2003)
[D] N. Hulugalle (pers. comm.)

2.4.1 Application of biosolids

Recycling of on-farm wastes to agricultural land has become a well accepted method of maintaining and/or improving the fertility of the soil; however, off-farm waste recycling is a relatively new concept for Australian agriculture (McLaren and Smith 1996). Off-farm wastes generally come in two forms: a composted product that may

contain domestic household waste (garden and kitchen wastes, paper etc.) or waste water products (sludge, dewatered cake, composted products, i.e., 'biosolids') from the water treatment process. Biosolids are defined as 'organic solids derived from biological wastewater treatment processes that are in a state that can be managed to sustainably utilize their nutrient, soil conditioning, energy, or other value' (EPA Victoria 2004). Recently, agriculture has been identified by the water industry as a potential market for a wide range of biosolids (ARMCANZ 1995; Reid 2002; DSE 2003). Using municipal biosolids as a fertilizer and to reclaim degraded land is a common and beneficial alternative to land filling (Brown et al. 2003).

Biosolids have been used in agricultural land in many countries around the world (Lerch et al. 1990). Initial application of biosolids on agricultural land was mainly a means of disposing of municipal waste. Before 1990, most solid wastes in Australia were disposed of by ocean dumping. Environmental concerns associated with increasing coastal pollution prompted waste treatment authorities to recycle biosolids. Biosolid composition can vary greatly (Evans et al. 1996). Although most biosolids contain large amounts of N and P that are important for plant growth, they can also contain high concentrations of heavy metals such as Cu, Zn or Cd (Epstein 1980). This can cause problems if the heavy metals are taken up by plants and then ingested by animals or humans, because excessive levels of most of the heavy metals are toxic (Underwood 1977). As for example, Table 2.1 also showed that the P content of biosolids was four times higher than cattle manure and two times than poultry litter. It also had high N content compared to poultry litter, cattle manure and coposted cotton gin trash (Table 2.1). Soil enzymes play an essential role in the decomposition of SOM. These enzyme activities are affected by herbicides and heavy metal contamination present in amendments of organic residues and biosolids (Saffigna et al. 1989; Benitez et al. 1999). The addition of biosolids increased organic C and available nutrients; similarly, hydrolytic enzymes such as urease, alakaline phosphatase, arylsulfatase activities were increased by biosolids application (Kizilkaya and Hepsen 2004).

Gaskin et al. (2003) studied the effect of long-term repeated biosolid application over periods of up to 12 years on metal concentration in soil and bermudagrass forage from fields that participated in a biosolid land application programme by a City of Augusta-Richmond Country (Georgia) waste water treatment plant. According to their

findings, the water-soluble metals increased with long-term application of biosolids and Cu and molybdenum (Mo) concentrations in the soil were significantly higher in the fields that had received biosolids for more than six years. They concluded that the forage quality from fields with long-term application of biosolids was similar to that having only commercial fertilizer and should not pose a risk to animal health.

According to the study done by Weggler-Beaton *et al.* (2003) on the influence of low rates of air-dried biosolids on yield and P and Zn nutrition of wheat and barley, application of 2 t biosolids were equivalent to a standard mineral fertilizer application in enhancing the growth and grain yield of wheat and barley. They also concluded that the biosolid-borne Zn enhanced plant growth and tissue Zn concentrations to a similar extent to the mineral source of Zn and also the Zn uptake in shoots and grain was positively correlated with biosolid application rates. The relative efficiency of biosolid-P compared with mineral P was shown to increase with time (McLaughlin and Champion 1987) and Pastene (1981) noticed that biosolid-P availability is lower in biosolids with a high content of Al and iron (Fe) due to P-fixation.

Yield increases associated with biosolids application have been reported in many locations and for various crops (Fresquez *et al.* 1990; Al-Mustafa *et al.* 1995). Cooper (2005a) studied two forms of biosolids, with and without lime, which were applied to acid soils and wheat and triticale were grown to determine the effect of biosolids on crop growth and yield. He found that the biosolids can produce large yield increases with cereals and those responses lasted for more than a single season. His results also showed that lime-amended biosolids significantly increased the soil pH and it can be used in place of agricultural lime on acid soils, and their effect on soil acidity lasts several years.

In a four year study, Binder *et al.* (2002) reported that application of biosolids to maize and sorghum resulted in maize yields which were similar to, or exceeded, those in plots where N was applied as mineral fertilizers. This suggests that a large amount of biosolid- N were available to plants in the first year of application. About 80% of biosolid- N was in organic form and much of this was mineralized over a period of four years. Sorghum yield also increased significantly due to application of biosolids. Benefits of biosolids application in terms of N use efficiency and profit were greatest in irrigated systems where crop growth was likely to be limited by factors such as water supply. Application of biosolids increased the seed yield of canola grown in

Dermosol in north-east Victoria, Australia as compared to both the unfertilized control and fertilized treatment (Stokes *et al.* 2004).

In summary, the sociological and economic factors which determine the use of biosolids in agriculture have to be balanced with the benefits and risks to the agricultural production systems in relation to contaminants (Hill 2005). Given the potential risks of heavy metal contamination, strict legal and regulatory constraints on the recycling of biosolids to agricultural land are necessary to ensure the protection of consumers and the environment.

2.4.2 Application of manure

Application of manure can improve the nutrient status of the soil and increase soil organic C levels. Aoyama *et al.* (1999a) observed an increase in SOM with addition of manure and consequently the formation of slaking-resistant macroaggregates. Application of manure or composted manure can result in increases in soil concentrations of nutrients and organic matter (Eghball 2002).

Cultivation is reported to reduce SOM content and changes the distribution and stability of soil aggregates. Macroaggregates are disrupted, and soils become more susceptible to erosion as cultivation intensity increases, which can contribute to a further loss of SOM (Elliott 1986; Camberdella and Elliott 1993; Six *et al.* 2000). Long-term application of animal manure increases the SOM content in two ways: (i) by adding organic matter in manure (Sommerfeldt and Chang 1985) and (ii) by adding organic matter in crop residues, due to higher crop yield in soils receiving manure (Angers and N'Dayegamiye 1991). One year after manure application, Sun *et al.* (1995) found a significantly higher proportion of water stable macroaggregates in eroded soils treated with cattle, hog, or poultry manure, compared with unamended soils.

The residual effects of increased nutrients and organic matter in soil following manure or compost application on crop yield and soil properties can last for several years (Mugwira 1979). Soils amended with manures that do not contain toxic elements (e.g. heavy metals) generally have higher biological activity than those managed with mineral fertilizers (Marshall 1977; Dick 1992). Manure additions have several effects on cropping soils ((in Hunt 1998). Additional C and nutrients are added to soil and they supply energy and minerals for microbial growth. Increased levels of organic

matter have beneficial effects on soil micro-climate and structure, both of which favour the development of higher levels of microbial population in soil.

Generally, fresh manure is expected to contain more mineral N, especially NH_4-N than compost and much of this NH_4-N is lost during the composting process leaving behind more stable organic N compounds (Parkinson *et al.* 2004). Eghball (2000) also found higher mineral N levels in soils when manure was applied to either no-till or conventional tillage systems.

Eghball *et al.* (2004) noticed that residual effects of N- and P-based manure and compost application on corn yield and N uptake can last for at least one growing season. Residual effects of N-based manure and compost applications on corn production were greater than the P-based treatments since the amounts of manure or compost applied were greater for N- than P-based management systems. They ultimately concluded that both the corn production and soil properties were improved by the residual values of applied manure or compost.

Zaller and Kopke (2004) studied the effects of application of traditionally composted FYM and two types of biodynamically composted FYM over nine years on different soil chemical, biochemical and biological properties. Soil pH, C/N ratio, available P and available K contents were all significantly increased by FYM application, but available P contents were generally lower in the deeper soil layers than the surface soil layers. Basal respiration, microbial biomass, microbial quotient (C_{mic}/C_{org}) were significantly lower in deeper soil layers, whereas metabolic quotient was not affected.

Application of high rates of cow manure (both fresh and composted) to soil resulted in enriched organic matter surface layer characteristics of a long-term no till field (Wang *et al.* 2006). They concluded that an initial high-rate application of organic amendments to a heavy clay soil can help to maintain maize yield. However, if manure is used as the organic amendment, it must first be composted or applied to the field early enough in advance of crop planting to avoid the problem of poor crop emergence, low plant populations, and decrease in yield.

In their study on the long-term effect of FYM on some important soil biochemical characteristics, Saviozzi *et al.* (1999) reported that the available P content was higher in the plots that received FYM than the control plots, indicating that long-term amendment plus mineral fertilization made the P budget highly positive under FYM

treated plots. The light fraction of organic matter was also higher in FYM treated soil, probably as a result of presence of straw in the FYM.

