THE EFFECTS OF SALINITY AND SODICITY ON SOIL ORGANIC CARBON STOCKS AND FLUXES

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DECLARATION OF ORIGINALITY

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university. To the best of the author's knowledge, it contains no material previously published or written by another person, except where due reference is made in the text.

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ABSTRACT

Soil is the world's largest terrestrial carbon (C) sink, and is estimated to contain approximately 1600 Pg of carbon to a depth of one metre. The distribution of soil organic C (SOC) largely follows gradients similar to biomass accumulation, increasing with increasing precipitation and decreasing temperature. As a result, SOC levels are a function of inputs, dominated by plant litter contributions and rhizodeposition, and losses such as leaching, erosion and heterotrophic respiration. Therefore, changes in biomass inputs, or organic matter accumulation, will most likely also alter these levels in soils. Although the soil microbial biomass (SMB) only comprises 1-5% of soil organic matter (SOM), it is critical in organic matter decomposition and can provide an early indicator of SOM dynamics as a whole due to its faster turnover time, and hence, can be used to determine soil C dynamics under changing environmental conditions.

Approximately 932 million ha of land worldwide are degraded due to salinity and sodicity, usually coinciding with land available for agriculture, with salinity affecting 23% of arable land while saline-sodic soils affect a further 10%. Soils affected by salinity, that is, those soils high in soluble salts, are characterised by rising watertables and waterlogging of lower-lying areas in the landscape. Sodic soils are high in exchangeable sodium, and slake and disperse upon wetting to form massive hardsetting structures. Upon drying, sodic soils suffer from poor soil-water relations largely related to decreased permeability, low infiltration capacity and the formation of surface crusts. In these degraded areas, SOC levels are likely to be affected by declining vegetation health and hence, decreasing biomass inputs and concomitant lower levels of organic matter accumulation. Moreover, potential SOC losses can also be affected from dispersed aggregates due to sodicity and solubilisation of SOM due to salinity. However, few studies are available that unambiguously demonstrate the effect of increasing salinity and sodicity on C dynamics. This thesis describes a range of laboratory and field investigations on the effects of salinity and sodicity on SOC dynamics.

In this research, the effects of a range of salinity and sodicity levels on C dynamics were determined by subjecting a vegetated soil from Bevendale, New South Wales (NSW) to one of six treatments. A low, mid or high salinity solution (EC 0.5, 10 or 30 dS/m) combined with a low or high sodicity solution (SAR 1 or 30) in a factorial design was leached through a non-degraded soil in a controlled environment. Soil respiration and the SMB were measured over a 12-week experimental period. The greatest increases in SMB occurred in treatments of high-salinity high-sodicity, and high-salinity low-sodicity. This was attributed to solubilisation of SOM which provided additional substrate for decomposition for the microbial population. Thus,

as salinity and sodicity increase in the field, soil C is likely to be rapidly lost as a result of increased mineralisation.

Gypsum is the most commonly-used ameliorant in the rehabilitation of sodic and saline-sodic soils affected by adverse soil environmental conditions. When soils were sampled from two sodic profiles in salt-scalded areas at Bevendale and Young, SMB levels and soil respiration rates measured in the laboratory were found to be low in the sodic soil compared to normal non-degraded soils. When the sodic soils were treated with gypsum, there was no change in the SMB and respiration rates. The low levels of SMB and respiration rates were due to low SOC levels as a result of little or no C input into the soils of these highly degraded landscapes, as the high salinity and high sodicity levels have resulted in vegetation death. However, following the addition of organic material to the scalded soils, in the form of coarsely-ground kangaroo grass, SMB levels and respiration rates increased to levels greater than those found in the non-degraded soil. The addition of gypsum (with organic material) gave no additional increases in the SMB.

The level of SOC stocks in salt-scalded, vegetated and revegetated profiles was also determined, so that the amount of SOC lost due to salinisation and sodication, and the increase in SOC following revegetation relative to the amount of SOC in a vegetated profile could be ascertained. Results showed up to three times less SOC in salt-scalded profiles compared to vegetated profiles under native pasture, while revegetation of formerly scalded areas with introduced pasture displayed SOC levels comparable to those under native pasture to a depth of 30 cm. However, SOC stocks can be underestimated in saline and sodic landscapes by setting the lower boundary at 30 cm due to the presence of waterlogging, which commonly occurs at a depth greater than 30 cm in saline and sodic landscapes as a result of the presence of high or perched watertables. These results indicate that successful revegetation of scalded areas has the potential to accumulate SOC stocks similar to those found prior to degradation.

The experimental results from this project indicate that in salt-affected landscapes, initial increases in salinity and sodicity result in rapid C mineralisation. Biomass inputs also decrease due to declining vegetation health, followed by further losses as a result of leaching and erosion. The remaining native SOM is then mineralised, until very low SOC stocks remain. However, the C sequestration potential in these degraded areas is high, particularly if rehabilitation efforts are successful in reducing salinity and sodicity. Soil ecosystem functions can then be restored if organic material is available as C stock and for decomposition in the form of either added organic material or inputs from vegetation when these salt-affected landscapes are revegetated.

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LIST OF ACRONYMS AND ABBREVIATIONS

AGO	Australian Greenhouse Office
ANOVA	Analysis of variance
CEC	Cation exchange capacity
CFC	Critical flocculation concentration
C_{mic}:C_{org}	Microbial quotient
DOC	Dissolved organic carbon
EC	Electrical conductivity
ESP	Exchangeable sodium percentage
FAO	Food and Agriculture Organisation
IRGA	Infra-red gas analyser
LSD	Least significant difference
NPP	Net primary productivity
NSW	New South Wales
P_{CO2}	Partial pressure of CO_2
POC	Particulate organic carbon
POM	Particulate organic matter
qCO_2	Specific respiration rate
REML	Residual maximum likelihood
SAR	Sodium adsorption ratio
SED	Standard error of difference
SIC	Soil inorganic carbon
SMB	Soil microbial biomass
SOC	Soil organic carbon
SOM	Soil organic matter
UN	United Nations
WA	Western Australia

CHAPTER 1: INTRODUCTION

1.1 Background

An understanding of the effects of salinity and sodicity on soil carbon (C) stocks and fluxes is critical in environmental management. Many soils in the Australian environment are naturally saline and sodic, largely due to extensive weathering of the regolith and deposition processes in conjunction with the arid climates that have occurred in the past. Saline soils, caused by high levels of soluble salts, have been estimated to cover over 17 million ha within Australia (Szabolcs 1989), while soils containing a sodic layer high in exchangeable sodium (Na), affect approximately 190-300 million ha of Australian soils (Northcote and Skene 1972). Because of their age and the extent of weathering that has taken place in the past, Australian soils are also relatively infertile. Levels of soil fertility are often strongly influenced by soil organic carbon (SOC), with low organic matter contents due to low biomass inputs and rapid turnover. It is estimated that 70 % of Australian soils display SOC levels of less than 1 % (Spain *et al.* 1983). Levels of SOC are largely a function of net primary productivity (NPP), or biomass accumulation, and therefore follow similar gradients to that of plant growth which is constrained by temperature and precipitation. A large part of Australia has an arid climate, and as a result of low biomass inputs, soils generally display low levels of soil organic matter (SOM). Land management practices which alter plant growth, including many agricultural practices, also have the potential to further alter soil C stocks and fluxes. Of particular importance are past and current land management practices that have resulted in an increase in saline and sodic soils.

The broadscale clearing of native vegetation since European settlement, its replacement with crops and pasture, and subsequent land use practices have resulted in increased rates and quantities of groundwater recharge. Prior to European settlement, the presence of deep-rooted perennial vegetation maintained hydrological equilibria in the landscape (Hatton *et al.* 2003). However, since settlement, large areas of native vegetation have been cleared primarily for agricultural purposes, and have been replaced with shallow-rooted annual crops and pastures. This change in vegetation has resulted in decreased transpiration, which allows more water to infiltrate through the soil profile to the groundwater, thus causing the water table to rise. As the water table rises, soluble salts

are mobilised and discharged into lower lying areas in the landscape (Burch 1986). Where the water table is within one or two metres of the soil surface, plant vigour is decreased as transpiration processes, evaporation and capillary action can draw saline water into the root zone of plants. This overall process of altered hydrology in the landscape has resulted in a redistribution of the salt stores in the soil profile, causing salinisation of land and water.

While soil salinity is the result of high levels of soluble salts, soil sodicity is caused by high levels of exchangeable Na adsorbed on the surfaces of clay particles. Increasing sodicity in soils causes aggregates to disperse. As a result, those soils that are sodic are increasingly susceptible to water erosion. The dispersed clay particles also fill in the soil pores to form a massive structure, causing decreased infiltration and permeability to water, and the formation of surface crusts and seals. Within Australia, a soil is considered sodic when the exchangeable sodium percentage (ESP) exceeds 6 % (Isbell 1996). This value is lower in Australia than those recorded in other parts of the world due to the low electrolyte levels of Australian irrigation waters and soil solution systems, and the dominance of rainfed agriculture. The lower electrolyte levels in Australian systems result in a higher tendency for soils to disperse for a given ESP. Amelioration of saline areas in other parts of the world can be effected with the use of high quality irrigation water or rainfall, which leaches soluble salts in the profile. However, saline areas in Australia are dominated by Na salts, namely NaCl, $NaHCO_3$ and $NaCO_3$, which may result in a soil that is high in exchangeable Na^+ and hence sodic.

Since the amount of C present in the soil is dependent on C inputs and losses, increasing salinity and sodicity levels have the potential to decrease C inputs into the soil through their effects on vegetation and impact on C dynamics. Not only can increasing salinity and sodicity directly impact upon plant vigour through changes in osmotic potential, ion toxicities and ion deficiencies, indirect effects on vegetation can result from altered soil conditions such as increased dispersion and decreased permeability. Changes in salinity and sodicity affect soil physical and chemical properties, which subsequently alter nutrient cycles, aggregation and biotic activity. Erosion also has the potential to be increased, which affects C stocks in a catchment. Thus, there is a clear linkage between land management practices, through their effects on salinity in particular from the clearing of native vegetation as described above, and their potential to alter soil carbon stocks and fluxes in the landscape. Despite the large area affected by salinity and

sodicity, both in Australia and globally, data on the mechanism and magnitude of changes in soil C stocks in these degraded environments is sparse.

Few data exist on C cycling in degraded landscapes, particularly those affected by salinity and sodicity. The Australian Greenhouse Office (AGO) currently classifies these areas as abandoned agricultural land, but also recognises that it is an issue of national significance (AGO 1999). C cycling in saline and sodic landscapes is complicated by waterlogged conditions and the common occurrence of highly alkaline subsoils caused by the presence of carbonates and bicarbonates. While alkalinity and its effects on C dynamics are important issues, it is beyond the scope of this project.

SOC displays a continuum of decomposition and turnover times. As a result, it is frequently partitioned into discrete pools according to the length of time required for turnover, and usually varies between two and five pools (Jenkinson and Raynor 1977). These pools usually consist of an active pool, with a turnover time of weeks to months, a slow pool which exhibits a turnover time of decades, and a passive pool which requires millennia to turn over. The active C pool is comprised of the soil microbial biomass (SMB), its metabolic products and the dead biomass, and has the potential to act as an early indicator of soil C dynamics due to its faster turnover time compared to the SOC pool as a whole. While it only comprises a small portion of the total SOM (1-5 %; Killham 1994), it can be used to determine changes in soil C dynamics under changing environmental conditions prior to detection in the total SOC pool. Its importance lies in the function of the SMB, as all organic material passes through this pool for decomposition or transformation.

While the amount of C in the soil is a function of factors such as soil temperature, moisture and texture, long term field trials have established that land use and land use change have a direct effect on soil C contents and mineralisation (eg. Dalal and Mayer 1986). The distribution of organic C into these discrete pools, particularly the faster cycling pools, is influenced by soil management factors such as land use, irrigation, crop rotation, tillage and fertiliser application. Therefore, any changes in management regime, including both degradation and rehabilitation processes, have the potential to affect the carbon flux and the amount and proportion stored in a particular pool. As the areal extent of soils affected by salinity and sodicity increases, SOC stocks and decomposition processes will also be altered. However, the extent to which C stocks

and processes will be altered by salinity and sodicity is not known. This thesis will address these knowledge gaps in relation to landscapes which have become degraded by salinity and sodicity impacting on C stocks and dynamics.

1.2 Aims and Objectives

The overall aim of this project is to determine how soil C stocks and turnover are affected by land degradation through increasing salinity and sodicity, and the extent of hysteresis these systems exhibit upon rehabilitation. This project has the following objectives:

- Quantification of the effects of different levels of salinity and/or sodicity on carbon stocks and fluxes along a salinity and sodicity gradient under controlled conditions in the laboratory,
- Determination of the behaviour of the labile carbon pool in a saline-sodic soil, and with gypsum amendment over a 12-week period in controlled conditions,
- Determination of how decomposition is affected in saline-sodic soils with and without gypsum amendment following the addition of organic material in controlled conditions, and
- Quantification of soil C stocks in salt-affected scalds, eroded scalds, revegetated and unaffected vegetated profiles.

1.3 Thesis Outline

This thesis will be presented according to the structure shown in Table 1.1.

Table 1.1 Thesis structure

Chapter	Description
Chapter 1: Introduction	Introduction to thesis; aims and objectives
Chapter 2: Literature Review	Review of background literature on salinity, sodicity and soil carbon dynamics
Chapter 3: The effects on the soil microbial biomass and soil respiration following leaching with salt solutions	Quantification of the effects of increasing salinity and/or sodicity levels on carbon stocks and fluxes along a salinity and sodicity gradient under controlled conditions in the laboratory
Chapter 4: Soil microbial biomass and soil respiration rates in salt-scalded profiles	Determination of the behaviour of the labile carbon pool in a saline-sodic soil, and with gypsum amendment over a 12-week period in controlled conditions
Chapter 5: Decomposition of added organic material in salt-affected soils	Determination of how decomposition is affected in saline-sodic soils with and without gypsum amendment following the addition of organic material in controlled conditions
Chapter 6: Carbon stocks in saline, saline-sodic and sodic landscapes	Quantification of SOC stocks in salt-affected scalds and vegetated soil profiles
Chapter 7: General discussion	Linking of results related to processes found under controlled conditions in the laboratory to C stocks found in the field
Chapter 8: Summary and conclusions	Summary and conclusions
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CHAPTER 2: LITERATURE REVIEW

2.1 Introduction

Worldwide, approximately 932 million ha are estimated to be salt affected, with salinity affecting 23 % of arable land, and saline-sodic soils affecting a further 10 % (Szabolcs 1989). In Australia, it is estimated that salinity affects an estimated 17 million ha while sodicity affects approximately 340 million ha of land (Szabolcs 1989). Salinisation and sodication of soils are serious land degradation issues in Australia. Sodicity affects soil physical properties, causing a decline in soil structure due to increased swelling, dispersion and slaking upon wetting, and increased crusting and hardsetting on drying, with a concomitant decline in permeability, infiltration and hydraulic conductivity (Table 2.1). Many areas also exhibit severe erosion, particularly gully erosion, as well as an increase in waterlogging and altered hydrologic processes. Salinity affects soil chemical properties through the presence of high soluble salt concentrations. This adversely affects soil biota and vegetation by altering the osmotic and matric potential of the soil solution. Saline and sodic soils also affect plant growth by inducing ion deficiencies in certain micronutrients and nutrient toxicities in others.

Table 2.1 United Nations (UN) Food and Agriculture Organisation (FAO) classification of saline and sodic soils.

	EC_e (dS/m)	ESP (%)	Typical pH	Structure
Saline	> 4	< 15	< 8.5	Good
Sodic	< 4	> 15	> 9.0	Poor
Saline-Sodic	> 4	> 15	< 8.5	Fair to good

Notes: EC_e is the EC of a saturated paste extract; ESP is the exchangeable sodium percentage. In 1:5 soil:water extracts, the EC of a saline soil is > 1.5 dS/m (Murphy and Eldridge 1998)
Source: van Lynden *et al*. (2004)

C dynamics as influenced by salt-related degradation will only increase in significance in the future, as the extent of salinisation and sodification is projected to increase by up to 40 % in some dryland areas (NLWRA 2001). However, the issue of C turnover as affected by salinity and sodicity is complicated by processes associated with salt-affected soils, such as waterlogging and the presence of inorganic C, usually in the form of calcium carbonate and sodium bicarbonate. Peck and Hatton (2003) predict that, in general, most of southern Australia which lies in the annual rainfall range of 250-800 mm with deeply weathered regolith has the potential for salinisation following clearing.

Whilst the deleterious effects of soil salinity and sodicity, termed collectively as salt-affected soils, have been extensively studied in the past, particularly in regards to soil structure and vegetation health, the effects on C dynamics with respect to emissions, or losses from soils, and stocks, is not as well documented. This is particularly pertinent, given the large area affected by salinity and sodicity, usually coinciding with agricultural areas, where C stocks are likely to be directly related to decreased plant inputs due to low biomass production and hence, low SOM accumulation. This review will present an overview of studies in salinity and sodicity, their relationship with SOC, and identify where knowledge gaps exist.

2.2 Salt-affected soils

Many Australian soils are naturally saline, as discussed in Section 2.1, and are found predominantly in arid to subhumid regions, where they are characterised by high levels of soluble salts and/or exchangeable Na. The distribution of these soils generally follows climatic gradients, dominant in parts of Australia where the average annual rainfall lies within the 250-600 mm range (Northcote and Skene 1972). The issue of salinity and its subsequent impacts on plant health have received much attention in recent years as a result of anthropogenic-related changes in landscape hydrology and subsequent redistribution of salts. These activities are largely related to the widespread removal of deep rooted perennial native vegetation and its replacement with shallow rooted annual crops and pastures. This process causes an increase in the amount of water infiltrating through the soil profile, which mobilises and transports soluble salts (Burch 1986). Where the water rises to within two metres of the soil surface, evapotranspiration processes and capillarity cause salts to rise, and hence affect the root zones of plants.

High levels of exchangeable Na are commonly present in Australian soils, where it can impact on soil physical and chemical properties. A soil is defined as sodic where the exchangeable sodium percentage (ESP) $\geq$ 6% (Isbell 1996). The ESP is defined according to the following equation:

$$ESP = (Na_{exch}/CEC) * 100 \qquad \textbf{Equation 2.1}$$

where Na_{exch} is the amount of exchangeable Na^+ and CEC is the cation exchange capacity, both expressed in cmol/kg soil.

The ESP of a soil describes the level of exchangeable Na in the soil relative to the other exchangeable cations present. The sodium adsorption ratio (SAR) is also frequently used to describe the sodicity level of the irrigation water or soil solution, reflecting the balance between Na^+ and Ca^{2+} and Mg^{2+}, where:

$$SAR = [Na^+]/0.5\ [Ca^{2+} + Mg^{2+}]^{1/2} \qquad \textbf{Equation 2.2}$$

and Na^+, Ca^{2+} and Mg^{2+} are in meq/L

In general, sodicity has received comparatively less attention than issues associated with salinity, as it is not as closely linked to anthropogenic activities.

The deleterious effects of increasing salinity and sodicity on soil physical and chemical properties and the processes involved have been extensively studied and reviewed (eg. Levy 2000; Levy *et al.* 1998; Qadir and Schubert 2002; Rengasamy and Olsson 1991; Rengasamy and Sumner 1998), as have the remediation measures available for the amelioration of saline and sodic soils (eg. Keren 1996).

2.2.1 Saline Landscapes and Salinisation

While Australian soils are naturally saline (Hubble *et al.* 1983), anthropogenically-induced salinity occurs in dryland and irrigated agricultural areas of Australia. Prior to salinisation, salt stores generally occurred below the major rooting zones of native vegetation, and were largely immobile before land clearing (Hatton *et al.* 2003). It is currently accepted that salinisation of land and water has occurred due to the extensive clearing of native perennial vegetation for annual crops and pastures. Following clearing, the recharge rate of groundwaters can increase by up to 20 times the rates prior to clearing, causing new aquifers to develop in the unsaturated zone, which allows salt stores to be mobilised (Salama *et al.* 1993a; b). Evapotranspiration is reduced following

removal of native vegetation, with an excess amount of water available for runoff and recharge.

Landscapes where saline soils occur are characterised by their heterogeneity, with the expression of salinity dependent on a number factors including geological structures, groundwater hydrology and geomorphic controls. Ephemeral perched aquifers can form on top of a clay B horizon in areas where duplex soils occur, allowing water and salts to be transferred laterally, possibly to non-saline areas (Peck 1978). Where permeability is decreased as a result of the texture change in the profile, water can flow laterally, usually in the form of subsurface flow. Discharge occurs in lower lying areas or where a break of slope occurs, causing waterlogging (Hatton *et al.* 2002; McFarlane and George 1992). Extensive waterlogging can occur particularly where an existing perched aquifer responds rapidly to rainfall events (Cox and McFarlane 1995; Eastham *et al.* 2000). Perched aquifers, which may be of lower salinity, can also act as a major recharge mechanism for deeper aquifers, which are often highly saline (George and Conacher 1993). In general, salinity increases along groundwater flow paths from catchment divides and areas of recharge, to valley floors and discharge areas (Salama *et al.* 1999). In some areas, such as the Dundas Tableland in Victoria, the clearing of native vegetation has not appreciably affected groundwater recharge rates but resulted in an increase in duration of seasonal waterlogging of low lying areas due to increased subsurface water flows (Dalhous *et al.* 2000).

Transient salinity occurs extensively in areas dominated by sodic subsoils, and is used to describe the temporal and spatial variation of salt accumulation in the root zone not influenced by groundwater processes and rising saline water-tables. However, it has received little attention compared to dryland salinity (Rengasamy 2002). This process, affecting vegetation health, is related to the increase in the concentration of salts in the root zone of plants, as water is removed from the soil profile due to evapotranspiration, causing an increase in the soluble salts. As a result, salinity fluctuates with depth and changes in concentration, and affects plant growth according to seasonality and rainfall (Rengasamy *et al.* 2003). In general, duplex soils and those soils with a sodic subsoil have a high potential for transient salinity.