Manure application significantly increased soil total C and N (compared with fertilizer addition) through improved formation of water stable aggregates and increased aggregate-associated C and N (Mikha and Rice 2004). According to Aoyama et al. (1999a), additional accumulation of SOM with manure was mainly from manure application itself and not from changes in plant biomass. They also observed an increase in particulate organic matter and mineral-associated organic matter in aggregate fractions with manure application. They explained that some of the mineral-associated organic matter derived from the decomposition of particulate organic matter, stimulates formation and stabilization of macroaggregates.

Mikha and Rice (2004) observed increased labile C content in both macro- and microaggregates which indicated that the manure addition could improve SOM conservation and reduce nutrient loss. These results are similar to that of Angers et al. (1997) who showed that the amount of residual new-labeled ^{13}C and ^{15}N recovered in microaggregates increased with time. Aoyama et al. (1999b) observed a three-fold increase in protected C and a four-fold increase in protected N following manure application in aggregates 250 to 1000 μm in diameter.

Despite benefits of soil aggregation from applying animal manure to cultivated soils, long-term manure applications can also contribute to nutrient accumulation, particularly P, in agricultural soils. As P accumulates in soils, the risk of P transport to water bodies through erosion, leaching, and runoff processes increases (Lennox et al. 1997; Whalen and Chang 2001). Improvements in soil aggregation from manures might reduce P transport from agricultural soils through reduced erosion, but there are conflicting reports on the distribution of P and other nutrients in macroaggregates from cultivated, manure-amended soils.

In 2002, Whalen and Chang observed that long-term addition of manures can shift the aggregate size distribution from larger (> 12.1 mm) to smaller (< 2 mm) dry-sieved aggregates. Consequently, a greater proportion of soil mass was at risk of wind erosion following many years of continuous manure applications because of increase of macroaggregate dispersion. They also found that dry-sieved aggregate fractions

from soils amended with manure contained more total C, N and P than unamended soils.

In their survey, Reddy *et al.* (2005) reported that farmers generally believed that application of FYM would improve soil productivity due to its favourable effects on soil physical conditions (tilth and loosening of soil) and water-holding capacity. This perception was based on their working experience with FYM over an extended period of time. In spite of the perceived positive effects of FYM on crop yields, there was little awareness of its plant nutrient value, since farmers usually under-fertilize their crops when FYM is applied (Reddy *et al.* 2005). The main drawbacks to FYM use were believed to be an increased incidence of weeds, pests and diseases.

2.4.3 Application of other organic amendments

Maintenance of SOM requires that efflux (decomposition of existing SOM) does not exceed influx of new C. Crop residue is an important source of new C for building and maintaining SOM. Buyanovsky and Wagner (1997) found that 80% of the crop residue added to a field is returned to the atmosphere within two years.

Corn stover is an important organic amendment which provides a food source for soil fauna, surface residue to minimize erosion, and contributes to nutrient cycling. Corn residue C converted to SOM ranges from 8 to 18% (Barber 1979). The small amount of new C that remained implies that a large C influx is important for maintaining SOM. Johnson *et al.* (2004) observed that amending soil with the fermentation by-product following ethanol production from corn stover could improve soil properties. In their experiment, the ratio of observed C released to N mineralized increased with the addition of corn stover. N immobilized into the microbial biomass was likely to be the reason for increasing this ratio. Corn stover addition led to immobilization of N into microbial biomass.

Agricultural application of [1]sewage sludge is becoming popular as a means of nutrient recycling in many areas of the world (Jacobs 1981). Because of a growing need to apply municipal sewage sludge on agricultural lands, there is a developing urgency to have criteria for disposal practices that will preserve the productivity of these lands and prevent entry of potentially toxic elements (Chaney *et al.* 1987). Agricultural

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[1] The term 'biosolids' is used to indicate sludge or the municipal soild wastes, differerent authors used different terms, but both are same.

management has a large impact on nitrate leaching (Finke 1993) and nitrate leaching from agricultural lands has been focused mainly on sandy soils as clay soils are usually not considered to have a high nitrate leaching potential (Stout et al. 2000). In his study, Lee (2004) showed that application of sewage sludge up to 50 Mg/ha would be safe with respect to leaching of nitrate and infiltration into ground water in clay soils. Saviozzi et al. (1999) noted that application of sewage sludge over a 12 year period significantly increased the available P content, reflecting the low K content in sewage sludge.

Piccolo et al. (1997) and Piccolo and Mbagwu (1999) suggested that application of humic substances (lignite or oxidized coal) would be an economically viable source of rehabilitation of degraded soils as humic substances are relatively inexpensive and only small amounts are required compared with much larger amounts of FYM applications.

Diaz et al. (1994) conducted a study to investigate the effects of two organic amendments (peat and urban refuse) on soil structure over a period of two years. They found that average percentage of stable aggregates increased with amount of urban refuse applied compared with the control. By comparison, no improvement in stable aggregate formation was found in peat application. The authors concluded that only one application of urban refuse was sufficient to regenerate degraded soils. However, application of urban refuse resulted in increased vegetation cover compared with the control and peat treatment, which was likely to have contributed to the long-term stabilization of soil aggregates.

In order to investigate whether different types of organic amendment differ in their effects on soil physical properties, Martens and Frankenberger Jr. (1992) amended an irrigated coarse loamy Alfisol with three loadings of poultry manure, sewage sludge, barley straw and alfalfa. They found that addition of sewage sludge and poultry manure resulted in higher soil organic C contents, barley straw resulted in highest improvement in aggregate stability and had the greatest effect on total polysaccharide content, infiltration rate and respiration rate. Interestingly, even the unamended plots increased in aggregation, which could have been due to the weekly irrigation schedule and the resulting positive influence of wetting and drying cycles.

Adediran et al. (2003) specifically investigated the effect of different organic amendments on crop yield. They used poultry litter with organic wastes, maize

residues, leaf litter, urban waste, weed biomass, and soybean residue and applied those to amaranthus and tomato crops. They found that for optimal crop yield, different amendments were required for different crops: while urban waste was best and soybean residue worst, for amaranthus production, maize and soybean residue proved to be best for tomato production.

2.5 Organic inputs and soil C

Maintenance of SOM and understanding of its dynamics is essential for developing sustainable cropping systems. SOM is important for many soil functions by providing energy, substrates, and biological diversity (Franzluebbers 2002). Adams (1973) and Hudson (1994) found that a 2% decrease in SOM increased bulk density by 0.1 Mg/m^3 or more. Water stability of soil aggregates is dependent on organic materials (e.g., polysaccharides, roots and fungal hyphae, persistant aromatic compounds; Tisdall and Oades 1982).

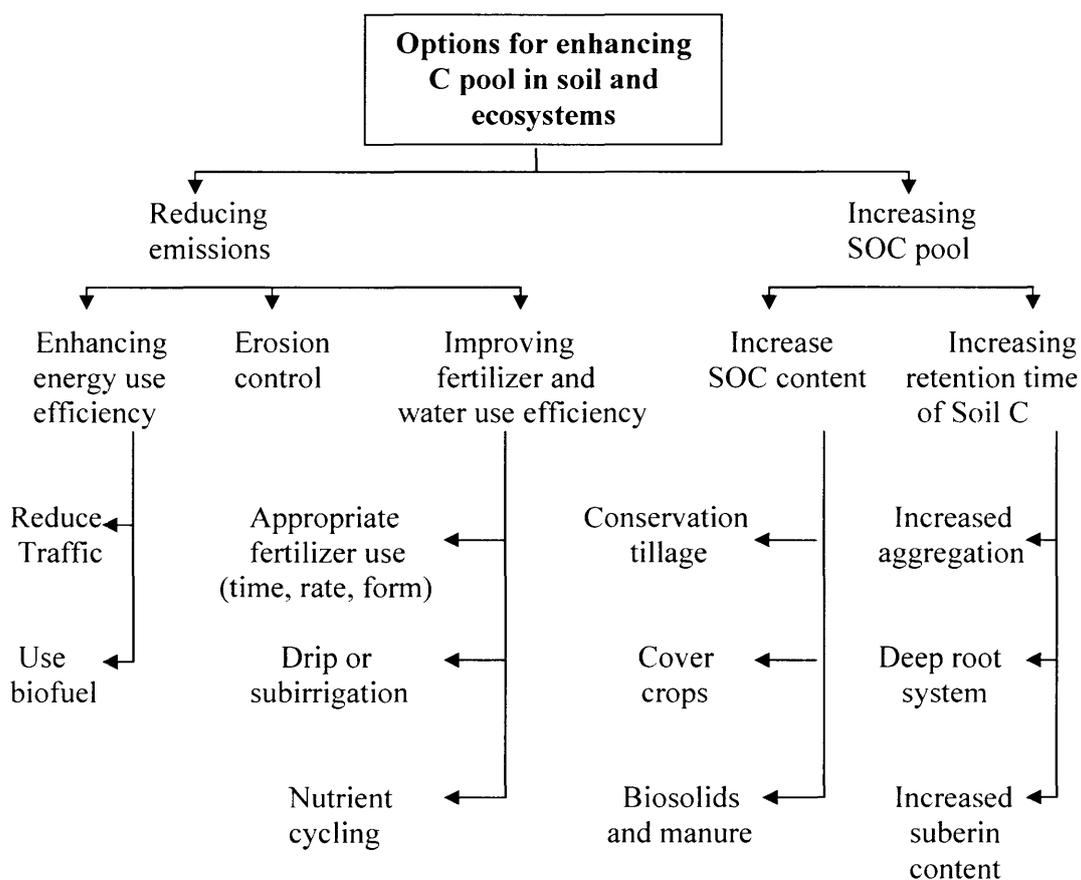Strategies exist to sequester soil organic C and options are summarized in Figure 2.1.