Where the regolith and groundwater hydrology have been significantly altered as a result of clearing of vegetation, large salt stores have the potential to be mobilised, accentuating the salinisation of soil and water. The distribution of paleodrainage systems, or relict channels can play a role in subsurface water flow and mobilisation of salts as these channels contain higher levels of salts than the surrounding landscape. Relict channels usually occur along or within the existing drainage network, and are usually reactivated following the clearing of native vegetation, which subsequently causes drainage to develop in topographic lows where these channels exist (Salama *et al.* 1993a). Salinity levels in relict channels can be higher than in aquifers, with salinity increasing in the direction of flow (McFarlane and Williamson 2002). Where flow along relict channels is impeded by geological structures such as dykes, veins and basement highs, these barriers cause groundwater upstream from the barrier to increase in height, resulting in salt mobilisation. Precipitation of minerals can occur in areas of groundwater discharge or where the water table is rising, as the mineralised porewater at or near the ground surface continually evaporates. Salt fluxes are generally greatest in these areas which are, thus, the most active sites of soil salinisation (Salama *et al.* 1999).

Salinisation of landscapes is also characterised by the time lag between time of clearing, increased recharge, and the expression of salinity caused by increased water table levels. The time lag between clearing and the development of salinity is dependent on certain characteristics within a catchment, such as the thickness of the unsaturated zone, the location of the recharge area in relation to discharge areas and the distance between them, local geomorphology and the presence of fractured bedrock. The response times following clearing of vegetation are largely related to the groundwater flow systems that exist at a catchment scale. For example, a study by Allison *et al*. (1990) concluded that salinity will continue to increase over the next 200 years in the Western Murrray Basin despite the region having largely been cleared for more than 40 years, due to the slow response of a large regional groundwater flow system. In contrast, faster response times have been identified in intermediate and local groundwater flow systems. For example, in the Cuballing catchment in Western Australia (WA), new unconfined and semi-confined aquifer systems were formed following clearing with the first signs of salinity noticed 20 years after clearing (Salama *et al.* 1993b).

Much conjecture surrounds the origins of salts in Australia, which are dominated by sodium chloride (NaCl). Sources have been attributed to cyclic salts deposited from rainwater over time periods of millenia (Bettenay *et al.* 1964; Herczeg *et al.* 2001), connate salts from marine sediments (Salama *et al.* 1999), atmospheric accessions of a terrestrial origin (Acworth *et al.* 1997; Acworth and Jankowski 2001), and mineral weathering (Gunn and Richardson 1979). Atmospheric accessions can be of oceanic or terrestrial origin, with the influence of oceanic salts on salt composition in rainwater decreasing with increasing distance from the coast, until terrestrial sources dominate (Hingston and Gailitis 1976). Within New South Wales (NSW), the occurrence of dryland salinity usually coincides with a number of broadscale land features, including the presence of Ordovician age metasediments with yellow and red texture contrast soils, native vegetation clearance from high parts in the catchment in grazing lands, and rolling hill and tableland country (Bradd *et al.* 1997). However, there also appears to be a direct positive correlation between winter dominant rainfall and the large number of dryland salinity sites, as high evaporation in the summer reduces potential for groundwater recharge in summer-dominant rainfall areas (Bradd *et al.* 1997).

Whilst dryland salinity has received more attention in salinity related research (eg. NLWRA 2001), effects related to irrigation salinity are more concentrated but less widespread in terms of areal extent. Increased recharge occurs following clearing in conjunction with recharge from the applied water. The use of saline and saline-sodic water of marginal quality for irrigation has greatly increased in recent years due to an increasing shortage of high quality water resources. Water used for irrigation can include groundwater, drainage water or treated wastewater. The chemical composition of irrigation water has the potential to affect the concentration of soluble salts in the soil solution due to precipitation or dissolution. Under irrigation, soil solution chemistry changes according to irrigation cycles, altering pH, redox potential and availability of ions for plant growth (Boivin *et al.* 2002). Salts can subsequently accumulate where the irrigation water used is saline (Gardner 2004), with the movement of salts occurring vertically and laterally. Salt movement is further complicated by water application patterns and crop rotation (Herrero and Perez-Coveta 2005). In the shorter term, it has been found that no discernible difference in pasture production can be observed when using saline irrigation water but in the longer term, soils can become moderately to highly saline and sodic, resulting in a significant reduction in pasture production and

quality (Rogers 2002). The reuse of saline-sodic groundwater for irrigation leads to accumulation of Na in the soil profile, and can result in the formation of sodic soils, particularly where water-tables are shallow and leaching restricted (Bethune and Batey 2002); this is described in more detail in section 2.2.2.

2.2.2 Sodic Soils: Processes and Properties

A considerable proportion of soils under agriculture within Australia suffer from constraints related to sodic subsoils, as described in Section 2.1. Sodic soils generally have poor physical properties, affecting water infiltration and permeability. These soils can develop naturally from saline soils, with their development related to the underlying parent material, climatic change, or as a result of human activities, such as the leaching of salts from a saline soil, as shown in Figure 2.1 and described in more detail in Chartres (1993). Briefly, salts in Australian soils are dominated by Na salts. The formation of a sodic soil from an initial saline soil has the potential to occur under irrigation or changing climatic conditions. Soluble salts are leached out of the upper layers of the soil profile, with clays translocated into the B horizon. Upon further leaching, high levels of exchangeable Na results, with low concentrations of soluble salts remaining in the soil profile. While effects due to salinity are largely related to altered soil chemical properties and osmotic potential affecting plant growth, effects related to sodicity are mainly due to influences on soil physical properties.

The sodicity of a soil is characterised by its ESP or SAR, previously described by Equations 2.1 and 2.2, respectively. However, the behaviour of a sodic soil is largely linked to both the level of sodicity and the electrolyte content of the soil solution or the applied water. Where the applied water has a low electrolyte concentration, physical effects include increased swelling and dispersion, and reductions in hydraulic conductivity and infiltration rates (Rengasamy *et al.* 1984). Slaking occurs upon wetting, causing larger aggregates to break into smaller aggregates as a result of swelling and air entrapment. On further wetting, dispersion occurs, causing clay particles to diffuse out of the aggregates. Spontaneous dispersion can occur when the EC of the applied water is low and the soil is highly sodic, as bridging between clay particles is dominated by Na. The uptake of water by Na^+ causes the interparticle distance to continuously increase and the individual clay particles to disperse (Rengasamy and Sumner 1998).

Where the applied water has a high electrolyte concentration, swelling and dispersion are limited, while hydraulic conductivity and infiltration rates are maintained. This is due the maintenance of soil structure, as the high electrolyte concentration of the soil solution results in flocculation, described in more detail below

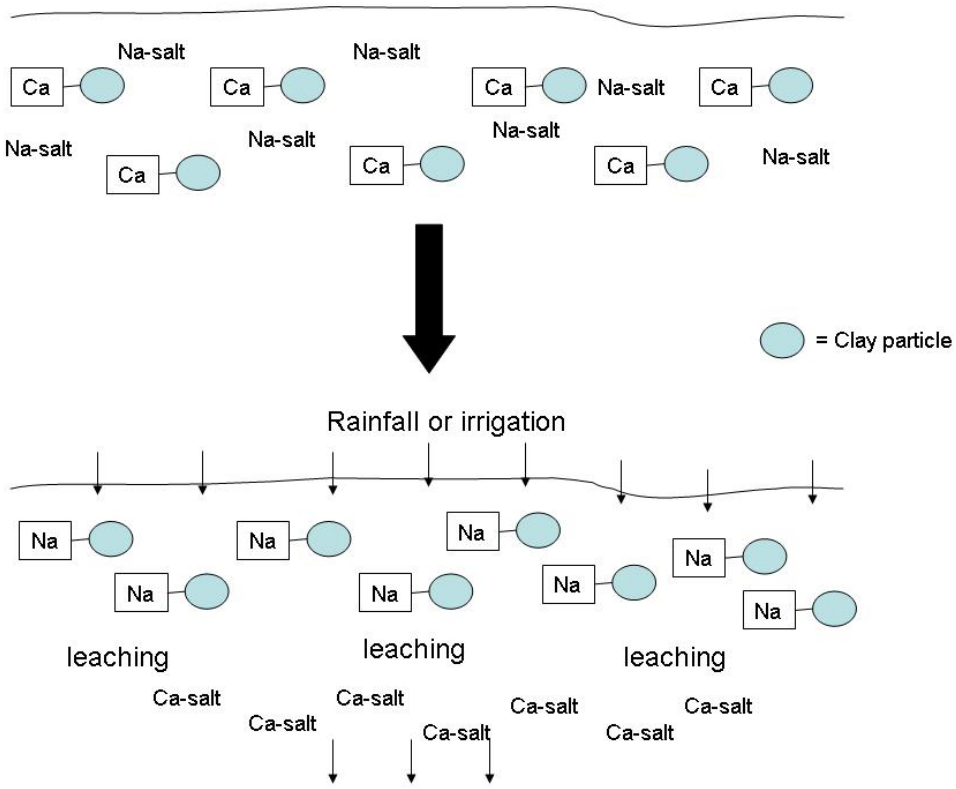


Figure 2.1 Formation of a sodic soil (b) from an initial saline soil (a)

The effects of sodicity can influence soil physical properties at a range of scales. Increased swelling and dispersion with increasing ESP also causes reductions in hydraulic conductivity, as disruption of aggregates causes larger pores to be blocked, decreasing water movement through the soil (So and Aylmore 1993). The reduction in hydraulic conductivity with increasing ESP is primarily due to the increased dispersion as a result of Na^+ ions, which reduces the proportion of transmission pores and increases the proportion of narrow pores, which are more susceptible to clay swelling.

The infiltration rate of a soil during rainfall is more sensitive to low ESP than to its hydraulic conductivity. This is due to its susceptibility to the mechanical energy of falling raindrops, in addition to the chemical effects of low electrolyte concentration of the applied water. Raindrop impact causes mechanical breakdown of aggregates at the

soil surface and, in conjunction with the relative freedom of particle movement at the surface, enhances the rate of chemical dispersion by stirring and compaction of a thin layer at the surface (Shainberg 1985; Shainberg and Letey 1984). The sealing of the surface is determined by aggregate breakdown, clay content and dispersion, with dispersion dependent on the ESP of the soil, while aggregate breakdown due to slaking is related to the rate of wetting of aggregates (Mamedov *et al.* 2001). In extreme cases, soils will form a massive structure when Na is involved in the association between clay particles, without any hierarchical arrangement between micro- and macroaggregates, becoming hardsetting when dry (Qadir and Schubert 2002). Removal of vegetation initially has the potential to enhance these processes due to the increased susceptibility to erosion by promoting the formation of stable colloid suspensions (Sumner *et al.* 1998).

Strong texture contrast duplex soils with highly impervious B horizons, the upper portion of which may be formed by the dispersed clay, can constrain water movement in the soil. Restricted water movement at the top of the impervious B horizon leads to waterlogging, erosion, by tunnelling, and lateral movement of subsurface water (Sumner *et al.* 1998). However, if soils shrink and swell, restructuring of the soil surface will constantly occur. In addition, pedoturbation brings subsurface clay to the surface such that the strong texture contrast common to non-swelling sodic soils is diminished (Shaw *et al.* 1998). Problems can arise during amelioration of these soils, since the ESP is readily reduced in the topsoil, but is more difficult to remove in the subsoil and may even increase (Surapaneni and Olsson 2002). Dispersion, erosion and eluviation of clay may lead to coarser textured A horizons which are less capable of retaining organic matter over time (Nelson and Oades 1998). These soils exhibit lower levels of C due to strong correlations between SOM and clay content of the soils, as SOM usually increases with increasing clay content.

Sodic behaviour can still be exhibited in soils at very low ESP levels, occurring where the electrolyte concentration is below the critical flocculation concentration (CFC), as shown in Figure 2.2. The concept of the CFC was introduced by Quirk and Schofield (1955), and is defined as the concentration of electrolyte required to develop a clear supernatant for a dispersed soil or clay suspension. As the ESP increases, the electrolyte concentration required for soil to remain flocculated, thereby maintaining soil structure

and permeability, also increases (Figure 2.2). As a result, permeability can be maintained through the application of water at the appropriate electrolyte level, depending on the degree of Na^+ saturation. While the extent of dispersion is due to high levels of Na^+ in a soil, complementary divalent cations, particularly Ca^{2+}, have the potential to promote flocculation (Keren 1996). The effects of dispersion and slaking in a saline soil on soil structure are minimal, due to the over-riding high electrolyte concentration of the soil solution, which causes the soil to flocculate rather than disperse. The high osmotic potential present in saline soils causes dehydration of the clay-water system, thus reducing the distance of separation between particles (Qadir and Schubert 2002), and results in the formation of stable clay-soil aggregates.

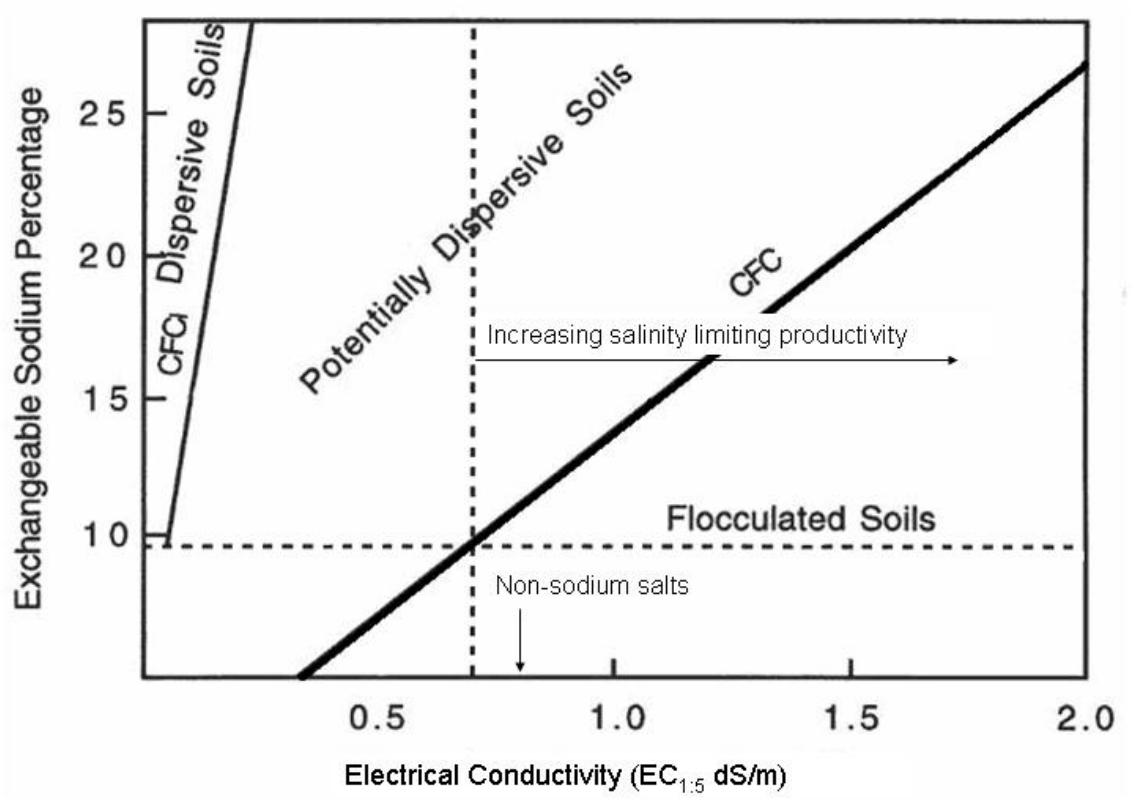


Figure 2.2 The relationship between ESP/EC and flocculated/dispersed soils
Source: Rengasamy *et al.* (1984)
Note: CFC is the critical flocculation concentration

Large influxes of water result in waterlogging in sodic soils due to poor internal drainage. Waterlogging of sodic areas results in anoxic conditions in the root zone which affect plant growth, while rapid drying of the surface and formation of surface crusts impacts upon root growth and seedling emergence. The dispersive nature of these soils also increases their susceptibility to mechanical stresses which further impact on

land management practices such as cultivation or tillage, as the development of compacted layers may occur with increased vehicular traffic (Rengasamy *et al.* 1984). This is particularly evident where soils lack structure. In these soils, the compaction of surface particles is the dominant process in seal formation, complemented by clay dispersion and clay accumulation in the conducting pores, rather than aggregate slaking and disintegration (Mamedov *et al.* 2001).

2.2.3 Effects on Vegetation

Soil C stocks in any particular area are a function of the C inputs, which are dominated by litterfall, root exudates and fine root decomposition, and are, therefore, dependent on biomass production, and outputs, which are dominated by microbial decomposition processes, leaching and erosion. As a result, declines in biomass production due to soil degradation will directly influence SOC levels. The effects of salinity and sodicity on plant physiology and physiological processes have been studied and reviewed extensively (eg. Akilan *et al.* 1997; Allen *et al.* 1994; Clemens *et al.* 1983; Craig *et al.* 1990), and will, therefore, not be covered in this review. Osmotic effects dominate in saline and saline-sodic soils, while declining soil structure dominates in sodic soils, adversely affecting nutrient and water supply. The adverse soil physical and chemical environment can affect plant growth directly, as shown in Figure 2.3, such as through specific ion and elemental toxicities (eg. Na^+, BO_3^- and Cl^-). The composition and concentration of salts in the soil solution adversely influence plant growth through osmotic effects by limiting water availability and the plant's ability to absorb water from the soil solution (Keren 2000). Increasing salt concentration increases the osmotic potential of soil water, resulting in plant cell dehydration and ultimately death. Indirect effects include decreased infiltration, especially in highly sodic soils, which affects the amount of water available for plants. Salinity and sodicity also induces Fe, Mn, Ca, Zn and Cu deficiencies, and B, Na and Cl toxicities (Naidu *et al.* 1992).

Salt and sodium stressed plants are further susceptible to high osmotic pressures, specific ion toxicities and nutritional disorders compounded by the poor physical properties of sodic soils. Root growth is also limited by suboptimal environmental conditions related to soil structure and toxic levels of Na^+. These factors directly limit plant growth through poor seedling emergence and root growth, and indirectly limit

plant nutrition by restricting water and nutrient uptake and gaseous exchange (Qadir and Schubert 2002). These limitations frequently occur where the B horizon has a high bulk density causing roots to concentrate in the surface horizons of the soil profile, thus increasing susceptibility of vegetation to stress during extended periods of drought (Curtin and Naidu 1998). High salinity levels in the seedbed also delay seed germination and increase stress during seedling establishment (Bell 1999; Oster *et al.* 1996).

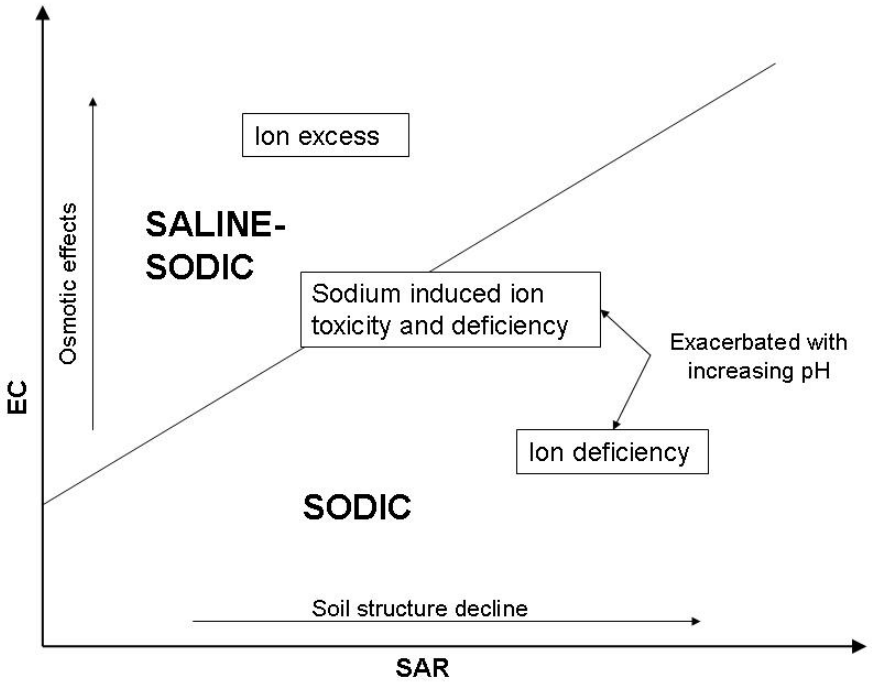


Figure 2.3 Nutrient constraints in sodic and saline-sodic soils
Source: Naidu and Rengasamy (1993)

Most Australian plant species are intolerant of both soil salinity and waterlogging, with the River Red Gum (*Eucalyptus camaldulensis*) being a notable exception (Akilan *et al.* 1997). Saline and sodic soils are subjected to prolonged waterlogging during rainfall events and rapid drying soon after. Waterlogging can reduce the ability of roots to exclude salt through increased passive diffusion of ions (Cramer and Hobbs 2002). Adequate soil water content is often difficult to maintain in areas affected by sodicity due to waterlogging at the surface, while the formation of surface crusts decreases infiltration, causing dry subsoils, and thus affects plant establishment (Oster *et al.* 1996). Seed germination is directly affected by waterlogging, due to the lack of oxygen

required for seed respiration, with extended periods of inundation resulting in failed germination (So and Aylmore 1993).

Decomposition processes in waterlogged soils alter the delivery and nature of nutrients. Waterlogging generally causes the pH to change due to changes in the partial pressure of CO_2, and creates anaerobic conditions whereby oxidation of organic matter decreases and results in its accumulation. Under theses conditions, atmospheric gases such as O_2 can only enter the soil by diffusion in the interstitial water (Ponnamperuma 1972). Low oxygen content is common where soils frequently waterlog, and can lead to chemical transformations of major nutrient ions (Fe, N, and S), rendering them unavailable to plants (Naidu and Rengasamy 1993). Redox potentials are, therefore, altered in waterlogged conditions while nutritional constraints are common due to altered ionic transformations. In waterlogged soils, organic matter breakdown is usually slower resulting in accumulation of SOM, and generates different end products compared to well-drained soils. In waterlogged soils, the breakdown processes produce partially humified residues, amines, NH_3, CH_4, H_2 and H_2S, in contrast to CO_2, NO_3, SO_4 and humus produced in non-waterlogged soils (Ponnamperuma 1972). N uptake is also restricted in waterlogged areas due to denitrification (Qadir and Schubert 2002), while NH_3 volatilization and inhibition of NO_3^- uptake by Cl^- also play a role in decreased N uptake (Gupta and Abrol 1990).