Figure 2.1 Technological options for enhancing C pool in soil and ecosystems (Lal 2002).

SOM affects soil compactibility, friability, soil water holding capacity, air and water infiltration, nutrient conservation, and soil permeability (Carter 2002). Improvement of aggregate stability results both from increased microbial activity utilizing carbohydrates and from the plant phenolics released during decomposition of structural components (e.g., lignin) (Martens 2000).

2.5.1 Significance of labile C pool

In their native state, soils are in equilibrium and have characteristic organic matter content. This equilibrium is disturbed when the soil is brought into cultivation due to the reduced accumulation of fresh organic materials and the accelerated breakdown of the already existing organic matter through cultivation (Nardi *et al.* 1996). Since SOM exists in a wide diversity of forms with considerable variability in decomposition rates (Bonde *et al.* 1992), the loss of the labile fraction could be greater than that assessed from the loss of total SOM upon cultivation (Dalal and Mayer 1986). Some workers have described labile C as the SOM components that can be extracted by water (Liang *et al.* 1995), and others have looked at the soil microbial biomass C (Degens and Sparling 1996) as the labile C. Other workers have recommended the use of 'light fraction' as the labile component of SOM (Magid *et al.* 1996). However, nuclear magnetic resonance (NMR) analysis has shown that light fraction has a chemical composition comparable to that of litter and plant material (Skjemstad *et al.* 1986) and may contain charcoal (Skjemstad *et al.* 1990), which has been identified as resistant C (Skjemstad *et al.* 1996).

Conteh and Blair (1998) studied the distribution and relative losses of soil organic C fractions from cracking clay soils and they concluded that the organic matter present in the cracking clay soils used for cotton production is highly decomposed, and most of it is concentrated in the microaggregates of the soil. According to their study, most of organic matter in the macroaggregates is dominated by undecomposed or partially decomposed plant materials, probably arising from leaf drops during defoliation which precedes harvesting of cotton.

The labile organic C provides a source of plant nutrients in soil due to its chemical composition and rapid turnover rate, and is responsible for temporary soil structural stability until it is further decomposed (Tisdall and Oades 1982). The lability of any organic C fraction could be due to either chemical composition (Skjemstad *et al.*

1996) or protection within the soil aggregates (Christensen 1996). Many workers believe that all SOM is readily decomposable and that only associations within the soil, at the molecular level, prevent decomposition from occurring (Piccolo 1996). The distribution of organic C among the labile and stable pool is affected by many factors including the type of tillage and length of cultivation (Tiessen and Stewart 1983; Camberdella and Elliot 1994). Labile organic C has also been reported to be present between and on the surfaces of microaggregates (Tisdall 1996) and has been suggested to be the major organic C pool depleted as a result of cultivation (Elliott 1986). The organic matter in smaller aggregates is more decomposed than organic matter in larger aggregates (Baldock *et al.* 1992). The light fraction of organic matter mainly consists of plant debris, both fresh and potentially decomposed, although other materials may be present in significant amounts. The light fraction of organic matter represents a transitory pool between fresh residues and humified, stable organic matter, and in arable systems this fraction is sensitive to changes in management practices (Janzen 1987). The higher light fraction content of the uncultivated sites was in accordance with the findings of Dalal and Mayer (1986), who observed losses of organic C from the light fraction following cultivation.

2.6 Organic inputs on K build-up

2.6.1 K chemistry in soil

Soil K is located mainly within minerals such as micas, feldspars and in interlayer spaces of phyllosilicates. Chemical weathering releases K in ionic form from minerals, which is then held in soil in several pools of differing availability (Johnston and Goulding 1990). Four pools of K are often considered viz., water-soluble, exchangeable, non-exchangeable and matrix K (Figure 2.2). The amounts of K in the first three pools tend to be in dynamic equilibrium and it is likely that matrix K is gradually released to the other pools by weathering.
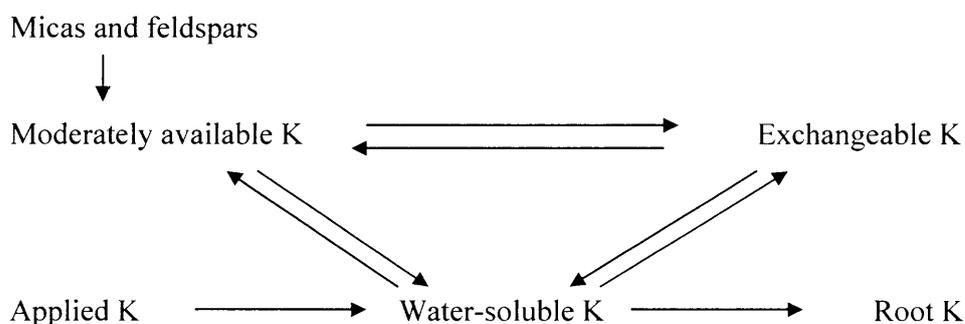
Micas and feldspars

Figure 2.2 Diagram showing equilibrium of K in soil

Water-soluble K and exchangeable K are readily available to the plants, but the proportion of these forms is relatively small, whereas non-exchangeable K is slowly available. K is released from non-exchangeable form when levels of exchangeable and solution K are decreased by crop removal (Singh and Goulding 1997).

Several physical and chemical soil properties and various pedogenic, biotic and weather conditions determine soil K dynamics in the field and influence plant available K. Texture, clay mineralogy, ionic environment and acidity have been recognized as factors affecting K fixation and release (Liu and Bates 1990; Stehouwer and Johnson 1991). The distribution of soil K among soil solution, water-soluble, exchangeable, and non-exchangeable forms is related to soil properties including surface area, mineralogy, surface charge density, and degree of interlayering of clay minerals (Shaviv *et al.* 1985). Clay minerals with high surface area and high lattice charge (e.g., illite, vermiculite) absorb K to a greater extent than the clay minerals of low charge (e.g., kaolinite). Bansal *et al.* (2002) studied the K dynamics in major benchmark soil series of India under long-term cropping. They observed that water-soluble K constituted from 0.19 to 0.44% in black (Vertosols) soils. They also reported that there was a significant decrease in exchangeable K content with time in Vertosols, but the non-exchangeable K content was higher than in red soils.

Water-soluble plus exchangeable K occupies a small part of total soil K. Accordingly the residual non-exchangeable K in the rhizosphere serves as an important source for plant K in some circumstances. For minerals, the processes of root-induced release of non-exchangeable K are well explained by two processes, i.e., exchange reaction, mineral dissolution or both. Hinsinger and Jaillard (1993) demonstrated that the release of interlayer K in phlogopite occurred in the rhizosphere of ryegrass (*Lolium perenne* L.) when the K concentration in the rhizosphere solution decreased below a threshold of about 80 μM. The release involves exchange of interlayer K by cations of high hydration energy and the consequent expansion of the interlayer space. Mineral dissolution by acids is another process to remove non-exchangeable K from minerals. Hinsinger *et al.* (1993) also reported that dissolution of phlogopite structure occurred in the rhizosphere of rape (*Brassica napus* L.) probably due to proton excretion by roots. The release of non-exchangeable K from feldspar has also been attributed to exudation of acids, from roots (Wang *et al.* 2000).

2.6.2 Effect of organic amendments on K availability

Availability of K to plants is to a large extent dependent on the rate and direction of the equilibrium reactions between solution and exchangeable/non-exchangeable phases of soil K. However, applied fertilizer K initially dissolves into the soil solution and is available to the plants, but after some time becomes unavailable when it is adsorbed onto clay particles of soil dominated with smectitic minerals. Dynamic equilibrium reactions among the forms of K also determine the release of K from non-exchangeable or slowly available forms to available forms under intensive cropping systems. Kinetics of non-exchangeable K release has been intensively studied by many workers (Martin and Sparks 1983; Dhillon and Dhillon 1990; 1992). The role of organic and inorganic manurial history of soils and organic matter content in K-exchange equilibria has also attracted some attention. But there are very few studies focused on non-exchangeable K release from soils receiving different amounts of K fertilizer and manures under long-term cropping (Rao *et al.* 1999). Rao *et al.* (1999) also reported a decline in non-exchangeable K and its release rate under a rice-rice cropping system without K application, and the application of a recommended dose of fertilizer plus 15 t/ha FYM maintained a larger K release rate in a long-term fertilizer experiment.