As a result of extensive weathering and lack of glaciation, many Australian soils are inherently infertile and deficient in many elements required for plant growth (Hubble *et al.* 1983), with salinity and sodicity interactions acting to further enhance deficiencies. The primary limiting nutrient in sodic soils is Ca^{2+} due to the high concentration of Na^+ in the soil solution. Ca^{2+} is also a limiting factor in terms of soil structural stability and plant uptake. Excess Na^+ in the soil solution causes enhanced uptake of Na^+ by plants, while uptake of Ca^{2+} is restricted, resulting in Na toxicity and concurrent deficiency in Ca^{2+}. This situation is compounded by the enhanced toxicities of other macro- and micronutrients, such as Zn, Mg and B (Curtin and Naidu 1998; Naidu and Rengasamy 1993). The majority of sodic soils in Australia have dense subsoils and an alkaline pH (Rengasamy and Olsson 1991), with micronutrient deficiencies exacerbated as a result. When the soil pH increases above 9, B toxicity becomes apparent due to the increasing concentration of $B(OH)_4^-$. This leads to a marked increase in B adsorption, which

accumulates where there is a low degree of leaching (Qadir and Schubert 2002). The toxicity of carbonate and bicarbonate, combined with a high pH can also lead to defiencies in Fe, Mn, Cu, Zn and P (Rengasamy 2002).

2.2.4 Increasing Carbon Stocks During Rehabilitation of Saline and Sodic Areas

The symptoms of sodicity are commonly ameliorated using one of two methods, both of which result in decreased dispersion and enhanced soil structure and improved infiltration and permeability: i) addition of Ca^{2+} salts as gypsum ($CaSO_4.2H_2O$), which facilitates the replacement of exchangeable Na^+ by Ca^{2+} by balancing the surface charge of the clay and restricting the development of the diffuse double layer; or ii) increasing the electrolyte level of the soil-water which causes compression to the diffuse double layer, thus preventing dispersion (Quirk 2001). Chemical dispersion is decreased when the electrolyte concentration of the soil solution is greater than the CFC. Gypsum is the most commonly used amendment to improve low water infiltration caused by low electrolyte content and/or high sodicity, while providing a source of Ca^{2+}. Other common amendments include lime ($CaCO_3$) and $CaCl_2$, as well as materials that enhance conversion of $CaCO_3$ to the more soluble $CaSO_4$. Elemental sulfur, and iron- and aluminium sulfates are also used and have potential for soil amendment. Dissolution is maximised with smaller gypsum particle sizes (Gupta and Abrol 1990). The addition of gypsum to soils with pH values greater than 9 causes precipitation of HCO_3^- and CO_3^{2-} complexes in association with Ca and Mg, lowering the pH to around 8.5. Similarly, application of gypsum to neutral sodic soils generally causes the soil pH to decrease by 0.5 to 1 unit (Rengasamy and Olsson 1991) by compressing the double layer and releasing protons.

The addition of organic matter in conjunction with gypsum has been successful in reducing adverse soil properties associated with sodic soils. Vance *et al.* (1998) found that addition of organic matter and gypsum to the surface soil decreased spontaneous dispersion and EC down to the subsoil, compared to the addition of gypsum alone. However, while soil strength decreased at the surface with additions of organic matter and gypsum, subsoil strength was not decreased, indicating that root growth was still restricted. Similarly, Chorom and Rengasamy (1997) found the application of green manure reduced soil pH in an alkaline sodic soil as a result of the decomposition of the

manure. Decomposition of the added manure caused an increase in the partial pressure of CO_2 which increased the solubility of $CaCO_3$. Where green manure was added in conjunction with gypsum, decomposition was enhanced, accelerating changes in soil solution composition.

Sodic soils are characterised by poor soil-water relations, which need to be considered during remediation processes. Whilst the addition of gypsum ameliorates soil chemical properties, tillage or deep ripping of clay layers is required to improve soil physical and hydraulic properties, and soil aeration. However, aggregate instability and recompaction due to increased trafficking can cause the ameliorative effects to be lost. The advantages of deep ripping are maximised when used in conjunction with gypsum incorporation, which maintains soil electrolyte levels at depth in the soil profile to prevent dispersion (Jayawardane and Chan 1994). As a result, rapid water redistribution to greater depths can occur compared to a soil that has not been ripped.

Where calcareous soils exist, or where sodic soils contain minerals that readily release soluble electrolytes, reclamation can be undertaken by leaching without additional amendments due to high electrolyte concentrations already present in the soil solution (Levy *et al.* 1998; Oster and Jayawardane 1998), provided drainage through the soil profile is sufficient. The presence of fine $CaCO_3$ particles in soils can improve the physical condition of sodic soils, stabilise soil aggregates and prevent clay dispersion by maintaining the soil solution at concentrations above the CFC values of soil clays (Levy *et al.* 1998), in addition to providing a source of Ca^{2+}.

Large increases in hydraulic conductivity of sodic soils can occur with the use of hypersaline irrigation water (EC > 20 dS/m) without the need for tillage or cropping in the remediation of a sodic soil. This technique involves successive dilution of saline irrigation water containing divalent cations, and can be applied when the soil's physical conditions has deteriorated and its hydraulic conductivity is low enough such that excessive time and/or amendment is required for reclamation (Keren 1996; 2000). The high salinity of the applied water prevents clay from dispersing by promoting flocculation, while providing a source of Ca^{2+} for the replacement of exchangeable Na^{+}, thereby decreasing sodicity (Keren 2000).

The use of gypsum in combination with other treatments has been found to improve overall soil properties to a greater extent than the use of gypsum on its own. For example, where lime and gypsum were combined in the amelioration of a sodic red-brown earth (pH <6.5), Valzano *et al.* (2001b) found higher levels of plant growth coupled with significant increases in total C in the soil over a period of three years. It was found that whilst gypsum was more effective than lime in displacing exchangeable and soluble Na, a combination of the two was more efficient at maintaining soil electrolyte levels and improving soil physical and hydraulic properties. This was due to the different solubilities of the two amendments, as gypsum could provide Ca^{2+} during the early stages of remediation due to its higher solubility, enhancing soil physical properties to allow greater throughflow of water into the soil, which would, in turn, allow for greater dissolution of lime in the later stages. Similarly, when gypsum was used as an ameliorant in conjunction with stubble retention and appropriate crop rotations, Valzano *et al.* (2001a) found interactions between all treatments which aided the improvement of soil properties. Gypsum addition decreased soluble and exchangeable Na^{+} concentrations, improving structural stability and hence, improved soil water relations. This results in higher crop yields, which build up SOC levels, thereby further improving soil structure. When stubble is burnt, macroporosity is reduced due to lower levels of biological activity and a reduction of throughflow of chemical amendment. The retention of stubble provides surface protection and prevents crust formation due to protection from raindrop impact. This improves infiltration while the use of leguminous crops may facilitate the leaching of gypsum through the soil profile, remediating soil properties at depth.

Rehabilitation of saline areas has largely been focused on three approaches: i) controlling recharge areas by revegetation; ii) controlling discharge areas by revegetation and stock exclusion, or iii) managing saline land and water by either fencing the area and removing it from production or the construction of drains, or a combination of both. In general, rehabilitation of saline areas focuses on controlling or minimising rising water tables in either recharge areas or discharge areas. Recommendations for salinity management usually rely on revegetation to control surface and subsurface flows. This includes placing deep rooted perennial vegetation to intercept surface and shallow groundwater before it interacts with deeper saline aquifers, thus intercepting recharge prior to where saline discharge areas occur (Hatton *et al.*

2003). Alternatively, placing vegetation in discharge areas may reduce the incidence of seepage, provided the planted vegetation is able to tolerate waterlogging. Planting of trees can induce substantial horizontal movement, in addition to vertical movement of water within the root zone, taking into account factors such as spatial distribution and tree density (Stirzaker *et al.* 1999).

Engineering options usually intercept saline groundwater flows, either diverting water flow or disposing of it at high river flows or to evaporation basins, with groundwater pumping and deep open groundwater drains often used as a last resort (Hatton *et al.* 2003). Reclamation of saline areas can occur through leaching of soluble salts out of the soil profile. Where good drainage conditions exist, saline soils can be reclaimed with continuous ponding, intermittent ponding or sprinkler irrigation (Harker and Mikalson 1990). However, if saline soils are also sodic, the use of high quality water may result in structural breakdown (David and Dimitrios 2002). Where sodicity is not at a critical level, leaching may reduce salinity in addition to causing reductions in sodicity. Leaching is preferable prior to revegetation in some instances to translocate the salt to below the root zone of plants to allow for the establishment of new vegetation, as the amount of salt removed by crop and pasture species is usually insignificant (Oster *et al.* 1996).

A large number of Australian tree species are able to control salinity by transpiring water from throughout the soil profile, with relationships found between increasing tree coverage in catchments and decreasing watertable levels (Bell 1999). One study has shown that while decreased growth of trees occurred in areas affected by salinity, access to fresh shallow groundwater led to increased growth rates compared to areas with no access to fresh groundwater (Feikama and Morris 2004). However, if the shallow groundwater was saline, growth was decreased. The ability to access fresh shallow groundwater is particularly advantageous as crop and pasture species are generally intolerant of waterlogging and salinity, which commonly occur in the lower positions in the landscape. Some perennial plants that are salt-tolerant and use saline groundwater have the potential to accumulate salts in their root zone due to salt exclusion processes. The accumulation of salt may result in the decline or death of nearby vegetation intolerant to salt (Barrett-Lennard 2002).

It is not economically viable to revegetate large areas with trees where average annual rainfall is less than 600 mm (Turner and Ward 2002), however, it is possible to lower groundwater tables through revegetation with perennial pastures. However, perennial pasture is unlikely to stop drainage below the root zone where average annual rainfall exceeds this amount (Ridley *et al.* 1997). In regions where rainfall is less than 600 mm/yr, perennial pasture species can decrease drainage compared to annual pasture or cropping, while rotations of perennial pasture with annual crops or pastures can provide a similar effect (Clarke *et al.* 2002). For example, lucerne (*Medicago sativa*) can extract water from deeper layers in the soil profile and has been shown to reduce potential groundwater recharge by up to 60 % annually compared to annual pasture (Ward *et al.* 2002).

Revegetation of sodic areas with trees or crops has also facilitated soil reclamation in the past where the vegetation could tolerate adverse soil conditions. The use of leguminous trees in India has been shown to reduce exchangeable Na^+ at depth as well as in the surface layers, decrease pH, and increase the soil microbial biomass (SMB) (Bhojvaid and Timmer 1998; Mishra and Sharma 2003). Ameliorative effects have been attributed to improved aggregation of soil particles which results in improved soil structure, and the production of CO_2 from plant roots. The increased CO_2 dissolves in the soil solution and lowers pH. In calcareous soils, the lower pH facilitates the dissolution of $CaCO_3$, releasing Ca^{2+} which displaces exchangeable Na^+ and results in a decrease in the ESP (Mishra and Sharma 2003; Figure 2.4), which aids in transformation of carbonates to forms available for exchange on clay particles (Lal and Kimble 2000a; Qadir *et al.* 2003). In northern Egypt, Ghaly (2002) found both ponding and gypsum were less effective in reducing salt content in comparison to the use of native grass species after the second year. This was attributed to increased salt uptake, as evidenced by increased sodium in the grass shoots, with the fine textured clay soil reclaimed within two years.

The presence of roots promotes aggregate stability through the *in situ* production of polysaccharides and fungal hyphae (Tisdall and Oades 1982). Decreased bulk density associated with tree root penetration can occur up to a metre in depth (Garg 1999), effectively improving hydraulic conductivity and soil structure. The physical effects of root actions which include the generation of alternate wetting and drying cycles, the

creation of macropores, and removal of entrapped air from the larger conducting pores enable reclamation of soils while providing financial and other benefits from crops grown during the rehabilitation process (Oster and Jayawardane 1998; Oster *et al.* 1996). Similarly, the presence of root channels in conjunction with gypsum aids in increased leaching of Na^+ and soluble salts (Ilyas *et al.* 1997), while Qadir *et al.* (1996) found that the presence of roots as a result of cropping decreased the SAR and removed Na^+ almost to the same extent as gypsum addition. Conversely, where soils are high in Ca^{2+} and Mg^{2+}, increased vegetation growth may cause the SAR to rise. Increasing root and microbial respiration in the soil may cause subsoils to become increasingly sodic as Ca^{2+} and Mg^{2+} are precipitated as $CaCO_3$ and $MgCO_3$ with increasing respiration, resulting in an increase in SAR (Gardner 2004).

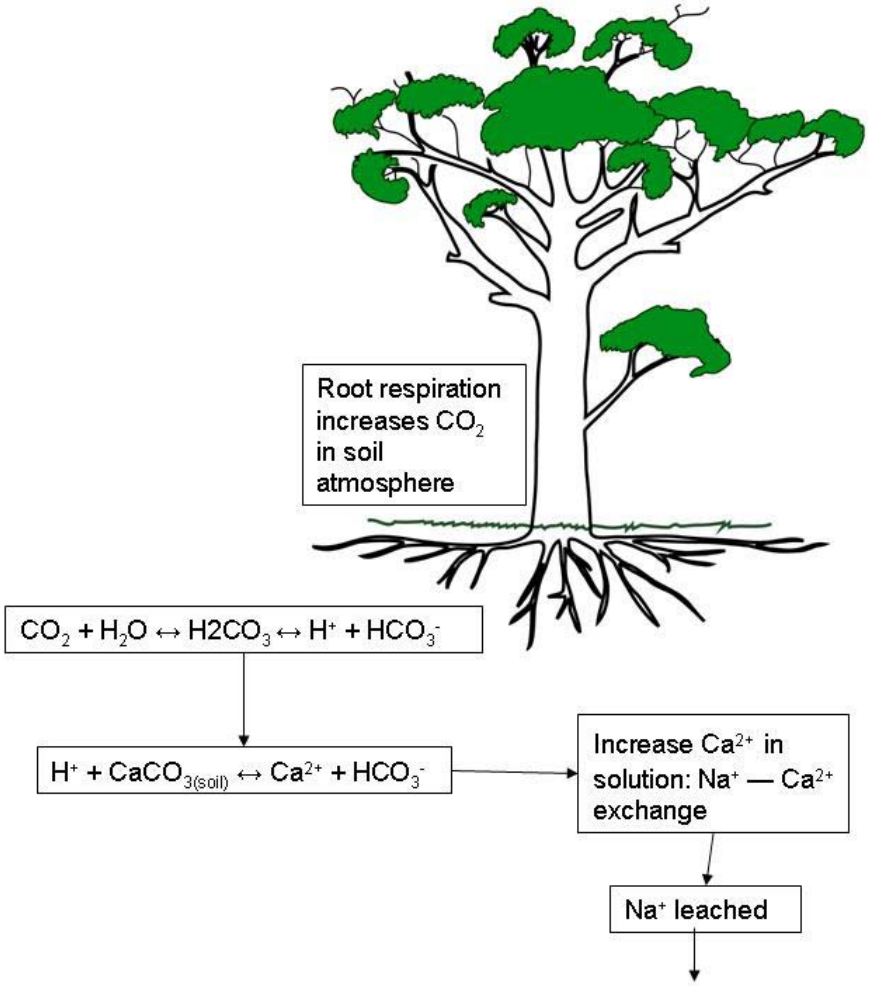


Figure 2.4 Processes involved in Na removal from sodic soils by vegetation
Source: Qadir *et al*. (2003)

Therefore, the restoration process by trees is primarily driven by two parallel mechanisms: the fertility building processes associated with organic matter addition, N

accretion and nutrient cycling; and alleviation processes driven by improved leaching which reduces soil dispersion and decreases Na toxicity (Bhojvaid and Timmer 1998). Higher microbial populations found in soils near the base of trees have been ascribed to accumulation of organic matter, stimulating microbial activity (Garg 1998) thus improving nutrient cycling and decomposition.

2.3 Soil Carbon Dynamics

Soil is the largest terrestrial C sink, and contains two thirds of the world's terrestrial C (Schimel et al. 1994), with approximately 1500 Gt of organic C in the top metre (Eswaran et al. 1993). The SOC pool contains twice as much C as the atmospheric pool, and three times as much as the terrestrial biotic pool (Lal 2003) and is, therefore, an important C store, with the potential to be a large C source under altered environmental conditions. The factors that influence soil C inventories closely follow that of soil formation, exhibiting gradients with climate, topography, vegetation, depth, which are then influenced by the management regime. The rate of net organic C accumulation or loss is a function of inputs and outputs according to the following mass balance equation:

d(soil C)/dt = Inputs (decomposition products + microbial/faunal residues) – Losses (heterotrophic respiration + leaching + erosion + burning) **Equation 2.3**

The decomposition of photosynthetic products is dependent on the productivity of the standing biomass and the quality of the substrate being decomposed, while losses are due to heterotrophic respiration by the microbial biomass, leaching and erosion. The amount of C in the soil at any particular time is dominated by inputs from vegetation in the form of leaf litter, fine root turnover and root exudates. As a result, C gradients largely follow that of plant biomass production, with soil C increasing with increasing precipitation (Burke et al. 1989) and decreasing temperature (Post et al. 1982) due to increasing biomass production and decreasing decomposition rates.

SOC can be partitioned into discrete pools according to its age or the amount of time it takes to turnover, as shown in Figure 2.5 (Jenkinson and Raynor 1977). Mean residence times are dependent on resistance to decay and the extent of protection against

decomposition. The three main SOC pools are: i) the active pool, with a turnover time in the order of weeks (ie. the SMB and particulate organic carbon; POC); ii) the slow pool with a turnover time in the order of decades (ie. humus); and iii) the passive pool with a turnover time in the order of millennia (ie. charcoal). The active pool is made up of readily oxidisable materials including the microbial biomass and its metabolites, and is largely controlled by climate and residue inputs (Schnurer *et al.* 1985). The slow and/or very slow pools contain macro- and microaggregates with chemically recalcitrant but moderately decomposable material, while the passive or recalcitrant pool includes recalcitrant and stable C formed from the turnover of microbial and slow SOC; this pool has organic compounds that are chemically resistant to, or protected from further microbial degradation (Schimel *et al.* 1994). Most C found in detritus and microbes is oxidised and cycled rapidly. Some is transformed into a slow reservoir with a turnover time on the order of decades to centuries, most of which will eventually oxidise. The remainder is converted to the passive pool with turnover rates on a millennial timescale (Stallard 1998).

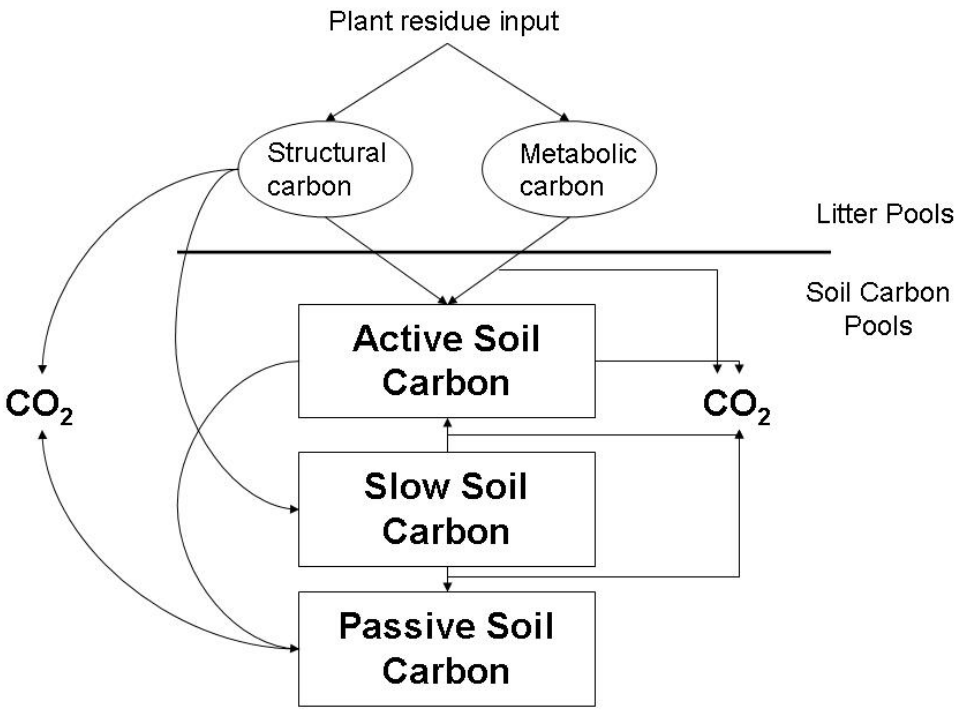


Figure 2.5 Conceptual model of soil C pools and turnover
Source: Jenkinson and Raynor (1977)

It is generally accepted that increasing levels of organic matter content in soils will improve soil structure, with organic matter having different roles at different scales and components (Figure 2.6). Whilst only present when plants are growing, plant roots, mycorrhizal hyphae and fungal hyphae at larger scales enmesh macroaggregates, inhibiting slaking and dispersion. At smaller scales, mucilages and colloidal organominerals are the primary binding agents in microaggregates (Table 2.2; Nelson and Oades 1998). The incorporation of organic matter into soil aggregates provides protection from rapid decomposition and is one of the key determinants of soil stability. Clay minerals can adsorb large organic molecules (Gregorich and Janzen 2000), which can provide physical protection, and hence, directly reduce their availability for decomposition. This was evident in aggregates with cores of organic material found by Waters and Oades (1991).

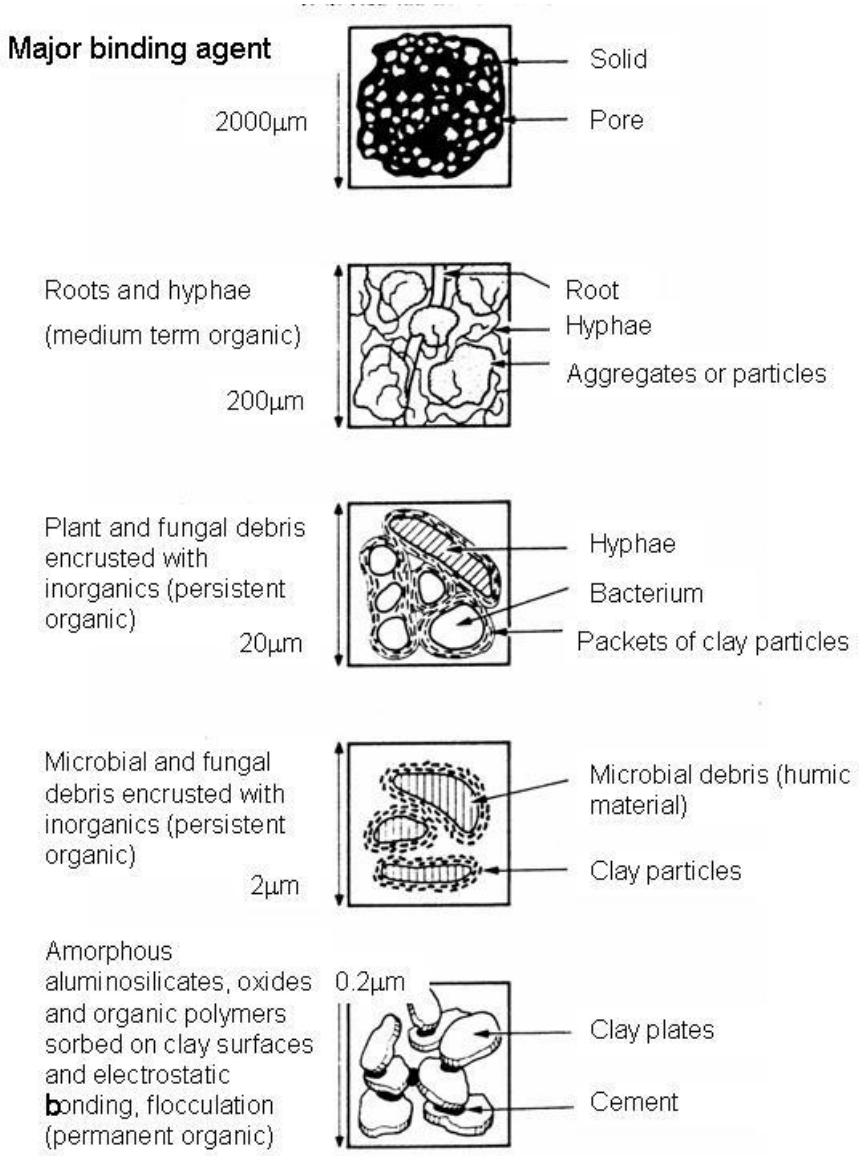


Figure 2.6 The role of organic matter in improving soil structure at different scales

Source: Tisdall and Oades (1982)

Table 2.2 The role of organic matter in the formation of aggregates

Type of SOM	Agents Involved	Description
Transient	Polysaccharides	Associated with large (>250 μm) transiently stable aggregates Polysaccharides decrease in importance with increasing organic matter contents. Decomposed rapidly by microorganisms.
Temporary	Roots	Associated with the growth of root systems and fungal hyphae
	Hyphae	Most likely associated with young macroaggregates.
Persistent	Polyvalent metal cations	Dominate in microaggregates
	Organomineral associations	Particles of clay sorbed on to organic matter core, rather than organic matter sorbed on to clay surfaces
	Strongly sorbed polymers	Most likely includes complexes of clay-polyvalent metal-organic matter
	Degraded humic material	Degraded aromatic humic material associated with amorphous iron, aluminium and aluminosilicates to form the large organomineral fraction of soil.

Source: Tisdall and Oades (1982); Gabriels and Michiels (1991)

SOC can become more stable by becoming biochemically recalcitrant or physically protected. C can also be precipitated out by Ca^{2+}, Mg^{2+} and Fe^{3+} as carbonate and rendered unavailable for microbial decomposition. Recalcitrant C or chemically protected C is often composed of residue decomposition products, which are considerably modified to form humic and fulvic acids, and humin. Material which exhibits large C:N ratios such as lignin and other resistant residues of plant origin (Rovira and Vallejo 2003) are often recalcitrant and difficult to decompose. Physically protected organic C can be located in pores too small for the microbial population to access or form into microaggregates, while the activity of those bacteria that are physically protected can also be limited with restricted flow of water and substrates (Hassink 1994). As clay content increases, mineralisation generally decreases, with textural effects either compounded or reduced by a range of factors including clay mineralogy, SOM chemistry and microbial composition (Wang *et al.* 2003). These effects, however, can also be influenced by management regimes.

Soil texture appears to exert the main control on soil C inventories, rather than climate or vegetation, with the retention of SOC proportional to clay mineral content (Bird *et al.* 2002). Ladd *et al.* (1985) found more extensive decomposition in coarse-textured soil compared to fine-textured soils with higher levels of SOM in similar climatic zones. Those fine-textured soils showed stabilisation of microbial products from decomposition. The role of clays is largely a result of the reactive surface area on clay particles which stabilise SOC in organomineral complexes. In addition, clay particles tend to form aggregates that physically protect SOC from decomposition (Schimel *et al.* 1994). The higher SOC content in fine-textured soils compared to coarse textured soils is due to differences in C input and long term decomposition dynamics, as fine textured soils tend to be more fertile than coarser textured soils (Franzluebbers *et al.* 1996a).

Clay content influences biological activity and C mineralisation to a greater extent than the level of sodicity by directly influencing interactions of substrate and organisms with clay mineral surfaces, with clay mineralogy exerting primary influence on microbial processes (Nelson *et al.* 1997). Killham (1994) has suggested that this may be due primarily to the bacterial portion of the microbial biomass adsorbed to clay particles, usually by ion bridges involving polyvalent cations. Conversely, SOC in the sand fraction is a very labile component of SOM resulting in faster turnover of C and N in coarser textured soils compared to finer textured soils, concurrently with faster turnover of the microbial biomass C and N (Juma 1993). Due to a larger porosity in sandy soils, water content fluctuates more rapidly than in more clayey soils, with periods of optimum conditions for microbial activity of shorter duration and occurring less frequently (Thomsen *et al.* 2003). However, when soils of different textures in the same study were adjusted to the same water content, mineralisable C was similar, with Thomsen *et al*. (2003) hypothesizing that the actual volume of water determines the proportion of total C that is in the potentially mineralisable pool. In coarse-textured soils, substrates are more readily available for mineralisation. Franzluebbers *et al.* (1996a) found that soil respiration per unit of microbial biomass was higher in coarse textured soils than in fine textured soils. This was attributed to the microbial biomass being more active either as a result of increased substrate availability or increased microbial predation, or being placed under greater stress due to larger water content fluctuations.

The dominant clay mineralogy also plays a role in the turnover of SOM, with organic matter associated with kaolinite exhibiting an average mean resident time of 357 years, while that associated with smectite was 1089 years (Wattel-Koekkoek *et al.* 2003). It was suggested that the faster turnover of kaolinite associated organic matter was due to the weak binding of organic matter to the mineral surfaces, such as iron oxides and the edges of octahedral sheets. Conversely, the fraction associated with smectitic clays was bounded by cation bridges and contained many aromatic compounds which are more difficult to decompose. In addition, mostly amorphous organic matter was found associated with smectites, indicating an advanced stage of humification and, therefore, turnover, while kaolinites were associated with more recognisable plant remains, indicating incomplete humification, and hence, faster turnover.

2.3.1 The Active Carbon Pool

The active pool is comprised of a living component, the SMB, and a non-living component, the dead biomass and its metabolic products, which comprise approximately 1-5 % of the total SOC (Sparling 1992). The ratio of SMB to total SOC can provide an indication as to whether SOM is being accumulated or lost (Anderson and Domsch 1989) and reflects the potential to transform organic C input into SOC and CO_2 (Santruckova *et al.* 2003). The active soil C pool is frequently used as an early indicator of SOM dynamics, due to its faster turnover rate (eg. Alvarez *et al.* 1998), as changes caused by management or environmental stresses can be detected earlier in this pool than in the SOM pool as a whole. Despite being a small proportion of the total SOC, the SMB is the driving force in any functioning terrestrial ecosystem, controlling microbially mediated processes such as the turnover and mineralisation rates of organic substrates, humification and nutrient mobilisation (Killham 1994).

Patterns of substrate utilisation and metabolic diversity in the active pool are more sensitive to management induced effects than the SOC pool as a whole and hence, reflect changes in soil quality earlier than chemical analysis of the SOM. This can be particularly important in cases where pasture or crop yields are affected (Franzluebbers and Stuedemann 2003). Its use as an early indicator is particularly evident where total SOC is low, such as that found in arid or semiarid areas (Garcia *et al.* 1994). The SMB exhibits fluctuations with temperature, moisture, and availability of substrate, which is largely dependent on vegetation. The SMB is dependent on both above- and

belowground C inputs, with substrate provided in leaf litter and animal dung aboveground, and root turnover and exudates belowground (Franzluebbers and Stuedemann 2003). As a result, changes to inputs are reflected in the SMB.

Soil respiration, SMB and SOM levels appear to be intricately linked. Levels of microbial biomass have been linearly correlated with the level of SOC (Anderson and Gray 1991), while Franzleubbers et al. (2001) found strong correlations with respiration rates and SMB across climate regions. Similarly, the SMB and microbial diversity, an indicator of functional diversity, is also correlated with total C and N content of soils which has been attributed to productivity and fertility of sites, providing favourable conditions for microbial growth and activity (Banu et al. 2004).

Soil respiration is frequently used as a measure for microbial activity, and to determine whether a microbial population is under stress. As with the SMB, respiration rates are dependent on biota, substrate availability and quality, and environmental conditions such as O_2 availability, temperature and water content. The total respiration rate is the sum of heterotrophic respiration (the mineralisation of litter and humus by microbes and soil fauna) and autotrophic respiration (live root respiration). The dependence on the amount of substrate available for decomposition is reflected in the determination of respiration rates at a steady state by the amount of C addition to the soil, which is usually proportional to the net primary productivity (NPP; Kirschbaum 1995).

Root and microbial respiration processes are difficult to separate *in situ.* Root respiration is estimated to contribute to approximately 40-50 % of total soil CO_2 efflux rates, dependent upon aboveground processes and conditions such as seasonal light and water variations (Hanson *et al.* 2000). Vegetation plays a large role in influencing soil respiration by altering the soil microclimate and structure, the quantity and quality of detritus supplied, and the overall rate of root respiration (Raich and Tufekcioglu 2000). Roots are a source of CO_2, in addition to providing substrate for mineralisation, including exudates, sloughed-off material and dead roots for decomposition (Buyanovsky and Wagner 1995).

2.3.1.1 Measures of Biological Activity

The metabolic quotient (qCO_2), the ratio of the rate of respiration per unit of microbial biomass, or the specific microbial respiration rate, has frequently been used to determine stress in the microbial population (Anderson and Domsch 1993). It is assumed that the microbial biomass produces more CO_2-C per unit microbial biomass per unit time as stress increases, and hence, results in an increase in qCO_2 (Anderson and Domsch 1993). As the microbial population is increasingly stressed, more C is lost through respiration rather than being converted to humus. It has also been suggested that soils with a smaller biomass, which may or may not be related to soil environmental conditions, will have higher maintenance energy requirements, reflected in higher respiration rates (Dahlin and Witter 1998). Therefore, may be possible to ascertain whether the microbial population is under stress as salinisation and sodication occur with the use of the qCO_2.

A number of studies have previously used the qCO_2 as an indicator for microbial stress in studies relating to the addition of heavy metals to soils. Chander and Brookes (1991b) found the qCO_2 to be higher in metal contaminated soils, than in non-contaminated soil due to increased diversion of C from biosynthesis to respiration as a result of stress. Similarly, Barajas Aceves *et al.* (1999) found a higher qCO_2 in soil with higher Zn concentrations, and attributed this to a lower C assimilation efficiency. Conversely, Chander and Brookes (1991a) found no differences in the qCO_2 of soils following incorporation of high-metal sludges and low-metal sludges. It was suggested that the large availability of fresh organic material overcame any inhibitory effects of high metal concentrations.

In forest soils, Wolters and Joergensen (1991) related increasing qCO_2 values with increasingly acidic soils, due to the inefficient use of C resources by the microbial population. However, in separate study by Anderson (1998), it was hypothesised that liming of an acidic forest soil should have decreased the qCO_2 due to the mediation of soil pH, but it was found that it did not significantly affect the metabolic quotient. In the same study, acid application to a limed plot resulted in an increase in the qCO_2 due to the reduced substrate use efficiency and stress under acidic conditions. Similarly, increasing pH in an alkaline soil can also cause an increase in the qCO_2 as a result of reduced efficiency (Li *et al.* 2007). It was also postulated that the higher metabolic quotient due to increasing alkalinity caused a shift to a more bacteria dominated

community which is less efficient at utilising C substrates. A study by Mendham *et al.* (2002) found no significant differences in the metabolic quotient between different land-uses, yet found the metabolic quotient to be negatively correlated with increasing clay and silt. It was suggested that this relationship may have been due to a number of mechanisms, which included physical protection of SOM, or the correlation between soil texture and climate, whereby drier regions tend to have sandier profiles compared to soils in wetter regions.

While the qCO_2 is the rate of respiration per unit of SMB, the microbial quotient is the ratio of the SMB-C to SOC (C_{mic}:C_{org}), and indicates the ratio of the living fraction of SOC relative to the non-living fraction. It has been suggested that this ratio is responsive to land management practices and can provide an indication to the substrate availability by increasing where organic input increases and decreasing where input decreases (Anderson and Domsch 1989). For example, Haynes (1999) suggested that the decreasing C_{mic}:C_{org} with depth was the due to the decreasing proportion of readily available substrate. As with the qCO_2, C_{mic}:C_{org} can also indicate stresses on the microbial population. Barajas Aceves *et al.* (1999) found that the C_{mic}:C_{org} decreased with increasing Zn concentrations in contaminated soils. However, Marinari *et al*. (2007) found that C_{mic}:C_{org} decreased following the addition of inorganic fertilisers which may have been due to the decrease in the number of bacteria, as bacterial communities are less efficient at converting substrate C into cellular C compared to fungi (Kuzunori and Oba 1994).

2.3.2 Effects of Land Use and Land Management Practices

Maintenance of SOM levels is particularly important in agricultural settings as a result of repeated removal of biomass due to cropping or grazing, in addition to its role in the stabilisation of soil structure and as a buffer in the soil environment. The effects of management practices, particularly those related to agriculture, have the potential to alter C stores and turnover, although the results are not always clear-cut. Losses of SOC due to land use change are largely related to practices which reduce inputs of organic matter, increase the decomposability of organic material, and increase accessibility of substrates for decomposition. However, techniques that improve soil, crop and water management can aid in increasing SOC stocks, including management of crop residue,

conservation tillage, nutrient management, site-specific farming and restoration of degraded soils (Lal *et al.* 1999).

A review of the effects of land use change on soil C stocks by Guo and Gifford (2002) indicated that soil C stocks increased following conversions from native forest to pasture, cropping to plantation, cropping to secondary forest and cropping to pasture, while the reverse of these conversions saw a decrease in soil C stocks. In general, pasture grasses maintain a continuous cover of vegetation which adds organic matter and decreases mineralisation rates by reducing soil temperatures compared to cropping. Concomitant with increases in SOC stocks are increases in the SMB and microbial activity, particularly where agricultural activities have been abandoned (Hedlund 2002). This is particularly important where degraded landscapes are to be restored, such as those common to salinity and sodicity, as those processes determined by the SMB also need to be restored if rehabilitation efforts are to be successful. Ros *et al.* (2003) found the SMB and basal respiration increased following remediation of a degraded soil in south-east Spain. Following the addition of organic amendments, the SOC content, SMB and soil respiration increased due primarily to the development of plant cover and the mineralisation of root exudates and plant material. This was attributed to the incorporation of easily decomposable materials which stimulated the native microbial population into activity, and incorporated exogenous microorganisms.

In grassland environments, management activities and land use conversions which increase aboveground production usually increase SOC levels despite environmental conditions (Conant *et al.* 2004). One notable exception exists where pastures have been afforested with *Pinus radiata*, as SOC stocks and SOM quality have been observed to decrease (Ross *et al.* 2002). A number of hypotheses for the decline in SOC stocks and SOM quality have been suggested. It has been noted that soil processes occurring under *P. radiata* forests are vastly different to those occurring in grasslands as lower levels of SMB were found in New Zealand under these forests compared to pasture (Saggar *et al.* 2001; Scott *et al.* 1999). Mineralisation rates were also found to be higher under pasture as a result of the higher rates of inputs related to the higher proportions of easily decomposable plant material. It is possible that stabilisation of SOC occurs in pasture soils due to root exudates and rhizosphere processes from the activity of live roots which may not occur in *P. radiata* plantation soils (Guo *et al.* 2005). A review by Cowie *et al.* (2006) have suggested that declines on SOC are related to lower soil C

input by trees compared to pasture, possibly due to differences in belowground C allocations, root turnover times, soil environmental conditions and nutrient supply. Afforestation with *P. radiata* also exhibits an inverse trend in terms of SOC compared to afforestation with native species in Australia (Guo and Gifford 2002), which may be related to induced soil changes such as decreased pH caused by organic acids and resins released by decomposing needles from *P. radiata* trees (Saggar *et al.* 2001).

Cropping has the potential to result in continuous losses of SOC compared to perennial pasture or native vegetation. In China, the largest losses of SOC following cultivation occurred in those areas used in dryland cultivation in semi-arid and semi-humid areas, characteristic of a zone between the north-east and the south-west of China (Wu *et al.* 2003). In Australia, losses of SOC have been observed where long term continuous cropping and cultivation have taken place, largely related to decreasing amounts of organic material being returned to the soil (Dalal and Mayer 1986). The retention of stubble however, may reduce the rate of net organic matter loss by increasing inputs of organic materials in the form of crop residues. As cropping intensity increases, SOC stocks can also increase where double cropping can be applied (Sherrod *et al.* 2003). Continuous cropping reduces the opportunity for the oxidation of SOM. As the number or length of summer fallow periods increase, losses of SOC stocks also increase through mineralisation processes due to increased accessibility and temperature. Similarly, Sparling (1992) found that by using permanent pasture as a baseline, continuous cultivation for maize caused a decline in SOC, again attributed to decreased organic material input into soils under cropping systems compared to permanent pasture. Parfitt *et al*. (1997) found SOC to decrease from native forest to perennial pasture, and decrease again to cropping with maize. The declines in SOC with cultivation are also the result of a greater proportion of readily decomposed crop residues, which is rapidly lost (Post and Kwon 2000). Conversely, when cultivated lands are converted into permanent pasture, SOC stocks are likely to increase due to continuous inputs of organic material.

The effects of tillage on soil C stocks and processes have been studied extensively in the past (eg. Balesdent *et al.* 2000; Cambardella and Elliot 1993; Franzluebbers *et al.* 2000; Izuaurralde *et al.* 2001b; Jackson *et al.* 2003; Lal *et al.* 1999; Sherrod *et al.* 2003). Tillage in sodic soils has been suggested as a method for improving physical and hydraulic properties, with deep tillage (1-2 m) able to break up hard pans and cemented

layers while concurrently mixing soil layers, thus altering the distribution of SOM in the soil profile. However, conventional and deep tillage have also been known to increase C mineralisation, with Franzluebbers *et al.* (1996b) observing that mineralisation increased under conventional tillage compared to no-till, with seasonal variations of mineralisation also greater. These effects can also be altered by changing the placement of residues, and the quantity, quality and timing of crop residues with tillage.

The formation of soil aggregates can physically protect soil C, such that any process which disrupts these aggregates will most likely increase C mineralisation. Under conventional tillage treatments, aggregates are frequently disrupted, resulting in fewer stable macroaggregates and the mineralisation of previously protected organic matter (Paustian *et al.* 2000). This is due to the more labile nature of the organic matter associated with macroaggregates, which is, therefore, more readily mineralisable compared to that associated with microaggregates (Waters and Oades 1991). Similarly, Cambardella and Elliot (1994) found organic C in macroaggregates to be highest in a soil that had not been tilled, compared to one that was bare fallow and one that retained stubble and had been tilled.

Where tillage is reduced, residues concentrate on the soil surface and decomposition rates decrease due to reduced contact with soil microorganisms, allowing for SOM to accumulate over time (Salinas-Garcia *et al.* 2002). The SMB is also affected closer to the surface by tillage than at depth, which is attributed to the accumulation of residues at the soil surface. Tillage often causes compaction due to agricultural traffic, which causes bulk density to increase and the volume of pore spaces to decrease in areas of high traffic, subsequently restricting biotic activity. Santuckova *et al.* (1993) found the SMB to be lowest in areas of high tillage compared to no tillage, and ascribed this to alternating cycles of disruption and gradual recompaction, causing SOM and SMB to decrease.

Grazing has the potential to alter the levels of SOM, with different grazing pressures found to alter biological activity and C stores in soils. In a study by Franzluebbers and Stuedemann (2003), POC, SMB and C mineralisation were higher under higher cattle grazing pressures, with a cattle stocking density of $8.7 \pm 1.9 \times 10^{-4}$ head/m^2, compared to a lower rate of $5.8 \pm 0.9 \times 10^{-4}$ head /m^2, due to the stimulation of the SMB and microbial activity with grazing and the return of dung to the soil. However, these pools

were decreased when the forage was harvested because organic C contents are a function of organic material returned to the soil. It is possible to increase SOC levels with grazing at light to moderate stocking densities, as it promotes an increase in belowground biomass, particularly in the fibrous rooting networks characteristic of grass and pasture species which promotes SOM formation and accumulation. Long term grazing in shortgrass communities in North America along an environmental gradient has resulted in larger plant basal areas in a mesic environment which may have been partially responsible for the increase in SOC (Derner *et al.* 1997). However, in ungrazed communities in a more arid environment with a greater proportion of large plants, greater amounts of organic matter are most likely incorporated into the soil and are more effective in capturing and redistributing organic matter. These apparent differences found by Derner *et al*. (1997) were attributed to changes in the population structure of the vegetation communities.