While evaluating the effect of long-term FYM application, Johnston and Addiscott (1971) observed that the quantity:intensity parameters of FYM-treated and control soils were identical. Goulding and Talibudeen (1984), however, observed a considerable decrease in K preference of FYM-treated soils. Sparks and Liebhardt (1982) also found lower K preference for surface (high organic matter) than subsoil (low organic matter) samples from the same soil profiles. In 1986, Poonia *et al.* studied the effect of FYM on K-Ca exchange and they reported that with the increasing levels of FYM, the standard free energy (The free-energy change for a reaction occurring under a set of standard conditions) of K-Ca exchange became gradually more negative, thereby there was a gradual increase in the preference for K in response to organic matter. The same trend of standard free energy of K-Ca exchange was reported by Mehta *et al.* (1983). Application of FYM increased K preference of soils, possibly due to an increase in the proportion of K-specific interlayer exchange sites (Poonia *et al.* 1986).

In their study on the effect of cattle manure application on K balance and release kinetics of non-exchangeable K, Singh *et al.* (2002) found that application of cattle not only arrested the decline of exchangeable and non-exchangeable K in solution, but also favoured their build-up. Thus, they concluded that the manuring resulted in increased easily-available K at the cost of non-exchangeable K.

As per the findings of Rao *et al.* (1999), long-term cropping, fertilization, and manuring influenced K release from the non-exchangeable fraction of the soil. Continuous cropping without K and with recommended optimum levels of N and P resulted in a greater decline in native K fertility of the soils. On the other hand, application of recommended fertilizer dose plus 15 t/ha FYM maintained higher K release from the non-exchangeable fraction of the soil. The effect of organic acids on releasing K and other structural cations is attributed to the dissociated H^+ ions and complexing organic ligands in the organic acid solutions. Song and Huang (1988) attributed the differences among organic acids in releasing K to differences among organic ligands and H^+ ions generated by organic acids and the complexing ability of the ligands.

Bernal *et al.* (1993) observed that the addition of pig slurry to a calcareous soil with a mainly illitic clay content caused a linear increase in exchangeable K content as rates of slurry application increased, whereas the soil with a low clay concentration showed a poor K^+ retention capacity. The low clay concentration soils have low exchange capacity, so the increase in the K concentration had very little effect on the exchangeable K^+ concentration.

2.7 Soil quality and cotton

2.7.1 Agronomic management of cotton

Cotton (*Gossypium* spp.) is a major crop in many countries. Cotton seed was first brought to Australia in the first fleet in the late 18th century and small trials were begun as settlement moved north from Sydney. Significant cotton was grown in Queensland during the American Civil war to fill a demand from cotton spinning mills in England at that time. Cotton production in Australia was unstable until about 1960 (Basinski 1963), when completion of the Keepit Dam provided a reliable water supply for cotton growing in the Namoi Valley of New South Wales (Anon 1981). About 75% of all cotton grown in Australia is now produced under irrigation in

northern New South Wales (Australian Bureau of Statistics 1993a). The three main cotton production areas in New South Wales are the Macquarie, Namoi and Gwydir Valleys which between them grew 176500 ha of cotton in 1992-93 (Australian Bureau of Statistics 1993b), representing 86.5% of total New South Wales production. Within 15 years, Australia was a significant cotton exporter and by the year 2000, 95% of the production was exported, particularly to Asia (Indonesia, Japan and Korea).

Cotton was often grown as a monoculture with intense land preparation (McGarry 1994). Research and grower experience showed the benefits of crop rotation (mostly with wheat) and minimum tillage (Constable et al. 1992). Benefits of rotation crops were claimed to include amelioration and maintenance of soil quality, reduced incidence of disease and weeds in the following cotton crop, and increased yield of cotton (Cooper 1993). The effect of different rotation crops on soil quality and cotton yield in Australian Vertosols have been addressed in few studies with the major focus being on soil structure and N balance (Hodgson and Chan 1984; Hulugalle and Cooper 1994; Hulugalle and Entwistle 1996; Rochester et al. 1998). Among the different rotation crops, wheat (Triticum aestivum L.) was the favoured rotation crop of most of the cotton growers in Australia (Hulugalle et al. 1999a). Soil structure has benefited from rotation with wheat, due to drying with a good fibrous root system and also from the nine month fallow between harvest of a wheat crop and sowing of a following cotton crop (Chan and Hodgson 1981; McGarry 1990). N fertility benefits have occurred with mineralization during the same fallow (Rochester et al. 1991).

Crop rotations have positive effects on soil properties related to their higher C inputs and diversity of residues returned to soils in comparison with continuous systems (Miller and Dick 1995; Entry et al. 1996; Moore et al. 2000). Short-term studies suggest that legumes sown after cotton can significantly increase root zone soil N (Hulugalle and Entwistle 1996; Rochester et al. 1998). Over the long-term, however, differences in soil N between a legume, such as field pea, and wheat are small due to presumed recycling of leached N by the latter (Hulugalle et al. 1999a). Cotton lint yield, quality, and cumulative gross margins were always greater with wheat as a rotation, whereas cotton-legume rotation systems resulted lower lint yield, fibre quality and cumulative gross margins and also the management constraints were greater with leguminous rotation crops than with wheat (Hulugalle et al. 2001). In

comparison with non-leguminous rotation crops, soil N and aggregate stability were also improved with leguminous rotation crops (Hulugalle and Cooper 1994; Hulugalle and Entwistle 1996; Rochester *et al.* 1998). Vetch (*Vicia sativa* L.), however, can reduce the intensity of bacterial black root rot of cotton but increases infestation of *Rhizoctonia solani* (Rothrock and Kirkpatrick 1990). In some situations, residues of leguminous rotation crops can have allelopathic effects on the following cotton crop (Hulugalle *et al.* 1998b).

Hulugalle *et al.* (1998a) reported that reduced crop growth rates can be a feature of long-fallow cotton. In comparison with continuous cotton, nutrient uptake, water uptake and cotton lint yield were better, whereas some of the soil quality indicators such as exchangeable Ca and magnesium (Mg), soil strength, plastic limit were lower with long-fallow cotton. The absence of any differences between cropping systems with respect to soil organic C suggested that long-fallow cotton (18 months), unlike long-fallow in cereal- and legume-based cropping systems (24-30 months), did not have any short-term detrimental effect on C sequestration. Continuous cotton sown after conventional tillage had lowest *in situ* macroporosity at field capacity, bulk density, soil organic C, exchangeable K and $EC_{1:5}$/exchangeable Na, and highest ESP (Hulugalle *et al.* 2005). The reverse occurred with cotton-wheat sown after minimum tillage.

As compared with the 1960s, current fertilizer programmes have replaced nutrients lost from continuous production of high-yielding crops, (Rochester *et al.* 2000a; 2000b; Rochester 2007). Current research and grower practice now include K and P fertilizer programmes. Heavy soil types and irrigation licence conditions aim to maintain runoff on-farm and retain fertilizer in the profile and on-farm respectively, rather than contamination of waterways or ground water. Irrigated cotton production in Australia has required high rates of N fertilization. This has produced average cotton yields of 1700 kg lint/ha, in many cases without the need for inputs of other nutrients such as P and K. Increasingly, growers are concerned with declining soil fertility levels, as cotton production is approaching 30 years in some area of Australia. Consequently, the use of P fertilizer by the Australian cotton growers has increased by 430% between 1980 and 2000 (C. Dowling, personal communication).

Cotton requires more intensive management than do many other crops. Furthermore, cotton fibre quality is also a concern since it can affect the value of cotton and

farmers' income. Numerous studies have indicated that soil fertility and physical properties are usually the most influential factors for cotton yield and fibre quality (Ping and Green 1999; Bradow and Davidonis 2000; Elms *et al.* 2001; Li *et al.* 2001). Soil properties, however, usually are correlated spatially and among them (inter-correlation) because of inherent soil forming processes (Mallarino *et al.* 1996; Mulla and McBratney 2000). Sand and clay contents, exchangeable Ca, Mg, nitrate, Olsen-P, relative elevation and slope were the important factors effecting lint yield and fibre quality of cotton (Ping *et al.* 2004).

For cotton, soils high in organic matter are usually preferred, because of the beneficial effects of organic matter on soil structure and greater capacity for net mineralization of native organic N. Following intensive cropping, organic matter levels typically drop and structural problems often develop, tending to increase the frequency and duration of temporary waterlogging. In addition to this effect, organic matter, may by its effect on redox potential, influence the severity of the waterlogging effect on plant growth and development. N losses through denitrification may also be affected by SOM levels either directly (Burford and Bremmer 1975) or indirectly through its effect on redox potential.