Increasing erosion, common in saline and sodic landscapes, has the potential to cause substantial SOC losses. Erosional processes can deplete the SOC content of the surface layer due to its lower density and higher erodibility. Because the labile particulate fraction is relatively unconsolidated it is therefore most prone to removal (Lal 2001). In addition, as soil aggregates break down during the process of erosion, there is an increase in exposure to microbial processes, and thus mineralisation. Eroded materials, which usually consist of humus and clay fractions, can contain 3.5 times more C than the original soil. Translocation into lakes, reservoirs and other aquatic systems, deep burial or downslope deposition into waterlogged areas of these eroded materials may result in sequestration as decomposition processes are slowed in such environments (Izaurralde *et al.* 2001a; Izaurralde *et al.* 2001b; Izuaurralde *et al.* 2001a; McCarty and Ritchie 2002; van Noordwijk *et al.* 1997). However, in general most displaced SOC is mineralised, with this effect compounded by the decreased biomass capacity of eroded soils (Jacinthe and Lal 2001).

2.4 Salinity, Sodicity and Carbon

Those areas susceptible to salinity and sodicity are also the most susceptible to significant SOC losses as a proportion of total SOC. Salt-affected landscapes are usually found in areas of marginal agriculture, in association with soils of lower fertility and

hence, lower biomass production (Isbell *et al.* 1983), with a high susceptibility to erosion, which further accentuates the losses of C from the soil.

2.4.1 Effects on Microbial Decomposition

The SMB controls the decomposition of organic substrates, with rates of decomposition dependent upon the microbial population, soil environment and quality of substrate. C:N ratios are commonly used to describe substrate quality, with wide C:N ratios indicative of less decomposable material and vice versa, largely related to N limitations (Gregorich and Janzen 2000). Because the SMB fraction can act as an early indicator of longer term changes in the total SOC, the effects on microbial activity caused by increasing salinity and sodicity should precede effects on the total C stock, and should be detected prior to the more obvious effects of declining vegetation health, changes in biomass C inputs and the SOM.

Chander *et al.* (1994) found the rate of mineralisation of organic matter increased as sodicity increased, while the SMB decreased. The smaller microbial population was most likely the result of decreased plant inputs due to stresses placed on plants with increasing sodicity, which can be measured as the qCO_2 or C_{mic}:C_{org}, as described in Section 2.3.1.1., while direct toxic effects and environmental stress play a smaller role, as described in Section 2.3.1. However, the same study found that the reduced biomass was just as effective in decomposing the smaller amount of organic residues as the biomass found in a non-sodic soil. Native and additional organic material can become more readily available or easier to decompose as a result of the presence of alkali salts, which have the potential to dissolve, disperse, or cause chemical hydrolysis of the organic material. Laura (1973) has shown that losses in total C increased with increasing concentrations of Na_2CO_3 during decomposition of organic material. As the concentration of the added Na_2CO_3 increased, exchangeable Na^+ also increased resulting in higher ESP, while pH increased as carbonates of Ca and Mg precipitated. As a result, losses of SOC occurred due to the processes described above. Similarly, Laura (1976) found losses of C to increase with increasing ESP. While the effects of increasing sodicity can be evident in the SMB in the order of weeks, as described in Section 2.3, the adverse soil environmental conditions with increasing sodicity will deleteriously impact on plant growth which will ultimately result in lower inputs of C over much longer time frames in the order of decades, and hence, lower levels of SOC.

The dissolving or dispersing action of Na on organic molecules and organomineral complexes can increase the concentration of organically complexed metals in solution. The complexed metals, dependent on their stability, can be released by low pH or mineralisation (Nelson and Oades 1998). Mineralisation of ground plant C has been found to increase with sodicity and decrease with salinity (Nelson *et al.* 1996). This may be due to the high solubility of organic matter in the presence of Na. Because Na is more soluble than Ca, mineralisation may be stimulated, causing increased losses of C as dissolved organic matter, with the effect greatest on small or colloidal anionic substrates and least for particulate uncharged substrates (Nelson and Oades 1998). In contrast, Nelson *et al.* (1997) found a slightly negative effect of sodicity on mineralisation, which may have been due to differences in the amount and quality of substrate added. It is likely that C substrates that are amenable to dissolution will also increase in solubility with increasing sodicity, while those that are less readily soluble and decomposable are inhibited by increasing sodicity.

Similarly, in a study conducted by Pathak and Rao (1998), C mineralisation decreased with increasing salinity due to a decrease in microbial activity, indicated in the smaller amount of decomposed plant material. However, the evolution of CO_2 at high salinity levels indicated that biochemical mineralisation by soil enzymes can still occur in saline and alkaline conditions. El-Shakweer *et al.* (1977) found addition of Na_2CO_3 and $CaCO_3$ favoured decomposition of clover straws, while sulfate and chloride salts decreased the rate of decomposition, with the slowest rates found with $CaCl_2$ and $CaSO_4$. A diminishing rate of decomposition of clover straw with time was also found with increasing salinity. However, remediation of sodic soils through the addition of gypsum can reduce mineralisable C, with increases in microbial biomass, as noted by Carter (1986). Where a combination of lime and gypsum was added, pH was restored in conjunction with increases in both biomass C and nitrogen, and microbial activity. Both the long and short term studies indicated that addition of gypsum caused a significant increase in the C:N ratio of the microbial biomass, and reduced its activity with regards to C release and mineralisation (Carter 1986).

While sodicity has been shown to increase the rate of mineralisation, salinity has the opposite effect due to its osmotic influence on microbial activity. The more efficiently soil microbes function, the less C is lost via respiration (Insam 1990). An increasingly

stressed microbial community, caused by increasing salinity and sodicity, results in less efficient use of C resources where a greater proportion of substrate C is lost as CO_2 per unit of microbial biomass through increased respiratory activity (Rietz and Haynes 2003). The study by Rietz and Haynes (2003) found that in an irrigation-induced saline and sodic sugar cane estate, high soluble salts were more important in inhibiting the growth and activity of soil microbes than in inhibiting plant growth. It was suggested that increasing salinity and sodicity resulted in a smaller, more stressed microbial community as indicated by a reduction in the rate of organic matter decomposition and the mineralisation of C. These results follow a general pattern found in naturally saline soils, with the SMB negatively correlated with the concentration of soluble salts, and positively correlated with SOC contents. Increasing levels of salinity have also been shown to decrease soil enzyme activities (Batra and Manna 1997), with inhibition of enzymatic and microbial activity greatest with NaCl, compared to $CaCl_2$ and Na_2SO_4 (Frankenberger and Bingham 1982). However, McCormick and Wolf (1980) found that when a C source is readily available in the form of organic material, the adverse effects of NaCl on microbial activity are reduced. The effects of salinity on microbial activity have been attributed to the similar deleterious effects on plant health, dominated by osmotic effects with increasing salt concentration, and specific ion toxicities causing nutritional imbalances for microbial growth and enzyme synthesis (Batra and Manna 1997).

Garcia *et al.* (1994) found that decreasing CO_2 emissions can also reflect a stressed microbial population, such as that found in a saline soil of an arid region in south-east Spain. Soils showed low microbiological activity, with the lowest values found at the most degraded site, determined by its low organic matter content and lack of vegetative cover. At this site, soil respiration was inhibited at high EC levels. In the same study, Garcia *et al.* (1994) observed that qCO_2 did not vary in the arid zone soils studied, and concluded that the index was stable and could not be used to assess soil degradation or fertility, in contrast to Rietz and Haynes (2003) and some of the studies discussed in Section 3.2.4.3. However, while salinity can depress plant growth, Sadinha *et al.* (2003) found significant microbial activity was still associated with saline and acidic sites. The combined effects of salinity and low pH lead to the conclusion that salinisation has a depressive effect on the microbial biomass, which is most likely due to a shift in community structure from one dominated by fungi to one dominated by prokaryotic microorganisms (mainly bacteria). The survival of specialised and adapted species in

saline conditions may result in a less competitive microbial community (Zahran 1997) dominated by bacteria, which is less active and less diverse (Pankhurst *et al.* 2001).

2.4.2 Effects of Organic Matter on Sodic Behaviour

Sodic soils usually exhibit low organic matter content due primarily to poor plant growth, which leads to low inputs of organic materials into the soil and increased losses due to erosion and leaching (Nelson *et al.* 1996). This is compounded by the generally lower C content in Australian soils compared to other soils globally (Spain *et al.* 1983). Sodic soils also often coincide with alkaline conditions, due primarily to the presence of inorganic C, with pH high enough to dissolve organic matter (Sumner 1993).

Despite the commonly held belief that an increase in organic matter levels improves soil physical and chemical properties, results from studies on the effects of organic matter on dispersion in sodic soils have been mixed. The accumulation of organic matter in sodic soils is difficult as Na-organic linkages are highly soluble, with organic matter dissolving in runoff and percolating water in the form of soluble Na-humates, which further enhance clay dispersion, mobilisation and losses of SOM from leaching (Sumner *et al.* 1998). Highly alkaline soils are unlikely to retain products of decomposition because organomineral interactions depend primarily on cation bridges involving mainly Ca^{2+} rather than Na^{+} (Naidu and Rengasamy 1993). Organic matter can enhance aggregate stability by forming linkages between particles which are stable in water (Mamedov *et al.* 2001). However, Na^{+} must first be replaced by polyvalent cations, which would subsequently enable the formation of stable linkages between particles by organic matter because the linkages formed between organic matter and Na^{+} are largely ionic and solvated in water. Ca^{2+} ions tend to form covalent bonds, which are more stable in water, suggesting that sodicity needs to be ameliorated prior to the addition of organic matter (Rengasamy and Olsson 1991).

Additions of organic matter to calcareous and non-calcareous soils have been shown to cause increases in clay dispersion at constant pH and high SAR values (Gupta *et al.* 1984). This was attributed to the effects of increasing soil pH following addition of manure, which increased the CEC and altered the surface properties of the clays, thus promoting dispersion. Rengasamy and Olsson (1991) have suggested that Na^{+}-organic linkages are generally weak, with accumulation of organic material in aggregates an

ineffective method in soil structure stabilisation. Conversely, Barzegar *et al*. (1997) found spontaneously dispersible clay to decrease with the addition of pea straw, with stabilisation occurring irrespective of SAR, indicating that the dominant binding mechanisms were not ionic. In contrast to Rengasamy and Olsson (1991) and Gupta *et al.* (1984), Barzegar *et al*. (1997) suggested that the addition of organic materials to sodic soils could be expected to improve structural stability without initial remediation of sodicity, as native organic matter and additional plant residues had a positive influence on stability. This effect of improved structural stability occurred irrespective of clay type or sodicity, with the effect greatest at high organic matter contents and low ESP where soils are not highly sodic. While the addition of humic materials can increase the CEC substantially, clay dispersion was found to increase where the ESP of the soil was between 10-30 due to the greater preference for Ca^{2+} to Na^{+} by organic matter compared to clay minerals (Sumner 1993). This subsequently caused an enrichment in Na^{+} in the inorganic clay fraction, while the contribution of low molecular ligands from the added organic matter also promoted dispersion.

A high soil pH can compound the dispersion potential as a result of an increasing negative charge on organic molecules (Rengasamy and Olsson 1991). However, the presence of polyvalent cations limits the swelling of clays, as these cations bridge clay particles and organic macromolecules together with the main cations involved being Ca^{2+} and Mg^{2+} in neutral and alkaline soils, and hydroxypolycations (Al^{3+} and Fe^{3+}) in acidic and ferallitic soils (Oades 1988). In addition, those soils with a high base status typically have higher clay content, and are generally more fertile with greater vegetation input (Baldock and Nelson 2000), subsequently producing more organic matter.

The effects of organic matter on soil physical properties are usually only related to a certain fraction of the organic matter. Soils high in organic matter are generally resistant to Na adsorption, and rarely display sodic behaviour; this is largely related to increased hydrophobicity caused by the presence of hydrophobic organic compounds (Rengasamy and Olsson 1991). Rengasamy and Olsson (1991) and Golchin *et al.* (1994) found that the stability of soil structure was more closely related to young and active SOM than to total SOM. The older humic acid fraction, which is most likely protected from microbial decay, is not associated with the soil matrix, and is, therefore, not directly involved in the stabilisation of soil aggregates. The encrustation of debris, found in aggregates 1-5μm in size associated with the humic acid fraction, is an important process in the

stabilisation of microaggregates (Waters and Oades 1991). Microaggregates are stabilised against disruption by rapid wetting and mechanical disturbance by organomineral complexes and polysaccharides, while the stability of macroaggregates depends upon roots and hyphae (Tisdall and Oades 1982). Microaggregates are relatively permanent and not influenced by changes in the organic matter content of the soil or management regimes, while the number of macroaggregates declines with decreasing organic matter content as roots and hyphae are decomposed and not replaced.

2.4.3 The Role of Inorganic Carbon

Whilst beyond the scope of this project, it should be noted that large amounts of soil inorganic C (SIC) exist in the subsoil of soils affected by sodicity but remain insoluble due to high soil pH, and have the potential to play a large role in C cycling. The SIC pool has been estimated to contain approximately 940 Pg of C to one metre depth (Eswaran *et al.* 2000). While the SOC pool dominates in soils of humid regions, SIC is the most common feature of C in arid and semiarid regions, usually where precipitation is less than 500 mm per year (Lal and Kimble 2000b). Pedogenic carbonate often occurs in soils across the southern and inland regions of Australia, and is estimated to cover about 50 % of the landscape, usually in conjunction with sodic soils (Fitzpatrick and Merry 2000).

Studies have linked the formation of pedogenic $CaCO_3$ to the development of sodicity (Pal *et al.* 2000). The formation of $CaCO_3$ removes Ca^{2+} from the soil solution causing sodicity to develop or increase in the subsoil. As sodicity increases, hydraulic conductivity of the soil decreases, resulting in an increase in ESP with depth, as the formation of $CaCO_3$ continues and leaching of Na^{+} decreases. It is often difficult to separate the effects of sodicity from those of pH, as sodic soils commonly occur in conjunction with alkalinity, usually due to the presence of carbonates.

Relatively little is known about the influence of SIC on C dynamics in degraded areas. The addition of amendments such as gypsum, green manure and glucose to an alkaline sodic soil has been shown to aid in reducing soil pH and improving soil physical properties by increasing $CaCO_3$ solubility through various mechanisms (Chorom and Rengasamy 1997). The presence of free $CaCO_3$ can inhibit SOM decomposition through bridging of Ca^{2+} to SOM aggregates, and thus, protect it from microbial

degradation (Clough and Skjemstad 2000). It has been suggested that $CaCO_3$ can control the decomposition of POC through stabilisation of relatively undecomposed plant debris (Golchin *et al.* 1994). Higher contents of active $CaCO_3$ and amorphous Al and Fe act to stabilise fresh and humified organic materials by forming complexes with organic molecules, leading to high organic C, lower C:N ratios and longer retention times (Baldock and Nelson 2000). The removal of Ca from a soil stimulates the decomposition of organic matter and mineralisation of N, while its addition inhibits the release of CO_2 and promotes the stabilisation of soil structure (Oades 1988) due to the formation of Ca-organic linkages (Baldock and Nelson 2000). Where soils are high in Ca, precipitation of carbonates can occur with a decrease soil moisture and increased evapotranspiration, an increase in ion concentration, a decrease in the partial pressure of CO_2 or a rise in pH, as shown in Equation 2.4 (Lal and Kimble 2000b). Conversely, the addition of organic matter can dissolve carbonate due to the production of CO_2, favouring the left side of Equation 2.4.

$$Ca^{2+} + 2HCO_3^- \leftrightarrow CaCO_{3(s)} + CO_2 + H_2O \qquad \textbf{Equation 2.4}$$

2.5 Summary

A large proportion of the Australian landscape is currently affected by saline and sodic soils. These areas often coincide with agricultural areas, with the extent of saline and sodic soils likely to increase in the future. Currently, in terms of C accounting, data on how these salt-affected areas are related to C dynamics are virtually non-existent. Understanding of the roles salinity and sodicity play in the decomposition of organic matter needs to be improved if these knowledge gaps are to be addressed. The conflicting results reported in this chapter are most likely the result of the overall balance between the opposing effects of salinity, sodicity and the behaviour of organic matter. These processes are dependent on factors such as the chemical properties of the soil, the amount and nature of added organic materials and their interactions with inorganic colloids, the degree of mechanical disturbance, the amount and nature of SOM, and other soil characteristics such as clay content. While the addition of organic materials has usually resulted in an improvement in soil structure, the results are not always clear-cut in sodic and saline soils. These issues will need to be addressed if C cycling in these degraded areas is to be fully understood. This thesis aims to address

these knowledge gaps in relation to SOC stocks and fluxes in south-eastern Australia, by studies in the field and under controlled conditions, as described in Table 1.1. Chapter 3 investigates the behaviour of the labile C pool in a vegetated soil following leaching with saline and sodic solutions. This will determine the effects of increasing salinity and sodicity on the SMB and soil respiration rates.

CHAPTER 3: SOIL RESPIRATION AND SOIL MICROBIAL BIOMASS IN SOILS TREATED WITH A RANGE OF SALINE AND SODIC SOLUTIONS

3.1 Introduction

Increasing soil salinity and sodicity are serious land degradation issues in Australia, which are predicted to increase in importance in the future. Recently, focus has centred on issues related to dryland salinity, with the main cause being largely attributed to the broadscale clearing of native deep-rooted perennial vegetation, as described in Section 2.2.1, and its replacement with shallow-rooted annual crops and pastures. This alters the hydrologic balance and mobilises salts in the landscape. Irrigation salinity also has the potential to become more apparent in the future as water use for agriculture continues and the area of irrigation increases. The use of lower quality groundwater and wastewater with higher levels of soluble salts, particularly those which are dominated by Na, will increase as high quality water of low EC and SAR is allocated to urban water supply (Surapaneni and Olsson 2002). Under current land use, the area affected by secondary salinisation and sodication is likely to increase, especially where salts dominated by Na^+ accumulate in the soil profile. This will cause reductions in crop and pasture production (Rogers 2002). For example, in the Murray-Darling Basin, where highly saline-sodic groundwater is used for irrigation during summer periods, soil EC and ESP have increased, while winter leaching by low salinity rainfall reduces soil EC and increases ESP as soluble salts are leached from the soil profile (Figure 2.1). This results in dispersion and reductions in permeability (Bethune and Batey 2002). The majority of irrigated soils in the region suffer from sodic subsoils with low hydraulic conductivity, which can cause salts to build up over time; this is known as transient salinity (Rengasamy 2006).

Few studies have examined the effects of salinity and/or sodicity on soil biological processes, and those available show contradictory results (eg. Chander *et al.* 1994; Laura 1973; 1976; Nelson *et al.* 1996; Rietz and Haynes 2003; Sarig *et al.* 1993), as described in Section 2.4. In particular, little is known about how the processes of salinisation and sodification impact on the SMB and microbial activity. This chapter examines the effects of a range of salinity (EC) and sodicity (SAR) levels in soil

solution systems on labile C in different layers of a soil profile from the Southern Tablelands region of NSW. Effects due to different levels of EC and SAR on the SMB and soil respiration rates were assessed under controlled temperature and moisture conditions to assess the effects of salinity and sodicity on the dynamics of soil carbon.

A number of methods exist to determine soil respiration, both in the field and under controlled conditions in the laboratory. However, currently a standard method does not exist. There are two commonly used soil respiration methods, i) dynamic chamber method which provides an instantaneous measurement of CO_2 evolution at a particular time, and ii) static chamber method, which absorbs CO_2, and gives a measurement that has been integrated over a longer time period usually ranging from one to several days.

The dynamic method involves the use of an infra-red gas analyser (IRGA) to which air in the chamber is actively analysed for CO_2. The IRGA measures the rate of change in CO_2 concentration in the headspace of the incubation chamber. Whilst useful for taking measurements to determine diurnal variations in CO_2 evolution, it is difficult to integrate measurements over longer time periods unless very large numbers of measurements are taken over a 24 hr period (Jensen *et al.* 1996). One notable drawback with the use of an IRGA is the cost associated with its purchase, if it is required.

Static methods use an alkali trap such as KOH, NaOH or soda lime to trap evolved CO_2. In the case of KOH or NaOH solution, the amount of CO_2 evolved is determined by titration against standard HCl (Anderson 1982), while with the use of soda lime traps, CO_2 evolution is determined by weight gain (Edwards 1982; Grogan 1998). In a comparison of a static, with NaOH, with a dynamic method, Jensen *et al.* (1996) found large spatial variability with the use of both methods which required large numbers of replicates. The observed variability was related to variability in water content, soil temperature and water evaporation in the field. Minderman and Vulto (1973) compared the use of soda lime with KOH and determined that both techniques were suitable for laboratory use over long observation periods of more than 15 hours.

In this study, soil respiration was determined under controlled conditions in the laboratory with the use of soda lime traps. The soda lime absorption technique is a relatively inexpensive and simple method which allows for a large number of replicates to be rapidly analysed. Whilst the method has been criticised for its high variability in

the past, Keith and Wong (2006) have shown that the soda lime technique can be reliable if used under the correct conditions described in their paper, with a 1:1 relationship found when compared with measurements made using an IRGA. The method is able to integrate the mean CO_2 flux over a longer time period, rather than taking a number of transient measurements. The soda lime absorption technique, and all the methods described above, was established for the determination of soil respiration in the field. However, the soda lime method, as with the other static methods described, is easily adaptable to a laboratory-based study, such as that used in Bauhus *et al*. (2002).

3.2 Materials and Methods

3.2.1 Site Description

The profile was located on a property, "Tarcoola" in Bevendale, approximately 40 km south west of Crookwell (34 30' 45" S, 149 05' 00" E, 510 m a.s.l), in the Southern Tablelands region of NSW (Figure 3.1). The locality is underlain by undifferentiated Ordovician and Silurian metasediments (Hird 1991). The soil profile sampled was a Yellow Sodosol (Isbell 1996). The area was dominated by red grass (*Bothriochloa* spp) and fenced off from stock (Plate 3.1). The profile consisted of an A horizon of a sandy loam overlying a B horizon which was a sandy clay loam.

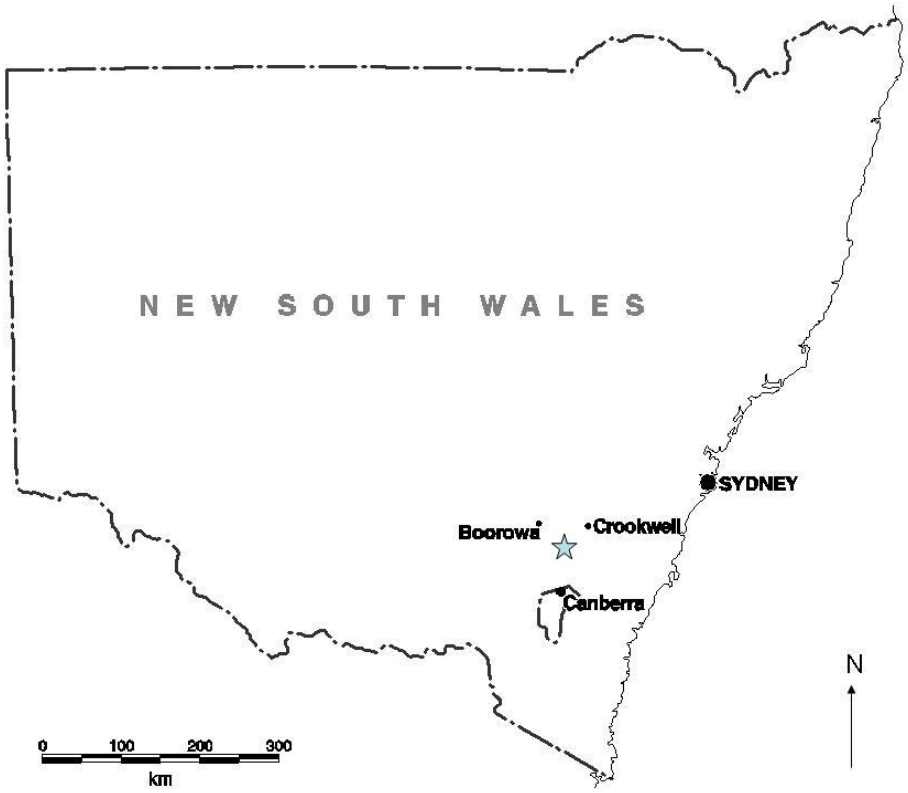


Figure 3.1 Star indicates the location of the property "Tarcoola" in Bevendale, NSW.

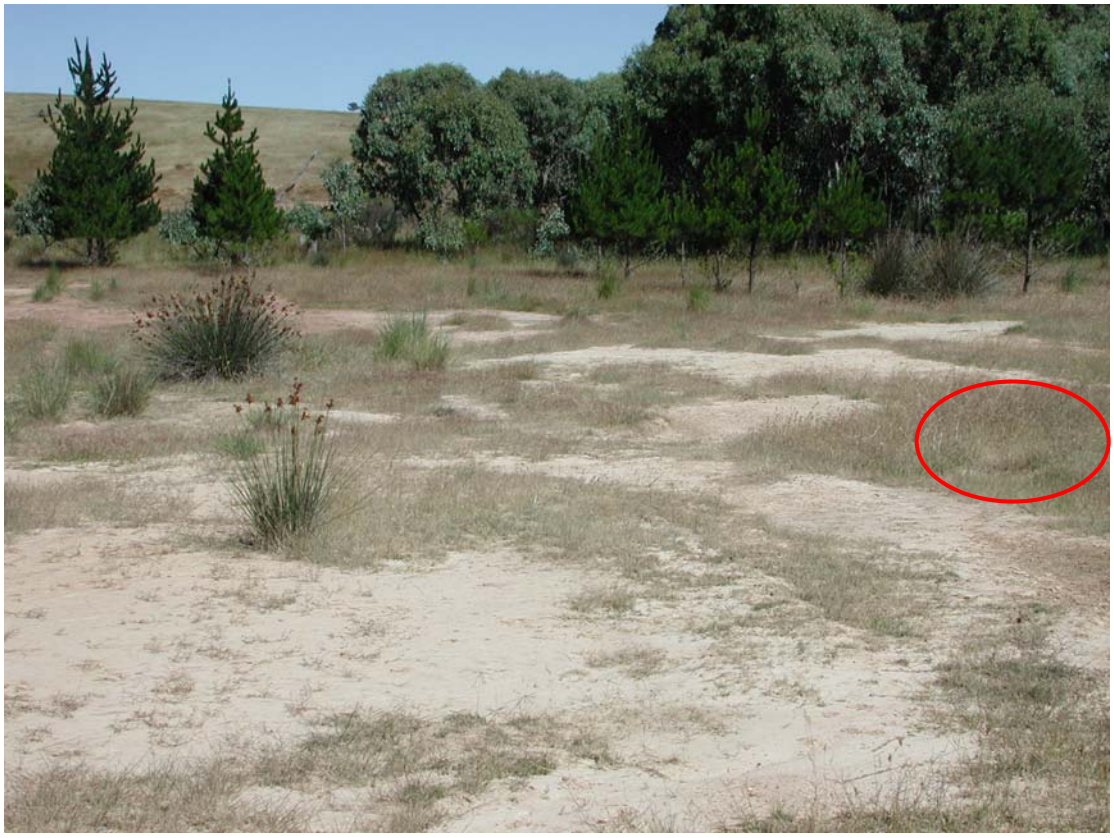


Plate 3.1 The paddock where the sampled profile was located at "Tarcoola." The red circle is an example of a "vegetated patch."

3.2.2 Field Sampling

Samples were taken from the 0-5, 5-10, 10-20, 20-30, and 30-50 cm depths of a vegetated soil profile, transported back to the laboratory in polyethylene bags and stored at 4^{o}C prior to analysis. Soils were sampled with a shovel from a soil pit at each depth interval. Bulk density cores were also taken from each depth as described in Section A1.1 in Appendix A.

3.2.3 Sample Preparation and Soil Chemical Analyses

Bulk density cores were oven dried at 105^{o}C for 24-hours, and from the known soil core volume and oven dry weight contained in the soil core, bulk density was calculated. This is described in detail in Appendix A. EC, pH and soluble cations were determined in 1:5 soil:water extracts. Soluble cations in the 1:5 soil:water extracts were analysed by inductively coupled plasma atomic emission spectroscopy (ICP-AES). Exchangeable cations were extracted by using 1 M ammonium acetate (CH_3COONH_4) extracts buffered to a pH of 7 and also determined by ICP-AES. A more detailed description of the analysis is found in Appendix A. The sodicity of the samples was determined by calculating the SAR from the soluble cations according to Equation 2.2, and ESP, from the exchangeable cations according to Equation 2.1.

Organic C, total N and total S were determined by high temperature combustion on a CNS LECO-2000 analyser. The samples were not pre-treated with acid prior to organic carbon analysis as the soil pH values (pH < 7) indicated that carbonates were not expected to be present. Particle size analysis was undertaken using the hydrometer method (Bouyoucos 1936).

3.2.4 Soil Biological Analyses

Soils that were analysed for microbial biomass and respiration were initially sieved without drying (field moist) through a 5 mm sieve. Six salt solutions of known EC and SAR values were prepared using a combination of 1 M NaCl and 1 M $CaCl_2$ stock solutions. The salinities of the solutions were 0.5, 10 and 30 dS/m, and were combined with two SAR values of 1 and 30 in a factorial design. These salt solutions were termed low-, mid- and high-salinity and low- and high-sodicity, respectively. The relative

volumes of the respective NaCl and $CaCl_2$ salt solutions added to achieve the range of final salinities and SARs are shown in Table 3.1; a total of 1 L of each solution was prepared. Distilled water was used in place of the low salinity-low sodicity solution as a control, giving a total of six solutions used for leaching (Figure 3.2). The salt solutions used for leaching were standard solutions, and were not intended to give similar EC and SAR values in the soil. More specifically, the following solutions were used:

- Distilled water (control)
- Low-salinity high-sodicity of EC 0.5 and SAR 30 (EC0.5 SAR30)
- Mid-salinity low-sodicity of EC 10 and SAR 1 (EC10 SAR1)
- Mid-salinity high-sodicity of EC 10 and SAR 30 (EC10 SAR30)
- High-salinity low-sodicity of EC 30 of SAR 1 (EC30 SAR1)
- High-salinity high-sodicity of EC 30 and SAR 30 (EC30 SAR30)

Table 3.1 Volume of 1 M NaCl and 1 M $CaCl_2$ used for leaching

Treatment	1 M NaCl (mL)	1 M $CaCl_2$ (mL)
Control	0.0	0.0
EC 0.5 SAR 30	45.4	4.6
EC 10 SAR 1	6.8	93.1
EC 10 SAR 30	84.2	15.8
EC 30 SAR 1	12.0	288.0
EC 30 SAR 30	298.0	2.0

The soils were treated with the above solutions as follows. Approximately 5 kg of the <5 mm fraction of soil were placed into a 9.6 L bucket with holes in the base, with filter paper placed over the holes. The soils were leached once a day for three days, initially with 1 L of solution on the first day, and 0.5 L solution on the two subsequent days before being allowed to equilibrate for 72 hours. Each depth layer was treated separately. The soils were then maintained in closed containers at a constant temperature environment at 25^{o}C and analysed for respiration and SMB, as described below.

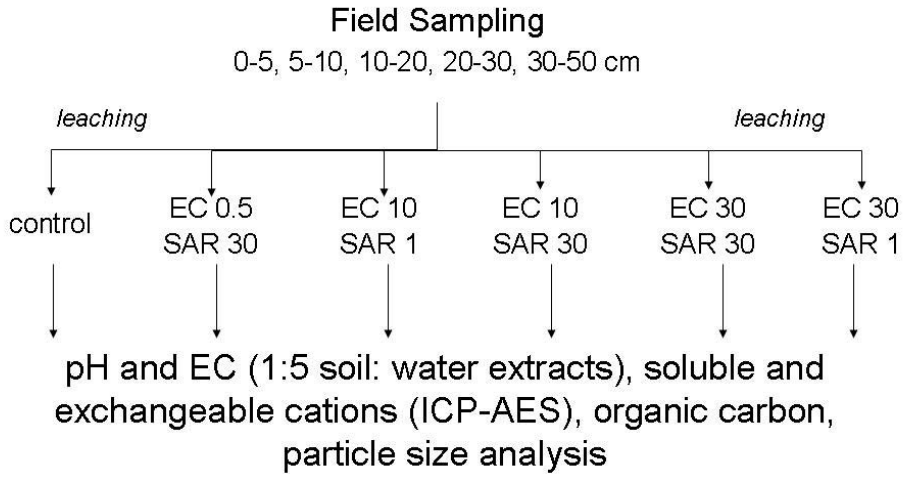


Figure 3.2 Sample preparation prior to laboratory analysis

3.2.4.1 Soil Respiration

Soil respiration was measured according to a modification of the method originally developed by Edwards (1982). Approximately 100 g of soil was weighed into 150 mL screw top jars without lids and placed into air-tight 1.75 L polycarbonate containers. The polycarbonate containers were sealed with duct tape to ensure that no leakage occurred. In addition to the soil, the polycarbonate container also had a petrie dish with 25 g of soda lime granules to trap the CO_2 evolved, and a small vial of approximately 15 mL of water to maintain the humidity (Plate 3.2). Three blanks were also prepared for every run (ie. every two weeks) according to the method described above without a soil sample in the polycarbonate container to account for the amount of CO_2 absorbed by the soda lime in the headspace of the chamber and chamber leakage. Soda lime reacts with CO_2 according to Equations 3.1a and 3.1b. The soda lime traps were oven dried at 105^{o}C for 16 hours prior to incubation. 4 mL of water was then added, as the reaction between hydroxide and CO_2 is facilitated by the presence of water. The soda lime traps were oven-dried prior to wetting up as soda lime also absorbs water when stored. Therefore, a constant amount of water could be added following oven-drying. The soils were left to incubate for a period of 12 weeks and were analysed for CO_2 evolution at biweekly intervals, with a new soda lime trap placed in the incubation chamber and analysed every two weeks.

Respiration is facilitated by the soils being maintained at constant moisture content; however, the soils tended to dry out during the respiration measurements. Therefore, moisture loss was determined gravimetrically at four-weekly intervals, with water added

to bring the soils up to their original weight. Moisture loss was most likely caused by the absorption of water by the polycarbonate container, as plastics tend to be porous to water vapour but not CO_2, resulting in a higher permeability to water than to CO_2. Because the humidity was maintained in the incubation chamber with a vial of water, a gradient was established such that there was high humidity inside the chamber, and ambient and lower humidity outside of the chamber, thereby promoting water loss through the chamber walls (S.C. Wong, pers. comm.).

The traps were oven dried at 105^{o}C for 24 hours after removal from the incubation chambers, and reweighed. The amount of CO_2 evolved was determined according to Equation 3.2. A correction factor of 1.69 was used to correct for chemical water loss during the drying process following its reaction with CO_2. Evolution of CO_2 was then expressed per kilogram of soil, according to Equation 3.3. All treatments were undertaken in triplicate.

Plate 3.2 Experimental set-up used for analysis of soil respiration; incubation chamber with soil sample, vial of water and soda lime trap

$2NaOH + CO_2 \rightarrow Na_2CO_3 + H_2O$ **Equation 3.1a**

$Ca(OH)_2 + CO_2 \rightarrow CaCO_3 + H_2O$ **Equation 3.1b**

$CO_2\ (g) = [(SL_a\text{-}SL_b) – (B_2\text{-}B_1)] * 1.69$ **Equation 3.2**

Where SL_a = weight of soda lime after incubation,

SL_b = weight of soda lime before incubation

B_2 = weight of blank soda lime after incubation

B_1 = weight of blank soda lime before incubation

mg-CO_2-C /kg soil = [CO_2 (mg) evolved / weight of oven dried soil (kg)]*(12/44) **Equation 3.3**

3.2.4.2 Soil Microbial Biomass

Soil microbial biomass was measured weekly by the chloroform fumigation procedure described in Vance *et al*. (1987). The technique involves measuring the difference in the DOC contents of fumigated and unfumigated samples of soil. The fumigated samples were prepared by weighing 50 g of soil into a 100 mL beaker at weekly intervals. The soil was placed in a dessicator with 25 mL of amylene-stabilised chloroform ($CHCl_3$) and wet filter paper to maintain the humidity within the chamber. The dessicator was evacuated until the chloroform started to boil; evacuation continued for a further two minutes. The dessicator was then placed in the dark for 24 hours. Concurrently, the non-fumigated soil was prepared by weighing 50 g of soil into a 500 mL bottle at weekly intervals, followed by the addition of 200 mL of 0.3 M K_2SO_4 solutions. After shaking for 30 min on a rotary shaker, the suspension was filtered through Whatmans No. 42 filter paper. On the same day, 8 mL of the filtered extract was placed into a conical flask with 10 mL of concentrated H_2SO_4, 5 mL of 85 % H_2PO_3 and 2 mL of 0.0667 M $K_2Cr_2O_7$. The mixture was heated on a hot plate for approximately 20 minutes and allowed to cool prior to being titrated against 0.033 M ferrous ammonium sulfate solution ($(NH_4)_2SO_4FeSO_4.6H_2O$) with ferroin indicator (1, 10-phenanthroline-ferrous sulfate solution). After 24 hours, the beaker of chloroform and filter paper were removed from the dessicator before being repeatedly evacuated to remove the excess chloroform. The fumigated samples were subjected to the same treatment as the non-fumigated samples. The amount of SMB-C present in the samples was determined by the difference between the extracted carbon in the fumigated samples and the unfumigated samples (E_C) expressed as mg-C/kg oven dry soil according to Equation 3.4. A constant of 2.64 is used to correct for the DOC that is not extracted (Vance *et al.* 1987).

SMB-C (mg-C/kg) = 2.64 E_C **Equation 3.4**

All measurements are expressed as oven-dry equivalent weights of soil. Three replicates of each soil were determined for SMB-C.

3.2.4.3 Microbial Indices

The specific respiration rate, qCO_2, was determined according to Equation 3.5 at the end of the 12-week incubation period to provide an indication of the effects of EC/SAR on microbial activity.

qCO_2 (mg CO_2-C/mg SMB-C/day)= r/SMB **Equation 3.5**

Where r = respiration rate (mg CO_2-C /kg/day)

SMB-C = soil microbial biomass-C (mg-C/kg)

The microbial quotient, C_{mic}:C_{org}, was determined as the ratio of SMB-C (mg/kg) to SOC (mg/kg) expressed as a percentage (%) at the end of the 12-week incubation period.

3.2.5 Statistical Analysis

Data were analysed using the GENSTAT 8.0 statistical analysis program (Payne 2005). Differences found between the different treatments were subjected to an analysis of variance (ANOVA). The block structure was given by depth within EC by SAR, within replicate, within week, and the treatment structure by depth, EC, SAR, week and their interactions. Differences found in the SMB and respiration over the 12 week incubation period were analysed by residual maximum likelihood (REML), as the two factors were found to be significantly correlated over time (P<0.05). The fixed effects were depth, EC, SAR and their interaction, and random effects were the interaction of depth, EC, SAR and week. Where significant differences were found (P<0.05), data were subjected to least significant difference testing (LSD). The SMB data were square-root transformed to satisfy the assumptions for ANOVA, with back-transformed means presented.

3.3 Results

3.3.1 Soil Characterisation

Soil chemical properties of the bulk soil are shown in Table 3.2. The pH was acidic to slightly acidic throughout the profile. The soil profile was non-saline (EC(1:5) < 1.5 dS/m; Murphy and Eldridge 1998). A large decrease in EC occurred between 0-5 cm and 5-10 cm depth, while SAR increased with depth to 30 cm. ESP increased with depth to 50 cm, while SOC, total N and total S showed highest values at the surface, and decreased with depth.

The particle size distribution and bulk density are shown in Table 3.3. Bulk density displayed an increase with depth to 30 cm, after which it decreased slightly. The soil texture was a sandy loam at the surface, grading to a sandy clay loam at depth.

Table 3.2 Soil chemical properties of the original soil before treatment with the EC and SAR solutions

Depth (cm)	$pH_{1:5(H2O)}$	$EC_{1:5}$ (dS/m)	SAR	ESP	SOC (%)	Total N (%)	Total S (%)
0-5	4.40	0.31	1.25	1.17	3.87	0.298	0.029
5-10	5.23	0.17	2.37	4.91	1.88	0.136	0.011
10-20	5.32	0.15	2.51	13.12	0.99	0.060	0.007
20-30	5.68	0.16	4.22	15.37	0.74	0.040	0.004
30-50	5.95	0.15	3.60	15.94	0.48	0.029	0.003

Table 3.3 Particle size distribution and bulk density of the bulk soil

Depth (cm)	Sand (%)	Silt (%)	Clay (%)	Soil Texture	Bulk Density (Mg/m^3)
0-5	71.0	13.6	14.5	Sandy loam	1.19
5-10	73.7	10.5	20.4	Sandy loam	1.48
10-20	71.7	3.6	24.0	Sandy clay loam	1.61
20-30	68.3	12.2	21.0	Sandy clay loam	1.68
30-50	71.6	5.5	22.1	Sandy clay loam	1.61

Following leaching with the solutions containing combinations of EC and SAR, the pH, EC, SAR and ESP values were measured again after the equilibration period of three days. Figure 3.3 shows that, following leaching, the pH of all combinations of EC and SAR in the 0-5 cm increased relative to the bulk (original) soil. However, below 5 cm, the pH of all treatments was different to that of the untreated soil, but no clear pattern emerged.

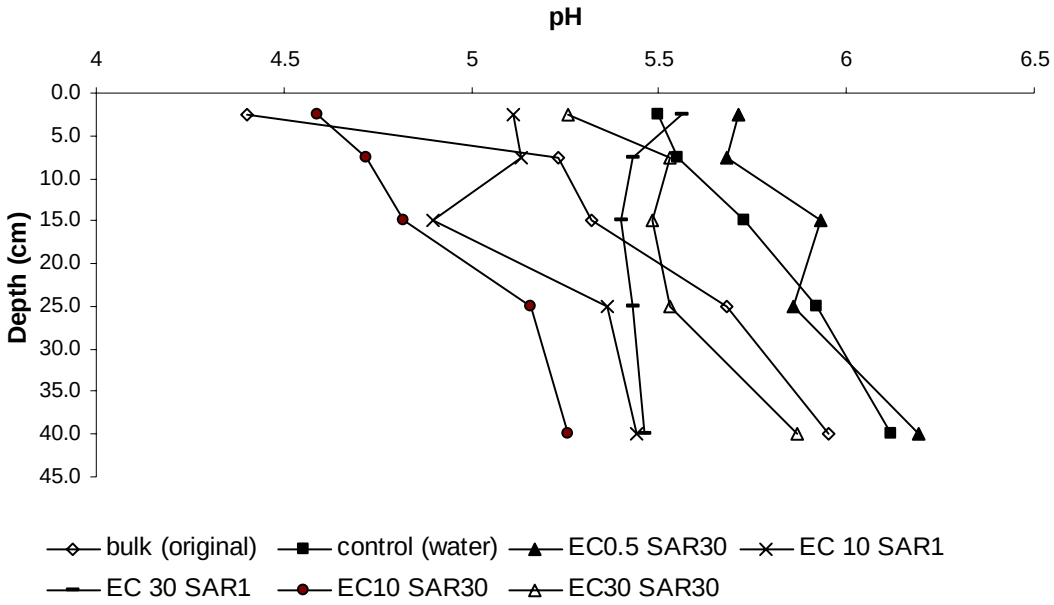


Figure 3.3 $pH_{1:5(H2O)}$ of the leached soils after equilibration

Figure 3.4 indicates that the original soil profile had a constant and low EC with depth. Following leaching, there was a decrease in EC with depth in the control and low-salinity high-sodicity treatments. The high-salinity treatments showed the greatest increases in EC following leaching, while the mid-salinity treatments showed a smaller increase.