Soil moisture regimes and different organic matter levels also affect the cotton growth. Soomro and Waring (1987) observed that the soils behaved differently in their interaction with the water treatments. With the control water treatment, cotton grown on the high organic matter soil produced more dry matter and assimilated more N but had a lower N concentration and apparent recovery of urea-N than did cotton grown on the low organic matter soil. This was probably due to a greater mineralization of native organic N. Whereas, for the flooding treatments, the low organic matter soil produced more dry matter and assimilated more N, but had a lower N concentration and a higher apparent recovery of urea-N than did cotton grown on low organic matter soil.

In Australia, the major soil types used for cotton production are grey clays, with heavy grey and brown soils and black earths also being common (Ward 1999), and are classified as Typic Haplustert (Soil Survey Staff 2006) or Vertisol (in Australia, Vertosol (Isbell 1996a). These relatively fertile soils can have clay contents above 50%, have low organic matter and can be sodic at depth. Many of these Vertosols are able to store 200-250 mm of water in the soil profile (Dalgliesh and Foale 1998). The

presence of subsoil constraints such as salinity, sodicity, and chloride in many soils reduces effective rooting depth and increases crop lower limit thereby reducing the amount of water and nutrients that plants can obtain from the soil (Dang et al. 2006).

2.7.2 Distribution and properties of Vertosol

Australia has probably the largest occurrence of Vertosols in the world, with approximately 88 Mha accounting for 11% of the continent (Ahmad 1996). These soils are relatively uniform in colour and have consistently high chemical reactivity. Due to their high fertility and high water holding capacity many Vertosols are utilized for irrigated cotton production in Australia. Vertosols are clay textured soils with shrink-swell properties that exhibit cracking when dry (Isbell 1996a). These soils are imperfectly drained with slow permeability after wetting and crack closure. Vertosols characteristically have a narrow moisture range over which they are suitable for cultivation. Vertosols are structurally unstable, tend to slake easily on wetting and are easily compacted. However, they are very resilient and can redevelop a good structure after only a few cycles of wetting and drying (Wenke and Grant 1994). Degraded subsoil structure has also been found in Vertosols under both rainfed (Chan and Hodgson 1995) and irrigated farming systems (McGarry and Chan 1984; McKenzie et al. 1991). SOM and fertility commonly decline with cropping on Vertosols (Skjemstad et al. 1994; Chan 1997; Dalal and Chan 2001). Vertosols are also known as cracking clay soils because of their ability to shrink and form deep cracks when dry (Hubble 1984).

The magnitude and location of charge in swelling clay minerals is also important in relation to the adsorption and fixation of K and ammonium (NH_4^+) ions. Before the recognition of high-charge smectites, K^+ fixation was not considered to be a major problem in Vertosols (Coulombe et al. 1996). Chen et al. (1989) reported that an increase in K fertilizer rate caused NH_4^+ fixation and resulted in a decrease in rice yield in Vertosols dominated by high-charge smectites. In 2002, Singh and Hoffernan found that smectite was the dominant mineral in Vertosols and kaolinite and mica were common associated clay minerals in northern New South Wales, Australia. They also reported that smectites are rich in Fe and have high charge densities (0.55-0.67 mol(-)/($O_{10}(OH)_2$) and are possibly formed from the weathering of mica. The high charge smectite plays an important role in the release and fixation of K^+ and NH_4^+ ions.

Vertosols, which are dominated by smectitic clay minerals, swell on wetting and form deep cracks on drying (Probert *et al.* 1987). Soil crack formation is facilitated by rapidity and intensity of profile drying (Bauder and Brock 1992) and by increasing the number of wetting/drying cycles (Pillai and McGarry 1999). Cracking can improve soil structure, facilitate water infiltration and drainage, and help to distribute roots, SOM, and nutrients in the soil (Probert *et al.* 1987; Van Noordwijk *et al.* 1993; Skjemstad *et al.* 1998). This concept has been applied in managing the soil quality of Vertosols by sowing crop sequences which maximize soil drying and cracking, such as wheat or safflower after irrigated cotton (Cooper 1999).

Much of Australian cotton production is carried out in cracking clay soils. These soils typically have a 20-50 mm deep self mulching layer of polyhedral units overlying a zone of blocky peds to depths of 300-500 mm (Hulugalle and Cooper 1994). A common standard system of manipulating these soils for growing cotton has been outlined by Constable *et al.* (1992). Cracking in Vertosols also helps in leaching of nutrients, fertilizers, herbicides, and pesticides (Baer *et al.* 1995). Given their central role in these processes, physical and chemical properties of soil lining the cracks should differ significantly from those of nearby soil away from cracks. Sachan and Sharma (1980) observed that with respect to SOM fractions, soil near cracks tended to have higher proportions of humic acids relative to fulvic acids, and the reverse was true as distance from cracks increased.

Other studies have reported that SOM, roots, soil bacteria and protozoan populations, and soil processes such as mineralization occurred at higher rates closer to soils cracks than further away (Van Noordwijk *et al.* 1993). These responses might be caused by better aeration of soil proximate to soil cracks. Crop residues and organic matter falling in cracks in Vertosols may also result in organic C values adjacent to cracks being higher than that in bulk soil (Skjemstad *et al.* 1998).

The structure of Vertosols has been described in terms of fewer structural units and more massive subsoil between about 0.2 and 0.4 m depth (Hodgson and Chan 1984). Cracking clay soils contain little air when wet (Hodgson and Chan 1982). As a cracking clay soil dries, there is a marked increase in bulk density and soil strength.

In their native state, Vertosols generally have high SOM content (Spain *et al.* 1983) but this decreases significantly during cultivation (Dalal and Mayer 1986). In

Vertosol, the organic C content decreases with depth. But Knowles and Singh (2003) observed that there was unusual higher content of organic C in the deeper layers which reflected the self-mulching characteristics of Vertosol. The inorganic C was lower in the subsoil (0.60-0.90 m) than in the topsoil (0-0.30 m). The 0.30-0.60 m depth also had a relatively high proportion of total C as inorganic C. This might be due to the formation of carbonates at depth around 0.30 m or dissolution of carbonates from the top layer. Pedogenic carbonates generally accumulate in the subsoil near the depth of the average maximum annual percolation of soil water (Arkley 1963).

Despite the high chemical fertility, these Vertosols do have inherent problems. Soil maps and surveys of Australian soil suggest that the surface/and or the subsoil in a majority of cotton growing area in Australia are sodic (Northcote and Skene 1972; McKenzie 1998). Sodicity has a negative effect on the physical properties of soils. As a consequence of the process of dispersion, sodic soils generally have low hydraulic conductivities (Quirk and Schoefield 1955) and reduced infiltration rates (Kazman *et al.* 1983). The pre-irrigation soil water content which determines the degree of cracking, governed the infiltration process in swelling clay soils, but the extent to which soil and water salinity (EC) and sodicity affect the degree and pattern of cracking has not been documented (McKenzie *et al.* 1993). Yates and McGarity (1984) showed that aggregate stability of sodic Vertosols was positively correlated with the lime content. Increased sodicity, measured as ESP, increases the dispersion, although the critical level of ESP for dispersion is highly dependent on the electrolyte concentration of the soil solution (Shainberg *et al.* 1989). These physical properties of sodic soil have the potential to limit the growth and nutrient accumulation of cotton by reducing root growth and function.

In their study on sodicity index for cotton farming systems in irrigated and rainfed Vertosols, Hulugalle and Finlay (2003) reported that exchangeable Na content and electrolyte concentration predicted dispersion best in tilled Vertosols sown with cotton-based farming systems. None of the sodicity indices evaluated were related to dispersion when untilled. SOM and biological factors played a dominant role in aggregate stabilization in the latter. Other chemical and physical factors such as exchangeable Mg, carbonate concentrations, iron and aluminium oxide concentrations, and clay mineralogy can also influence aggregate stability (Rengasamy and Olsson 1991; Sumner 1993).

2.7.3 Cotton growth and Vertosol

Most Australian cotton is grown under irrigation in heavy clay soils with a high pH, where the availability of some nutrients can be reduced (Lindsay and Norvell 1969). Soil pH (1:5 aqueous extract) averaged 8.2 with only a small range of 0.7 pH units between the lowest and the highest side (Constable *et al.* 1988). According to the survey of 35 fields by Constable *et al.* (1988), the average total plant uptake of Fe was largest followed by manganese (Mn), boron (B), Zn and Cu. The high values for Mn at some of the survey sites might result from waterlogging and the associated anaerobic conditions; adverse effects of poor aeration in cotton have been demonstrated on these soils (Hodgson 1982). As dispersion directly affects hydraulic conductivity, and consequently crop water uptake and waterlogging potential in Vertosols (So and Aylmore 1993), threshold values for dispersion are needed for cotton. In a study of a soil quality indicators Hulugalle *et al.* (1999b), using published data, classified dispersion index values of > 25 g/100 g as very poor, 15-25 g/100 g as fair, 5-9 g/100 g as good, and < 5 g/100 g as very good for cotton growth in Vertosol.

Cotton yield is thought to be influenced by a variety of soil physical and chemical properties including, but not limited to, salinity [electrical conductivity of the saturation extract (EC_e)], texture, bulk density, leaching fractions, B, nitrate-N, and plant available water (Corwin *et al.* 2003). Similarly, the apparent soil electrical conductivity measurements of high clay content soils were influenced by the properties of EC_e, texture, water content, bulk density and were often correlated with soil B levels and leaching fractions (Rhoades *et al.* 1999a; 1999b; Corwin and Lesch 2003). Soil properties such as pH, soil P, SOM were well correlated with cotton fibre yield (Johnson *et al.* 2002).

Research in dryland Vertosols sown with cereal-based cropping systems (Loch and Coughlan 1984; Potter *et al.* 1997) suggest that by combining zero tillage, residue retention and N fertilizer, SOM decomposition and presumably CO_2 emission can be reduced. The soil environment in irrigated Vertosols sown with cotton is different to rainfed Vertosols and is characterized by waterlogging, frequent wetting/drying cycles, poor drainage, 'soil inversion' and mixing due to soil shrinkage and swelling (Hulugalle 2000). Furthermore, warm, wet conditions are a feature of irrigated cotton production systems as cotton is a summer growing crop and irrigation takes place during summer.

The structural degradation of Vertosol can cause a 30% depression in lint yield which is a major concern to the cotton growers (McKenzie *et al.* 1991). Extended periods of poor aeration are associated with Vertosol structural degradation (McGarry and Chan 1984); the duration of the period of poor aeration is correlated with reduced cotton lint yield (Hodgson and Chan 1982). Reduced N uptake has been implicated as the main mechanism through which waterlogging restricts cotton growth (Hodgson 1982). Deep tillage at an appropriate water content and gypsum application, have been shown to improve soil structure and cotton yield in poorly drained clays, apparently by increasing the non-limiting water range (Letey 1985), because of both improved aeration and lower soil strength (McKenzie *et al.* 1984).

2.8 Organic amendments and Vertosol

The possibility of significant benefits to cotton from organic wastes applied as soil amendments has been recognized for more than 65 years (Reynolds 1930). Using the organic wastes as soil amendment would help to recycle plant nutrients that would otherwise be lost and possibly enhance the productivity of land used for cotton production.

2.8.1 Incorporation of residues

When organic matter is added to soil, the microbial population will react and its reaction will follow a sequential order, as observed for organic matter decomposition (Rawitz *et al.* 1994). Addition of organic matter can cause very pronounced transient effects on soil physical conditions in addition to whatever medium and long-term effect it produces. So, timing of residue incorporation with reference to the soil water regime (e.g. scheduled irrigations or the expected start of a rainy season) might be a way of improving infiltration and minimizing runoff (Rawitz *et al.* 1994).

In cotton cropping systems, considerable amounts of all nutrients are returned to the soil during defoliation. This source of C and nutrients is readily decomposed by soil micro- and macro-organisms and represents a flush of nutrients into the system at a time when crop demand is low (Blair *et al.* 1995). The incorporation of crop residues in soil is widely believed to improve SOM levels, because crop residues contain considerable quantities of C and nutrients (Maskina *et al.* 1993), so their management can markedly affect the input-output balance of plant nutrients and SOM. Dalal (1989) found that the incorporation of crop residues in Vertosols resulted in higher

organic matter levels, but that was under cereal cropping systems. Rochester *et al.* (Rochester *et al.* 1997) reported that recovery of fertilizer N was reduced by 10% in the stubble-burnt plots compared with the stubble-incorporated plots. Also, cotton lint yield tended to decline in successive crops with stubble removal compared with stubble retention.

Decomposition of crop residues may release significant amounts of N and influence the availability of N by affecting mineralization-immobilization processes in the soil (Hood *et al.* 2000). The quantity of N derived from crop residues by a succeeding crop is highly variable and depends largely on residue characteristics and the synchronization between N release and crop N uptake. To develop cotton production practices which use N resources efficiently, it is necessary to understand the processes, interactions, and the pools that determine N availability to a cotton crop. Substantial losses of N generally occur in cotton-growing soils. Wright and Catchpole (1985) attributed the major cause of N loss in the irrigated clay soils to denitrification. Immobilization is often associated with the input of readily available C substrates which the soil microorganisms use as an energy source.

Rochester *et al.* (1993) observed that incorporation of cotton crop residue tended to reduce soil mineral N which indicated the low availability of C from cotton crop residues, which have low C/N ratio. The reduction of mineral N due to cotton residue incorporation resulted from a combination of low availability of the added crop residue N and the crop residue inducing immobilization of the fertilizer N. Rochester *et al.* (1991) also indicated that the input of fresh organic material may not necessarily produce a substantial decrease in mineral N. Factors such as the amount of crop residue material, its C/N ratio, degree of incorporation and edaphic conditions conducive to decomposition will all affect the mineralization/immobilization balance.

In their study on short-term variations of soil physical properties due to cotton residue decomposition, Hadas *et al.* (1994) indicated that addition of cotton residues to soil activated a heterogeneous microbial population which in turn affected soil physical properties. The amount of residues added to the soil and the C/N ratio determined the kind of microbial flush which promoted aggregation as indicated by an increase in aggregate diameter. After addition of smaller amounts, the flush in activity was typified by dense fungal mycelia, but with additional residue applications, there was little or no fungal activity or flush and an increase in aggregation was observed only

in later stages. This response was attributed to bacterial activity and secretions. Their results suggested that maintenance of a soil surface with a stabilized structure required addition of residues at a high frequency.

Management of cotton following a cereal cover crop can be different from that with a winter fallow. Allelopathic compounds in winter cereal residues can affect cotton stands. Hicks *et al.* (1989) reported that compounds arising from decaying wheat cover crop residues can stunt and kill young cotton seedlings. When residues are left on the soil surface, allelopathy is less a factor in causing stand reductions (White and Worsham 1989). However, if residues are trapped within the seed furrow, these residues may cause damage to young cotton roots. Cotton plant morphology may be influenced by the presence of winter cereal residues on the soil surface. Stevens *et al.* (1992) reported 11% fewer cotton squares on the lower fruiting nodes of cotton seeded directly into wheat stubble than of cotton grown with conventional tillage. Kasperbauer (1998) found a lower root:shoot ratio in 7 day old greenhouse-grown cotton plants that were grown over wheat straw compared with plants grown over the bare soil.

2.8.2 Application of manures

Improving the N nutrition of cotton could substantially increase plant growth and yield. N plays an important role in balancing vegetative and reproductive growth and lint quality. However, increasing the rate of N fertilization to overcome N limitations may not be successful because production problems occur when N supply exceeds the quantity required by the crop (Anderson 1975).

Cotton responds well to the application of N, as well as other nutrients, which makes poultry litter an attractive source of nutrients. Poultry litter is a relatively inexpensive source of both macronutrients (N, P, K, Ca, Mg and S) and micronutrients (Cu, Zn, Fe, Mn and B) and has been reported to increase soil organic C and enhance soil microbial activity (Nyakatawa *et al.* 2001). Cotton does not produce enough residues to supply the C necessary to increase SOM and improve soil tilth in the seed zone (Reeves 1997). In addition, cotton residues do not last long after harvest to protect the soil from erosion and reduce loss of soil moisture from evaporation. Therefore, without additional residues to supplement cotton residues, soils under no-tillage

cotton might develop a crust at the surface and a compacted layer in the top 5 to 10 cm (Reddy et al. 2004).

Application of poultry manure as a source of N and P has been shown to increase yield of crops such as corn and pastures (Sims 1986; Ma et al. 1999). Furthermore, studies have shown that poultry litter improved soil chemical properties compared with inorganic sources of N such as ammonium nitrate (Nyakatawa et al. 2001). Application of poultry litter at the rate of 100 kg N/ha gave lower or similar cotton lint yield compared with ammonium nitrate at the same rate (Reddy et al. 2004).

In a study conducted by Sistani et al. (2004) to investigate the use of poultry litter for cotton production, they specifically investigated the optimal quantity of N from the litter or the combination of litter N and supplemental inorganic N for optimum cotton yield production. They found that the use of poultry litter was an alternative fertilizer source to improve N use efficiency under no-till systems. Supplemental N had no impact on cotton yield, whereas soils receiving poultry litter at approximately 2.24 Mg/ha on an annual basis for a long time might not require additional N for optimal cotton yield production.

Ogunwole et al. (2005) compared the two organic materials (cowdung and poultry litter) with the conventional method (mineral fertilizer and synthetic pesticides) on soil physical properties and upland cotton productivity for two years on Savannah. Their study reflected that there was a possibility of the cowdung amendment favouring the process of macroaggregate formation thereby increasing the water holding capacity and plant available water content of the soils. Therefore, organic materials like cowdung and poultry litter had a high fertilizing value and may possess good potentials for improving the physical fertility of low activity clay soils.