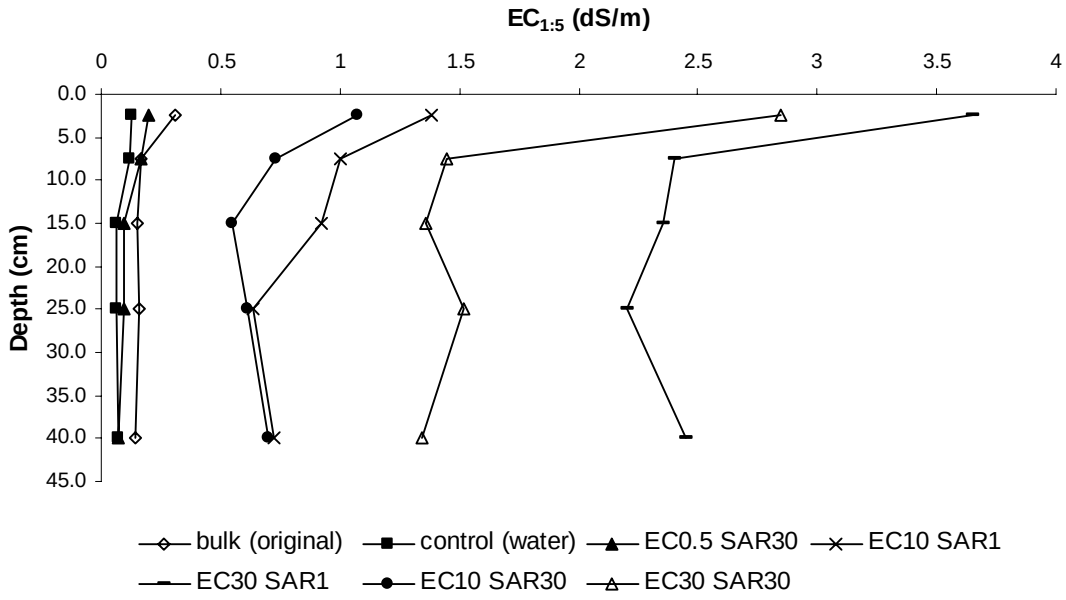


Figure 3.4 $EC_{1:5}$ of the leached soils after equilibration

Figure 3.5 indicates that the SAR of the original soil solution increased with depth. Following leaching, the SAR in the 0-5 cm layer increased in three of the treatments

(mid-salinity low-sodicity, mid-salinity high-sodicity, and high-salinity high-sodicity), while the SAR was greater than the bulk soil at all depths in the two high sodicity treatments. The soil solutions of the treated soils did not display the same EC or SAR as the salt solutions they were originally leached with.

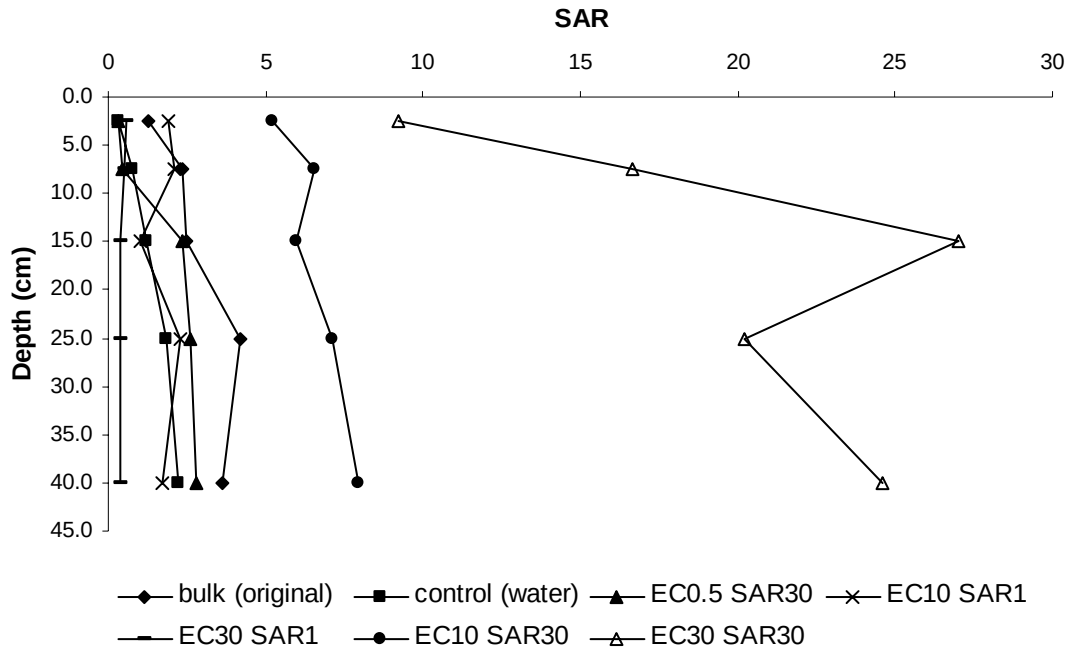


Figure 3.5 SAR of the leached soils after equilibration

Figure 3.6 indicates that the original soil increased in ESP with depth, and would be considered sodic from 10 cm to 50 cm (ESP > 6). Figure 3.6 also indicates that following leaching, the high-salinity high-sodicity and mid-salinity high-sodicity treatments showed the greatest increases in ESP. Leaching with distilled water (control) increased the ESP to a depth of 10 cm, before causing a decrease between 10 cm and 50 cm relative to the bulk soil. The mid-salinity low-sodicity treatment showed a similar trend, while the high-salinity low-sodicity treatment showed a lower ESP compared to the bulk soil at all depths with the exception of the 0-5 cm layer.

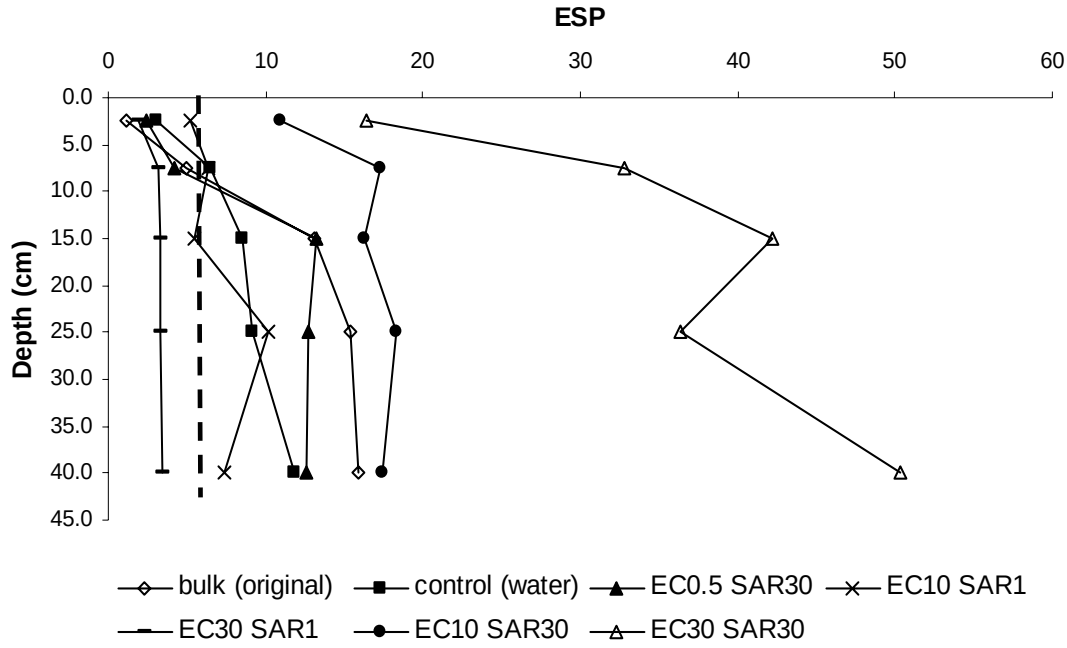


Figure 3.6 ESP of the leached soils after equilibration.
Note: The dashed line indicates ESP = 6

3.3.2 Soil Respiration

An example of the calculation of CO_2 evolution is shown in Table B1 in Appendix B. Because respiration was minimal at depths below 10 cm (soda lime sample ≈ blank soda lime), it was below the detection limit of the soda lime method, and hence, too low to measure with confidence at depth. Therefore, only results from the 0-5 cm and 5-10 cm layers are shown. Soil respiration was significantly different with EC (P<0.001) and SAR (P<0.01). There were also significant interactions between EC and SAR (P<0.01). Data pooled over the 12 weeks to 10 cm showed that respiration was highest in the control treatment (EC 0.5 SAR 1; Table 3.4). Respiration was lowest in the mid-salinity treatments, with the respiration rate significantly lower in the mid-salinity high-sodicity treatment compared to the mid-salinity low-sodicity treatment.

Table 3.4 Interaction of the treatment effects on soil respiration rates (CO_2-C mg/$kg_{OD\ soil}$/week) for 0-10 cm depth soil.

EC	SAR	
	1	**30**
0.5	80.0[a]	55.9[ab]
10	5.4[c]	1.3[d]
30	27.0[b]	37.9[b]

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Different letters within a column or within a row represent a significant difference (P<0.05).

Figures 3.7a and 3.7b shows the cumulative soil respiration measured at two-week intervals for the six treatments at the 0-5 and 5-10 cm depths, respectively. The control and low-salinity high-sodicity treatments showed the highest cumulative respiration rates with 2246 mg CO_2-C/kg and 1947 mg CO_2-C/kg evolved, respectively, at the end of the 12-week incubation period in the 0-5 cm layer. The mid-salinity high-sodicity treatment had the lowest cumulative respiration rate (705 mg CO_2-C/kg; Figure 3.7a). Similarly, in the 5-10 cm layer, the mid-salinity high-sodicity treatment also showed the lowest respiration rate (64 mg CO_2-C/kg; Figure 3.7b). In the same layer, the low-salinity high-sodicity treatment had a significantly higher rate of respiration (867 mg CO_2-C/kg) at the end of the 12-week incubation period compared to the other treatments. The cumulative respiration rates in the control and high-salinity high-sodicity treatments were similar after 12 weeks (551 mg CO_2-C/kg and 524 mg CO_2-C/kg, respectively), as were the respiration rates in the mid-salinity low-sodicity and high-salinity low-sodicity treatments (280 mg CO_2-C/kg and 309 mg CO_2-C/kg, respectively).

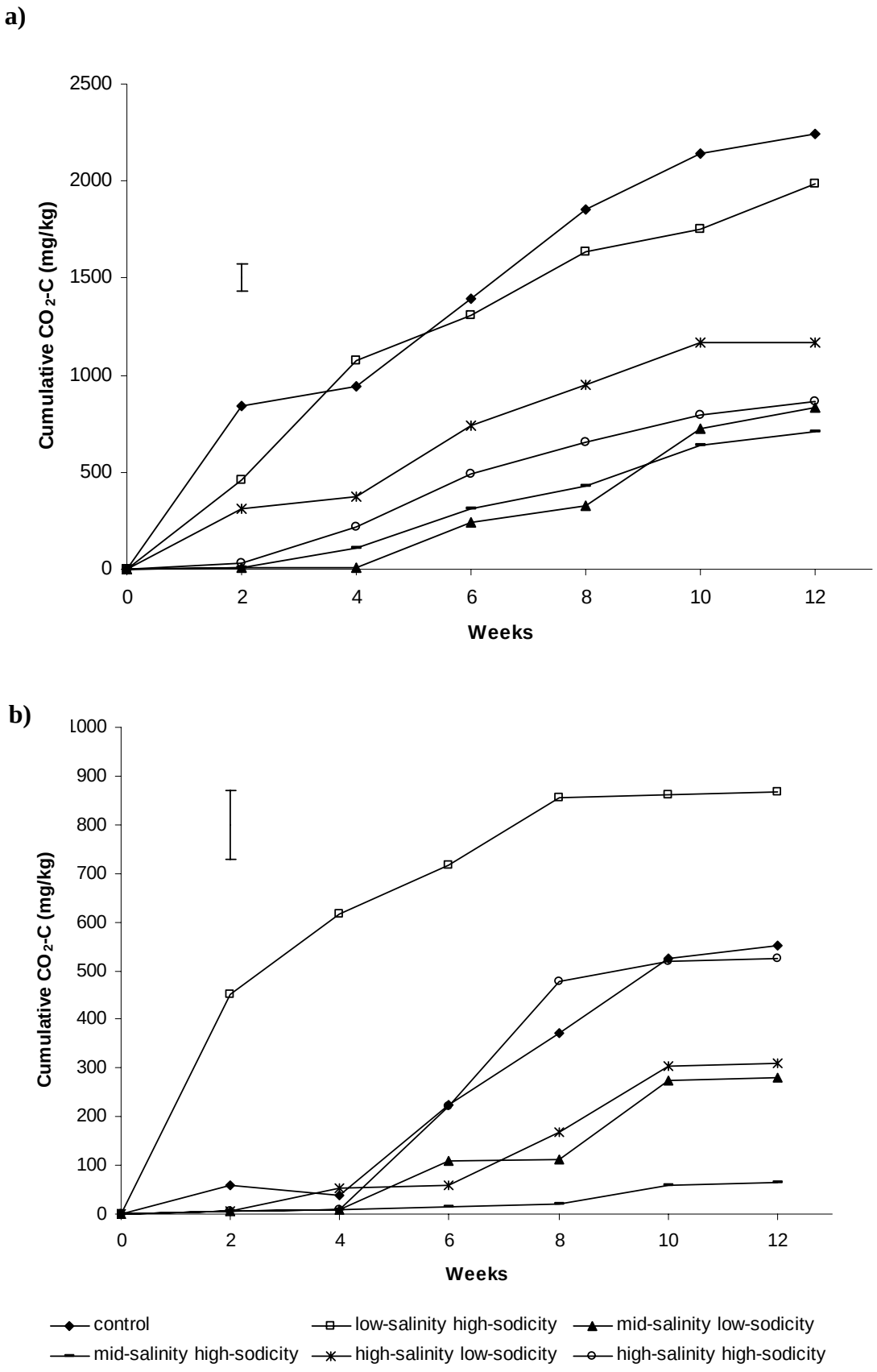


Figure 3.7 Effects of the different EC/SAR treatments on cumulative respiration rates over the 12 week incubation period at a) 0-5 cm and b) 5-10 cm.

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Vertical bar represents the SED.

3.3.3 Soil Microbial Biomass

SMB data pooled over the 12 weeks, showed the SMB decreased significantly with depth to 30 cm (P<0.001), increased significantly with increasing EC (P<0.001), and decreased significantly with increasing SAR in the mid-salinity (EC 10) treatment (P<0.001; Tables 3.5 and 3.6). There was also a highly significant interaction between EC and SAR (P<0.001; Table 3.6).

Table 3.5 Effects of the different EC/SAR treatments on SMB with depth, EC and SAR.

Depth (cm)	**0-5**	**5-10**	**10-20**	**20-30**	**30-50**
SMB-C (mg/kg)	803.72[a]	396.41[b]	236.24[c]	164.10[d]	145.93[d]
EC	**0.5**	**10.0**	**30**		
SMB-C (mg/kg)	165.38[a]	311.52[b]	510.76[c]		
SAR	**1**	**30**			
SMB-C (mg/kg)	337.82[a]	289.68[b]			

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Different letters within a row indicate a significant difference (P<0.001).

Table 3.6 Interaction of the treatment effects on the SMB (mg/kg).

EC	**SAR**	
	1	**30**
0.5	158.46[a]	172.42[a]
10	352.69[c]	273.01[b]
30	565.44[d]	458.73[d]

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Different letters within a column or within a row represent a highly significant difference (P<0.001).

Significant interactions also occurred between depth, EC and SAR (P<0.05; Figure 3.8). The control treatment had the lowest levels of SMB at the surface. SMB declined in all treatments with depthexcept in the high-salinity low-sodicity treatment, which declined to 30 cm, then increased in the 30-50 cm layer.

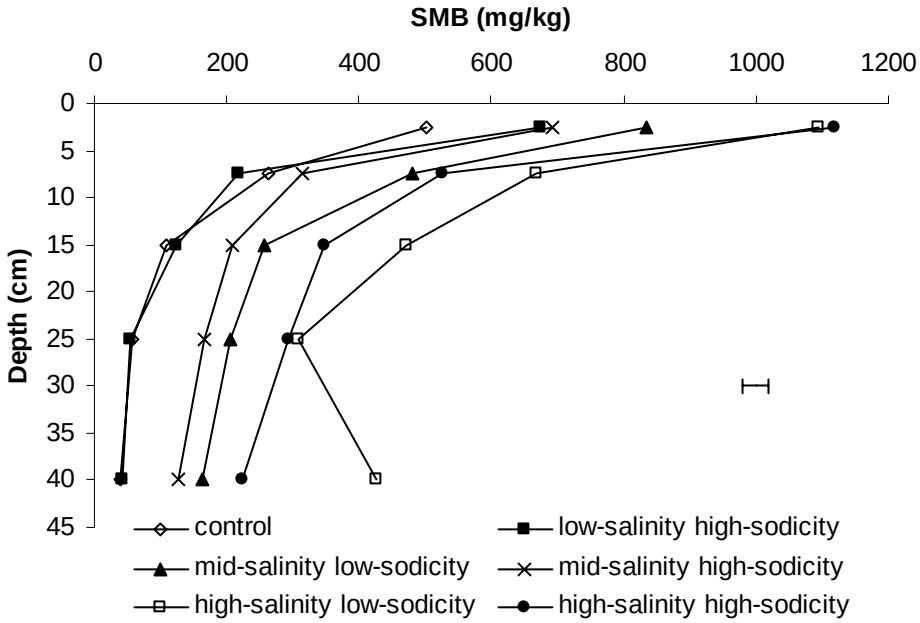


Figure 3.8 Treatment effects on the SMB with depth.
Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Horizontal bar represents the SED.

Figure 3.9 (a, b, c, d, and e) shows the effect of the EC/SAR treatments on the SMB for the 0-5, 5-10, 10-20, 20-30 and 30-50 cm depths respectively. There were significant interactions between treatments, depths and time over the 12-week incubation period (P<0.01). The high-salinity treatments generally displayed the highest levels of SMB over the 12-week incubation period, while the control and low-salinity treatments displayed the lowest levels of SMB. The SMB in the high-salinity treatments also increased at Week 1 at all depths, while the mid-salinity, low-salinity and control treatments all decreased. However, the mid-salinity low-sodicity treatment increased in SMB at depth (from 10 -50 cm). In the 0-5 cm and 5-10 cm layers, there is a gradual decrease in SMB over the 12 weeks in the control, low- and mid-salinity treatments.

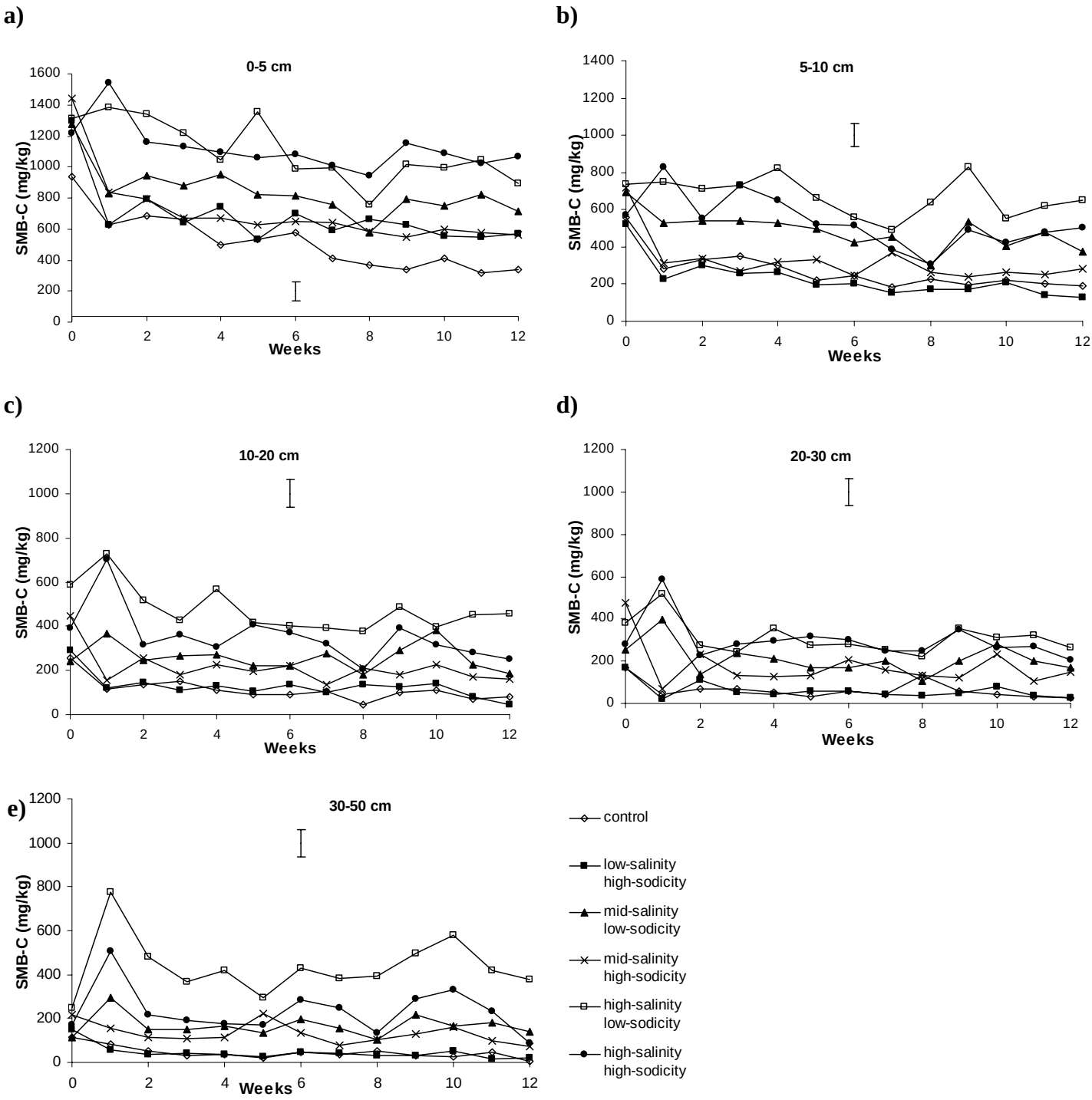


Figure 3.9 Treatment effects on the SMB over the 12-week incubation period at a) 0-5 cm, b) 5-10 cm, c) 10-20 cm, d) 20-30 cm and e) 30-50 cm.