Blaise et al. (2006) observed that long-term application of FYM resulted in a significant improvement in available P and Zn content of Vertosol. Application of FYM which not only supplies primary nutrients but is also a storehouse of secondary and micronutrients, improved SOM and consequently the physical properties. After doing the trend and stability analyses, they also pointed out that for achieving high level of yield stability of rainfed cotton, a combination of manure and fertilizers should be applied.

2.8.3 Use of other organic amendments

Substantial quantities of cotton processing wastes, which had been used as a cattle-feed supplement, is presently dumped and left to degrade in stockpiles around the gins. Cotton-producers have recognized this waste as a potential source of organic matter which could be used to improve soil conditions. Various studies have been focused on the use of earthworms in the stabilization of organic residues such as sewage sludge, animal wastes, crop residues and industrial refuse (Mitchell *et al.* 1977; Tomati *et al.* 1985). The vermicomposting process is a result of the combined action of the earthworms and of microflora living in earthworm intestines and in the growth medium (Grapelli *et al.* 1983). Earthworms accelerate composting by bioturbation and aeration, giving a final product less rich in N and organic matter, but enhanced with available nutrients to plants. During the vermicomposting process, the mineralization rate appeared to be accelerated. The vermicomposts were enriched in P, K and Na. Total N decreased but the decrease was partly balanced by a conversion into nitrate N (Grapelli *et al.* 1983).

Albanell *et al.* (1988) conducted a study to determine the variations in chemical composition of sheep manure alone and in combination with industrial cotton wastes during 12 weeks of vermicomposting. Their study showed that both type of vermicompost, compared with the materials from which they derived, resulted in lower pH, greater CEC, lower concentrations of soluble salts, organic matter and N and higher concentrations of total humic acids and mineral nutrients. They ultimately concluded that vermicomposting of cotton wastes was an alternative method for recovery of industrial wastes, at the same time producing a good, low-cost fertilizer. Edwards and Neuhauser (1988) reported the increased plant growth in potting-media enhanced with vermicompost derived from animal manures.

Options for controlling salinity and sodicity in irrigated cotton systems of Australia are commonly application of gypsum or lime in combination with a suitable leaching fraction (McKenzie 1998; Moss *et al.* 1999). Irrigation with treated sewage sludge effluent caused many changes in soil properties and most of them were not detrimental to soil quality of Vertosol or cotton yields (Hulugalle *et al.* 2002). Hulugalle *et al.* (2006) conducted a study of an eastern Australian Vertosol to evaluate if gypsum application had any effect on soil quality changes due to irrigating with treated sewage effluent. They found that irrigation with treated sewage effluent

increased exchangeable Na in all depths, and exchangeable Ca and K in the clayey-textured surface. Noticeable declines in exchangeable Ca and K, and soil organic C occurred in the coarser clay-loam textured depths > 0.6 m. Even though treated sewage effluent was rich in Na and chloride salts, salinization of the site did not occur.

2.9 Conclusions

This review has identified that SOM plays a vital role in maintaining soil quality. Since organic matter is the controlling factor in nutrient cycling, the amount and type of organic matter in a soil both reflects and controls soil productivity. The effects of organic amendments have been studied mostly in sandy and loamy, non-expansive soils in Australia and overseas. The benefits of using the amendments are claimed to include improved soil physical properties and better crop growth and quality. The disadvantages include increased disease incidence due to partially-composted organic manures and possibility of soil contamination with heavy metals.

There is a general paucity of information on the effects of organic waste products on quality of heavy clay soils, such as the cracking clays used for irrigated cotton production systems in Australia. The limited research data does suggest that organic C and K availability could be improved by organic amendments. At the same time, many cotton growers have been experimenting with organic amendments such as feedlot manure, vermicompost and composted cotton gin trash, even though the lack of information with respect to their use creates many difficulties. Furthermore, it is also unknown whether any economic benefits occur by using organic soil amendments, and how these benefits compare with those resulting from the application of cheaper inorganic amendments.

With the above background, therefore, it is essential to have the quantitative information on the economically feasible organic amendments and also the appropriate rates of application which can be used for the cotton production systems.

Chapter Three

General Materials and Methods

This chapter describes the materials and methods that are common to the experiments in other chapters. Experimental procedures that are specific to each chapter are detailed separately.

3.1 Analysis of organic amendments

Chemical analyses of the organic amendments were performed in the Plant Nutrition Laboratory of University of New England, Armidale, NSW. The concentration of P, S, K, Ca, Mg and Na in the amendments were determined using the Sealed Chamber Digestion method described by Anderson and Henderson (1986): 0.2 g of finely ground material was weighed into a pre-weighed empty 50 ml glass bottle and pre-digested with 2 ml of a 7:3 70% $HClO_4$/30% H_2O_2 solution for a minimum of two hours. One ml of H_2O_2 was added to the samples, which were then tightly capped and digested for 30 minutes at $80°C$ in a warming oven. On cooling, 1 ml of H_2O_2 was added to the bottle and the samples were digested for 1 hour. Two further additions of H_2O_2 were made, although the time of digestion was reduced to 30 minutes. The solutions were made to an approximate final volume of 25 ml with distilled deionized water and stood overnight. The final extract was determined gravimetrically, by weighing the completed digest and subtracting the weight of the empty bottle. The samples were filtered through a Whatman No. 1 filter paper into a glass vial and analysed using ICP-AES.

The nutrient concentration in the organic materials was weighed and volume corrected and expressed as a percentage (%) and $\mu g/g$.

The total C and N of the amendments were determined using Carlo Erba NA 1500 elemental analyzer connected to a Europa Tracemass Isotope Ratio Mass Spectrometer.

The vermicompost samples were analysed commercially. A dried sample (0.5 g) was taken and 5 ml HNO_3 added to the microwave vessel. Then the sample was heated up to $200°C$ and held at that temperature for 15 min. Once the sample was cooled then the volume was made up to 50 ml and then analysed by ICP-AES. The C and N was analysed by LECO combustion.

3.2 Soil analyses

Soil and plant chemical analyses have been done in the Plant Nutrition Laboratory at the University of New England, Armidale, NSW.

The soil samples were air dried and then passed through < 2 mm sieve and then analysed for the selected chemical properties.

3.2.1 Exchangeable cations and anions

3.2.1.1 Exchangeable cation concentration

The exchangeable cations viz., Al, Ca, Mg, Mn, K and Na were analysed using 0.1M NH_4Cl and 0.1M $BaCl_2$ (Gillman and Sumpter 1986).

40 ml of 0.1M NH_4Cl and 0.1M $BaCl_2$ solution were added to centrifuge tubes containing approximately 4 g of soil. The tubes were tumbled for 2 hours and then centrifuged @3000 rpm for 5 minutes. Then the supernatant was filtered through Whatman No.1 filter paper and exchangeable cation concentration was determined using ARL 3560B ICP-AES (Inductively Coupled Plasma – Atomic Emission Spectrometer). This method was used to estimate the exchangeable cation concentrations in all the samples.

3.2.1.2 Exchangeable anion concentration

3.2.1.2.1 Standardization of the method

Exchangeable anion viz., fluoride, bromide, nitrate, phosphate and sulphate concentrations were measured using synthetic ion-exchange resins. The method was standardized in the Plant Nutrition Laboratory at the University of New England, Armidale, NSW.

3.2.1.2.2 Brief review of the methods used by others

The use of synthetic ion-exchange resins to examine ion bioavailability in soil and sediment systems has attracted much attention over the years (Qian and Schoenau 2001). The first use of ion-exchange resin to measure nutrient availability dates from 1951 (Pratt) and 1955 (Amer *et al.*). Since then, ion exchange resins have been adopted to measure the plant available nutrient ions in soils and the rates at which they are released (Salomon and Smith 1957; Martin and Sparks 1983). The use of ion-exchange resin in sheet or membrane form has also attracted researchers'

interests, since the work of Saunders (1964), due to its simplicity in handling and use, and the ability to measure flux to an easily defined surface area (Qian *et al.* 1992; McLaughlin *et al.* 1993).

Curtin *et al.* (1987) studied the importance of exchangeable cations and resin sink characteristics in the release of soil P. Resin-extractable P was determined using different resin systems such as (i) anion exchange resin (AER) in HCO_3^- form, (ii) anion exchange form (OH^-) plus cation exchange resin (CER) in H^+ form and (iii) AER (OH^-) + CER (Na^+) form. The CER and AER were enclosed in separate bags (1 g resin/bag). 1 g soil sample was shaken for 16 hours with the appropriate resin in 35 ml distilled water. The bag containing AER was then washed free of soil with distilled water and shaken in a NaCl solution to displace P. Soil samples (1 g) were washed with 35 ml of 1M NaCl for 2 hours and then with 35 ml distilled water prior to extraction with the resin for 16 hours. They found that P extracted by AER-HCO_3 was well correlated with Olsen-P (r = 0.91**). They also concluded that HCO_3^- resin was more effective in desorbing P than either Cl^- or SO_4^- forms and was preferred in studies of labile-P (Sibbesen 1978). In their study, HCO_3^- resin was highly correlated (r = 0.99**) with P removed by sequential extraction with water. The combination CER-Na and AER-OH was more effective in removing P than AER- HCO_3, but less effective than the AER-OH/CER-H system.

Dhillon and Dhillon (1990) used the CER immersed in 1M HCl (resin:solution ratio 1:5) for 48 hours and washed with 0.1M HCl and deionized water until it was chloride free and then kept immersed in deionized water before use. 1 g soil was placed in a plastic centrifuge tube and wetted overnight with 0.5 ml of deionized water. Resin saturated with H^+, Ca^+, Na^+, or NH_4^+ (2 g) along with 10 ml of deionized water was also added to the centrifuge tube. The soil-resin-water mixture was gently and continuously agitated for intervals ranging from 0.25 to 720 hours at $25^\circ C$. At the end of each reaction period, a fine spray of 25 ml deionized water was used to rapidly (< 5 min) separate the resin from the soil on a 500 μm sieve. The resulting soil suspension was centrifuged at 15000 rpm for 10 min. They found that cumulative K released by different CER was in the order of: H^+-resin > Ca^+-resin > Na^+-resin > NH_4^+ -resin.

McLaughlin *et al.* (1993) developed a technique using cation/anion exchange resins for extraction of Ca, Mg, K, Mn, Al and P from the soil solution. According to their method, one strip of each CER and AER shaken with 1 g soil suspended in 20 or 40

ml water. The resin/soil/water mixture then shaken for 16 hours at 20°C. Soil was separated from resin by rinsing with distilled water and the elements were desorbed from the resin using 40 ml $BaCl_2/HCl$ (0.25M $BaCl_2$/0.1M HCl) in a batch shake system. Then to 40 ml of each two multi-element standard solutions, one of each CER and AER were added. The solution and resin membranes were the same as previously mentioned. After that ion concentration in $BaCl_2/HCl$ was determined by ICP, flamephotometry for K and colorimetry for P. They found that except Al, there was reasonably good correlation between the amount of elements extracted by ACEM and the conventional method.

In 1994, again McLaughlin *et al.* studied the comparison of cation/anion exchange resin methods for multi-element testing. They used four resin techniques: (i) 5 g soil shaken with AER and CER beads (5 g each) suspended in 100 ml deionized water, (ii) resin beads confined in polyester mesh bags (212 μm pore size), (iii) resin membranes (one strip each AER and CER) shaken with 1 g soil suspended in 20 ml water, (iv) as in (iii), but soil and resin were suspended in 40 ml water. The resin/soil/water mixture was then shaken on a reciprocating shaker (20 mm travel, 135 cycle/min) for 16 hours at 20°C. Soil was removed from the resin by rinsing with distilled water and the elements were desorbed from the resin by using dilute 0.1M HCl or $BaCl_2/HCl$ (0.25M $BaCl_2$/0.1M HCl) in a batch shaker system. They concluded that the ratio of resin + soil:solution had a significant impact on extraction of elements from soil. Lower resin + soil:solution ratios resulted in greater amounts of Ca, Mg and K and lesser amounts of Mn and Al, but P concentration was unaffected. So a constant and accurate resin + soil:solution ratio must be used for each extraction or errors will result. They also found that a mixed solution of 0.25M $BaCl_2$ and 0.1M HCl was found to be more successful in recovering cations from CER beads, whereas recovery of cations from CER membranes averaged 97% using a single extraction with $BaCl_2/HCl$. They ultimately suggested that all subsequent data for resin membranes refer to the technique using 40 ml volume of water for extraction.

Fernandes and Warren (1996) studied the comparison of resin beads and resin membranes for extracting soil phosphate. They used an aerobic incubation and suspension incubation method. In the 1st method, 20 g soil samples from each soil was transferred into 250 ml glass beakers. Two samples were moistened with P solution and the other two with water to give the required water and P contents. All

the samples were incubated for 11 hours at 30°C. After incubation, the soil in each beaker was mixed thoroughly and four sub-samples ($\sim$ 4 g) were taken from each beaker for P extraction, and then transferred into bottles. 100 ml of water was added and either a resin bag, or a resin strip were placed in each bottle and then the samples were shaken at 1 rpm. After 2 hours, the bags/strips were removed, the bottles shaken for 2nd period of 2 hours and fresh bags/strips again placed in the bottle and shaken for a third period of 20 hours. In the 2nd procedure, 4 g samples with 100 ml water were placed into bottles which were then shaken (1 rpm) at 50°C for 3 days with 0.5M HCl (50 ml). The amount of P extracted with bags was significantly higher than with strips. But the amounts of P recovered after P addition in both bags and strips were highly correlated (r = 0.95, $P < 0.01$, and r = 0.88, $P < 0.05$). The comparison between the aerobic and suspension incubation methods did not show consistent differences. But the strip method was one that can be recommended for the assessment of change in P status. They also recommended that a single extraction of 2 hours was quicker and more convenient than the cumulative total of three extractions.

Pierce and Morris (2004) compared nine different extraction techniques for measuring exchangeable cations in calcareous soil. The method they found had the highest exchangeable K is as follows. First the soil samples were dried at 40°C and ground to pass through 2 mm sieve prior to testing. The soluble salts of the sample were removed with 12.5 ml 60% w/v aqueous ethanol (665 ml of a solution containing 96% absolute ethanol, 2% water and 2% methanol) brought to 1 L with type-1 water. This was followed by a second wash with glycerol. Then the mixture was shaken on a reciprocating shaker for 5 min at 150-200 cycle/min. After that the samples were centrifuged and the washing solution discarded. The samples were then sequentially extracted with 3 additions of 25 ml and 1 addition of 15 ml extracting solution (approx. 60% v/v denatured ethanol and 1M NH_4Cl buffered to pH 8.5). Each extraction was done by using a reciprocating shaker for 5 min at 150-200 cycle/min. These extracts were combined with 10 ml 2M HNO_3 to bring the final volume to 100 ml and then analysed on ICP-AES following appropriate dilution.

3.2.1.2.3 Our laboratory method

After examining the previous methods and their advantages and disadvantages, we standardized the procedure as follows.

3 g soil was initially extracted with 35 ml of distilled water for 16 hours using the ion-exchange resins. Then the resins were removed and placed in a tube containing 20 ml of 0.5M HCl which was used as eluent. The tubes were shaken for 1 hour and the supernatant was analysed on IC (Ion Chromatography). This method was used to estimate the exchangeable cation concentrations in all the samples.

3.2.2 Other soil chemical analyses

Particulate organic matter (free light fraction and occluded fraction) was measured in three different times during the experiment; firstly, before application of the treatments, then during the active growth stage of cotton in both years. It was measured as a fraction of organic matter using the method described by Golchin *et al.* (1994) with our laboratory modifications. Instead of using Na polytungstate, 20 g air-dried samples were floated in 75 ml Na iodide at a density of 1.6 g/cm^3. After 48 hours, the soil was filtered through glass fibre filter paper and the light fractions, placed in petridishes, were put into the oven at $60°C$ for overnight and then weighed. Another 75 ml of Na iodide was added to the previous container and left to settle for another 48 hours. Then the samples were sonified at half power for 12 minutes 30 seconds (or, at full power for 6 minutes 15 seconds) in an ice bath. After sonification, the samples were left for 48 hours to settle, then filtered through glass fibre filter paper and the occluded fraction of organic matter was weighed after overnight oven-drying at $60°C$.

Soil pH (H_2O) and pH (0.01M $CaCl_2$) were measured by adding water or $CaCl_2$ to soil in a 1:5 soil:solution ratio. The samples were stirred, left for one hour and the pH measured measured in a 1:5 soil:water as well as a 1:5 soil:$CaCl_2$ extract while stirring using a combination electrode. Total C and total N were measured in Carlo Erba NA 1500 elemental analyzer connected to a Europa Tracermass Isotope Ratio Mass Spectrometer.

Electrical conductivity (EC) was read on a conductivity meter, after a 1:5 soil to water solution was stirred for one hour.

3.2.3 Soil microbial biomass and respiration

Soil was broken into small clumps with stones, large invertebrate animals and roots being removed. Around 80 g of these soils was added to a respirometer pot and three replicate pots were used for each soil sample. Soil moisture was adjusted to

approximately 75% field capacity (after prior determination of field capacity using filter paper method) where microbial respiration is optimal (K. King pers. comm.). Soil was then incubated at 20°C for 2 days before being placed in the water bath (20°C) in a respirometer. CO_2 evolution was measured in an electronic respirometry system (Respicond® respirometer, [2]Nordgren Innovations AB, Umea, Sweden). Firstly, average basal respiration rate was measured over 48 hours (mg CO_2/hr/100 g DM soil). Microbial biomass was determined by the Substrate-Induced Respiration method (Anderson and Domsch 1978) whereby the increase in respiration rate after the addition of glucose substrate is used to calculate the amount of microbial C in soil (mg microbial C/100 g DM soil).

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[2] Naming the company does not mean endorsement of their products