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Vertical bar represents the SED

3.3.4 Microbial Indices

Table 3.7 shows the effects of EC and SAR on qCO_2 and C_{mic}:C_{org} in the 0-5 and 5-10 cm layers at the end of the 12-week incubation period. The qCO_2 was highest in the control (EC 0.5 SAR 1) and low-salinity high-sodicity (EC 0.5 SAR 30) treatments at both depths. The qCO_2 decreased as EC increased in the 0-5 cm layer. Similarly, the qCO_2 was lower in the mid- and high-salinity treatments in the 5-10 cm layer compared to the control and low-salinity high-sodicity treatments. The C_{mic}:C_{org} increased with increasing EC in both the 0-5 and 5-10 cm layers (Table 3.7).

Table 3.7 Effects of the different EC/SAR treatments on qCO_2 and C_{mic}:C_{org}

Depth (cm)	EC (salinity)	SAR (sodicity)	qCO_2 (mg CO_2-C/d/mg SMB-C)	C_{mic}:C_{org}
0-5	0.5 (control)	1 (control)	0.080	0.88
	0.5 (low)	30 (high)	0.060	1.02
	10 (mid)	1 (low)	0.014	1.86
	10 (mid)	30 (high)	0.015	1.47
	30 (high)	1 (low)	0.013	2.31
	30 (high)	30 (high)	0.010	2.77
5-10	0.5 (control)	1 (control)	0.035	1.00
	0.5 (low)	30 (high)	0.080	0.68
	10 (mid)	1 (low)	0.009	1.99
	10 (mid)	30 (high)	0.003	1.52
	30 (high)	1 (low)	0.006	3.58
	30 (high)	30 (high)	0.012	2.72

3.4 Discussion

3.4.1 Effects of Leaching

Following leaching, pH values were altered but there was no distinct pattern. The decrease in EC throughout the profile following leaching with distilled water and in the low-salinity high-sodicity treatment was most likely due to soluble salts contained in the profile being leached out with a low EC solution. Conversely, the increase in EC following leaching with the higher salinity solutions (mid-salinity and high-salinity) was due to the addition of soluble salts from the leaching solutions. However, this effect is dependent on the initial EC values of the original soil, and occurred in this case because the EC values of the original soils were lower than those of the mid-salinity and high-salinity treatments.

The largest increases in sodicity, measured by SAR and ESP, occurred following leaching with the high-sodicity solutions combined with the mid- and high-salinities. In these solutions, the highest concentrations of soluble Na^+ in the leaching solutions increased the SAR of the soil solution and concomitantly increased the ESP of the exchange complex. Similarly, Crescimanno and De Santis (2004) found Na^+ is adsorbed by the exchange phase on soils when leached with solutions high in Na^+, while Ca^{2+} is displaced from soil as a result of Na-Ca exchange, indicating that Na can be progressively accumulated, enhancing sodification of soils. In this study, following leaching with the high-salinity low-sodicity solution, the addition of soluble Ca^{2+} provided excess Ca^{2+} for exchange, causing the resultant decrease in the SAR and ESP.

3.4.2 Measures of Biological Activity

In the current study, the qCO_2, the specific respiration rate, was lowest in the high-salinity treatment and highest in the low-salinity treatments. However, a study by Wichern *et al*. (2006) showed that the qCO_2 did not differ significantly with salt content, indicating that the microbial biomass was in a similar physiological condition, despite the salt content. They suggested that a microbial community previously prone to salinity has adapted to it. Similarly, a study by Anderson (1998) showed that irrigation with acidic waters to an already acidic soil did not affect the qCO_2, as the microbial population had already adapted to the conditions. Wardle and Ghani (1995) have suggested that while qCO_2 may provide a measure of the efficiency by which the SMB is utilising C resources, its use as an indicator of disturbance and stress can be confounding. They suggested that a reduction in stress by imposing a chemical disturbance may increase microbial efficiency and decrease qCO_2, but this was dependent on the nutrient status of the system in question. Increased stress has been shown to reduce qCO_2 (eg. Chander and Brookes 1991b) due to shorter life span and lower efficiency, or increase qCO_2 due to either a diversion of energy to maintenance rather than growth of the microbial population, or a shift in the bacteria to fungi ratio (Anderson and Domsch 1993). In this study, it is possible that the increase in the C_{mic}:C_{org} with increasing EC may reflect increasing substrate availability with increasing salt concentration for microbial synthesis but decreasing respiration; thus the qCO_2 may reflect a shift in population structure to one that is dominated by less active microorganisms with lower respiration rates compared to a population dominated by

more active mircroorganisms (Adu and Oades 1978; Sadinha *et al.* 2003). Alternatively, more SMB-C was respired as a proportion of SOC at low salinity than at high salinity, resulting in high respiration rates but low SMB-C at low salinity levels, compared to the high salinity treatments treatments. However, both the qCO_2 and C_{mic}:C_{org} should also be used with caution as they were determined at the end of the 12-week incubation period only, and can merely provide an indication on microbial activity due to EC and SAR.

While the SMB showed the largest increases in the high-salinity treatments in this study, biological activity will cease at very high salinities as a result of high osmotic pressure. McCormick and Wolf (1980) also found microbial activity ceased at a salt concentration of 100 mg/g (EC = 37.30 dS/m in 1:5 soil:water extracts). In the same study, the addition of NaCl at all concentrations inhibited respiration, including at the lowest rate of 0.25 mg NaCl/g, which gave an $EC_{1:5}$ of 0.19 dS/m. In the current study, salinity levels were well below those previously reported to cause microbial activity to cease, with the highest EC in the study measured at 3.65 dS/m in a 1:5 soil:water extract, following leaching with a solution at an EC of 30 dS/m. However, EC levels following leaching with the mid- and high-salinity solutions were higher than the 0.19 dS/m reported by McCormick and Wolf (1980) which decreased respiration.

It has been suggested by Beltran-Hernandez *et al*. (1999) that applying salt to a soil may deleteriously affect microorganisms not adapted to saline conditions, however, this does not appear to be the case in this study. The initial microbial biomass increased in the high-salinity treatments following the equilibration period as soil environmental conditions in terms of temperature and moisture were optimal, with the easily decomposable substrate being mineralised first. The gradual decline in SMB over the duration of the 12 week experimental period (Figure 3.9) may indicate that the system is reaching a steady state after the initial disturbance of salinisation and sodication. The microbial biomass may also be affected by gradual changes in the osmotic potential due to an increased concentration in salt from moisture loss, as water was not added to samples analysed for SMB during the incubation period. Simultaneously, as the period of the incubation increased, it is probable that the amount of easily decomposable substrate decreased, resulting in an increase in the proportion of substrate that is of a lower quality, or material that is physically protected in aggregates, and hence, more difficult to decompose. While the determination of changes in SOM chemistry may

have provided an indication as to whether these processes occurred, time and budgetary constraints precluded this measurement. It is therefore suggested that the microbial population adjusted to compensate for the substrate becoming increasingly more difficult to decompose. However, a study by Baldock and Oades (1989) showed that increasing the electrolyte concentration caused alterations in the rate of decomposition. At the end of the experimental period, they showed that the amount of material decomposed was similar between treatments as the change in rate was attributed to alterations in the osmotic effect. In the current study, the electrolyte concentration appeared to alter the rate of decomposition, but the extent of decomposition cannot be confirmed.

In the low-salinity high-sodicity treatment, the concentration of Na^+ available for exchange was not sufficient to cause problems associated with sodicity, and caused a decrease in ESP (Figure 3.6), indicating the availability of substrate was not related to increased dispersion. Figure 3.10 shows that the two effects related to increased dispersion and increased solubility of organic matter, both of which increase substrate availability to microbial population, can occur. Increasing sodicity can increase the disruption of microaggregates due to slaking and dispersion which alters soil physical properties (Rengasamy and Sumner 1998). Dispersion is most likely the dominant process causing increased substrate availability in those soils which had been leached with the high-sodicity solutions, particularly in the high-salinity high-sodicity treatment (Figure 3.10). Both macro- and microaggregates have been shown to contain organic matter in their cores (Tisdall and Oades 1982), which is physically protected from decomposition. Thus, under sodic conditions, SOC can be rapidly lost when these aggregates disperse, and the organic matter contained within the aggregates is available for decomposition. Conversely, increasing electrolyte concentration causes soil to flocculate, offsetting those effects caused by sodicity on a soil's physical properties (Shainberg and Letey 1984). However, sodic behaviour is dependent on the electrolyte concentration of the applied water (Quirk and Schofield 1955), with the electrolyte concentrations of the leaching solutions in the mid-salinity low-sodicity and the high-salinity low-sodicity treatments likely to be sufficient to prevent dispersion. In those soils which are likely to remain flocculated following leaching with the mid- and high-salinity solutions, the increased availability of substrate is probably due to the effects of the salts increasing the solubility of the organic matter present. However, time and budgetary constraints prevented the measurement of DOC.

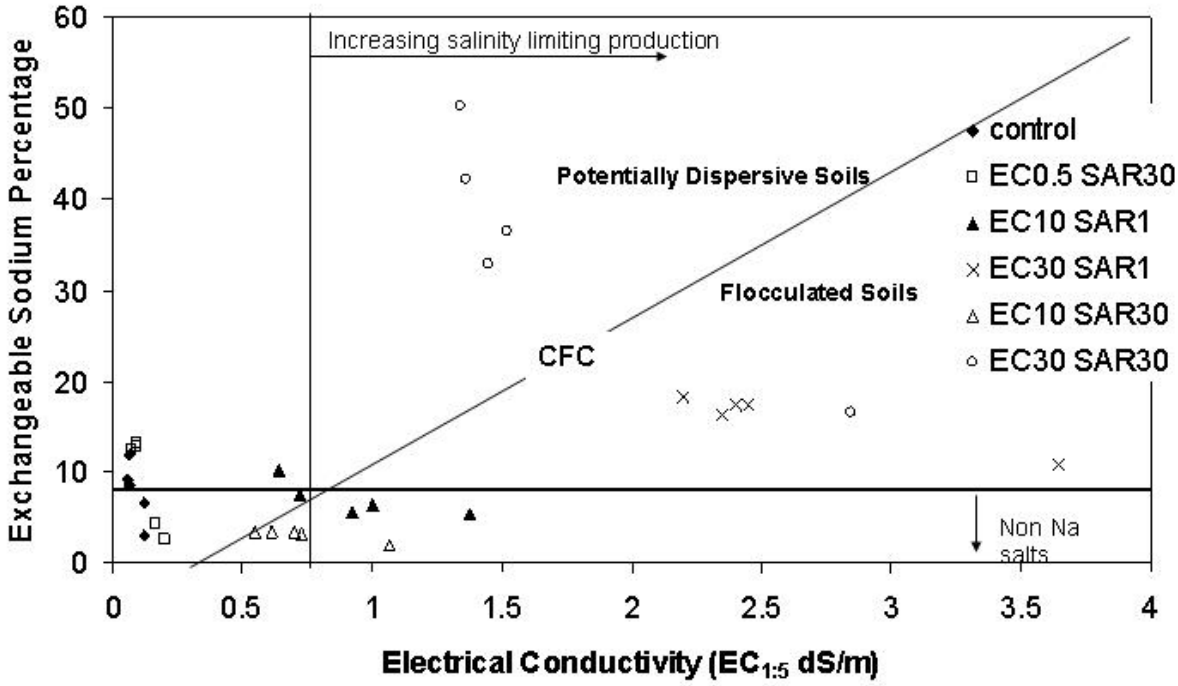


Figure 3.10 The likely effect of dispersion and aggregation following leaching with the EC/SAR treatments.
Note: CFC indicates the Critical Flocculation Concentration.
Source: Adapted from Rengasamy *et al*. (1984).

Respiration does not appear to correlate with SMB, although it does not necessarily need to follow the same trends. The low-salinity and control treatments showed the highest rates of CO_2 evolution, despite displaying the lowest levels of SMB throughout the incubation period. Respiration rates can be confounded by factors such as the substrate availability and the composition of the microbial population (Wang *et al.* 2003), which may be altered under different physicochemical conditions such that the size of the SMB may not reflect biological activities. Sarig *et al*. (1993) found a greater accumulation of SMB under saline irrigation water (EC 5 and SAR 10) compared to regular water (EC 1 SAR 10). This can be attributed to increasing osmotic stress causing an increase in the microbial population (Polonenko *et al.* 1981), and lower levels of C mineralisation, and hence, lower respiration rates.

Water-soluble C is considered to be the most active and immediately available organic substrate for the microbial population (Liu *et al.* 2006). Hence, additional water-soluble C can become available in all treatments due to the increase in moisture content

following leaching. This could result in the increased respiratory activities of the microbial population, with the highest rates of respiration occurring in the low-salinity and control treatments as osmotic stresses may play a role in the mid- and high-salinity treatments. The higher levels of SMB and lower rates of respiration in the high-salinity treatments may be the result of a shift in the composition of the microbial community, as suggested earlier. Rasul *et al*. (2006) found the proportion of fungi in the total microbial biomass to be lower in a saline soil compared to a non-saline soil. Similarly, Pankhurst *et al.* (2001) found that increasing salinity caused a shift towards a less active bacterial dominated community that was less diverse. However, the determination of any shifts in community structure is beyond the scope of this project.

3.4.3 Salinity and Sodicity Effects on Soil Carbon Dynamics

Two competing processes occur in saline and sodic soils which affect microbial activity: increasing osmotic potential as salt concentration increases (low SMB turnover and low respiration rate) and increasing availability of organic matter (high SMB turnover and high respiration rate). Availability of organic matter can be increased through either dispersion or increased dissolution and hydrolysis by salts. Increasing salinity and sodicity have the potential to increase the amount of DOC available to the microbial population by either i) dissolving organic matter, or ii) converting it either to a more dispersed form (disaggregation) or one that is more easily decomposable, and hence, more readily available. Jandl and Sollins (1997) have suggested that soluble C can provide a large proportion of the microbial substrate, and has the potential to be replenished rapidly by the continued dissolution of organic matter.

When organic matter is solubilised into colloidal form, the increased availability of substrate can counter some of the environmental stresses on the microbial population (Pathak and Rao 1998), such as that caused by increased osmotic potential and ion toxicities. In a separate process, additional substrate for the microbial biomass may also be provided through the process of desorption of SOC from clays. High EC solutions, particularly those high in Na^+, can rapidly alter the composition of exchange sites on clays, causing SOC sorbed on to clay surfaces to be desorbed, which may have occurred in this study. In the high-salinity treatments, it is possible that substrate was readily available and also easily decomposable to offset some of the stresses caused by the increased salt concentration. This process may be indicated by the higher C_{mic}:C_{org} in

the high-salinity treatments compared to the low- and mid-salinity treatments, as the ratio usually increases where substrate availability increases (Anderson and Domsch 1989; Haynes 1999). As a result, the microbial population increased due to increased nutrient supply. After the initial increase in SMB within a week, there was a small decrease in SMB, accompanied by low respiration rates in the high-salinity treatments over the 12-week period. As previously mentioned, the C_{mic}:C_{org} should be used with caution in this study as it was determined at the end of the incubation period only and therefore only provides an indication on the effects of the EC/SAR treatments on C fluxes.

In the mid-salinity treatments, it is possible that the salt concentrations in solution were not high enough to dissolve additional organic matter. However, it is also suggested that the salt concentrations were high enough to increase the osmotic stress, and hence, decrease the microbial respiration (indicated in the cumulative respiration), and the size of the microbial population relative to the high-salinity treatments. Thus, processes that increase the solubility of organic matter could conceivably increase the microbial population in the short term. However, in the longer term, continued dissolution of organic matter and its mineralisation can lead to increased losses of SOC stocks, particularly in areas where biomass inputs are decreased as a result of degraded environmental conditions.

3.5 Summary and Conclusion

The effects on the SMB are more evident with increasing salinity than with increasing sodicity. This has implications for natural resource management and C accounting. Where salinisation and sodification of soils is occurring, it is suggested that soil C stores are becoming depleted as organic matter is increasingly solubilised, providing additional substrate for the microbial population, while plant inputs decrease due to stresses caused by increasing salt content, induced ion toxicities and deficiencies, and declines in soil physical conditions. As this process continues, SOC is likely to be rapidly depleted as mineralisation of SOM continues and inputs of C decrease. However, in this study, saline and sodic effects on the SMB and microbial respiration were artificially created from soil sampled from a vegetated profile. Salinisation and sodication occurs over longer time frames than that measured in this study, and hence, may allow the microbial population to adapt to hostile environmental conditions. Chapter 4 describes the SMB and respiration rates from salt-scalded profiles sampled from the field.

CHAPTER 4: LABORATORY DETERMINATIONS OF SOIL MICROBIAL BIOMASS AND SOIL RESPIRATION FROM SALT-SCALDED SOILS

4.1 Introduction

Extensive research has been undertaken in the past on the physicochemical properties of saline and sodic soils and their amelioration, particularly in regards to soil structure and vegetation health. However, the effects of salinity and sodicity on C dynamics, with respect to C mineralisation or losses from soils, are not as well documented or understood.

As described in Section 2.3, the rate of C accumulation or loss is dependent on the balance between the amount of C input and C loss. C input is dependent on plant inputs and biomass accumulation, as SOC levels are dominated by deposition from litterfall and roots. C inputs in salt-affected soils are also likely to decrease as vegetation health declines due to the direct effects of toxic ions and changes in osmotic potential, as described in Section 2.2.3 and indirect effects in the form of declining soil structure. Sodic soils can also indirectly impact on plant growth due to their adverse effects on soil physical properties which alter plant-water relations.

This chapter addresses key issues in regard to C dynamics in saline-sodic soils under controlled conditions in the laboratory. It assesses how soil respiration and the SMB are affected by salinity and sodicity in existing scalded soils, and assesses the effects following amelioration with gypsum. Soil respiration was determined with the use of soda lime traps, while recognising the issues discussed in Section 3.1. Soils affected by secondary salinisation from Bevendale and Young were sampled and used in the laboratory experiments; these effects were compared with those on prepared saline and saline-sodic soils, as described in Chapter 3.

4.2 Materials and Methods

4.2.1 Site Descriptions

The soil samples used in the laboratory experiments were collected from salt-scalded profiles located on two properties. The first profile was located on a property, "Tarcoola" in Bevendale, approximately 40 km south-west of Crookwell (34 30' 45" S, 149 05' 00" E; Figure 4.1), in the Southern Tablelands region of NSW. The area has been affected by seepage salinity and hence, exhibits scalding of the soil surface. This scalding is estimated to have existed for approximately 60 years (Wagner 2001). The soil profile sampled was a Yellow Sodosol (Isbell 1996), located in an area that was extensively scalded (Plate 4.1). The profile consisted of a loamy sand overlying a sandy loam. The second profile was a Red Kurosol (Isbell 1996), located on a property, "Avoca," approximately 20 km north-west of Young (34 ^{o} 14' 52.31" S, 148^{o} 24' 37.02" E; Figure 4.1) in the South West Slopes region of NSW. The profile consisted of a loamy sand overlying a heavy clay. The profile was located in an area that showed patches of scalding (Plate 4.2); the scalds at the Avoca site have become apparent within the last 10 years (B. Murphy pers. comm. September 2003).

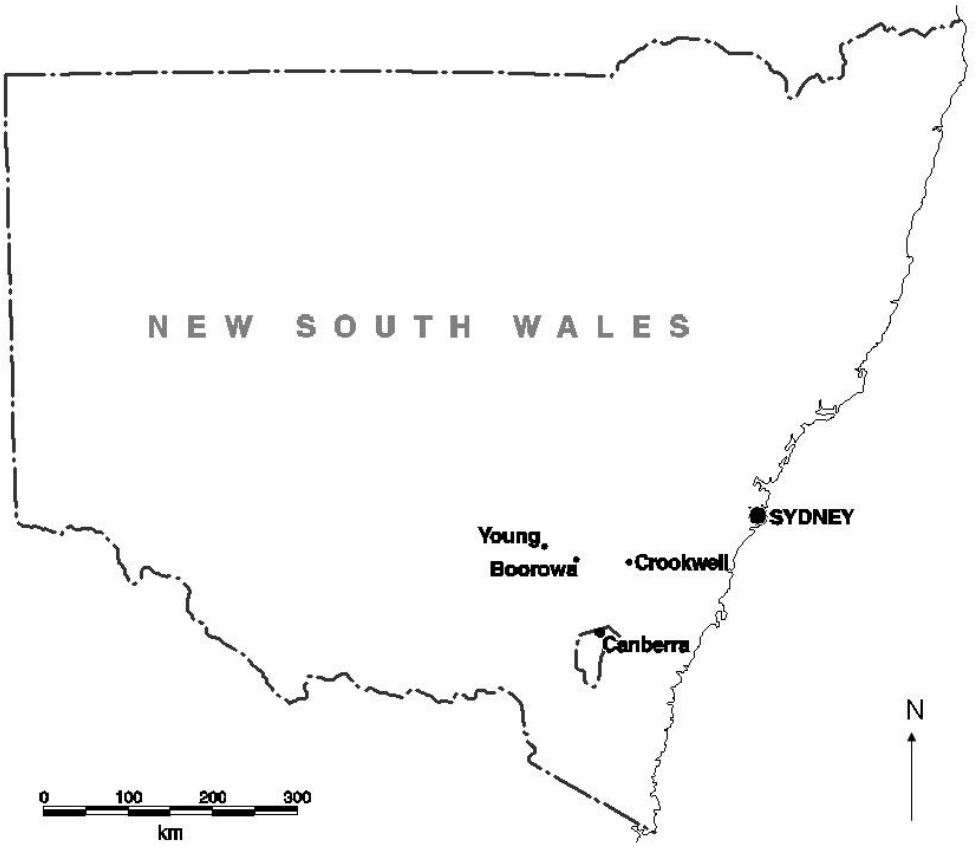


Figure 4.1 Location of the two field sites, "Tarcoola" approximately 40 km south west of Crookwell, and "Avoca," 20 km north-west of Young.

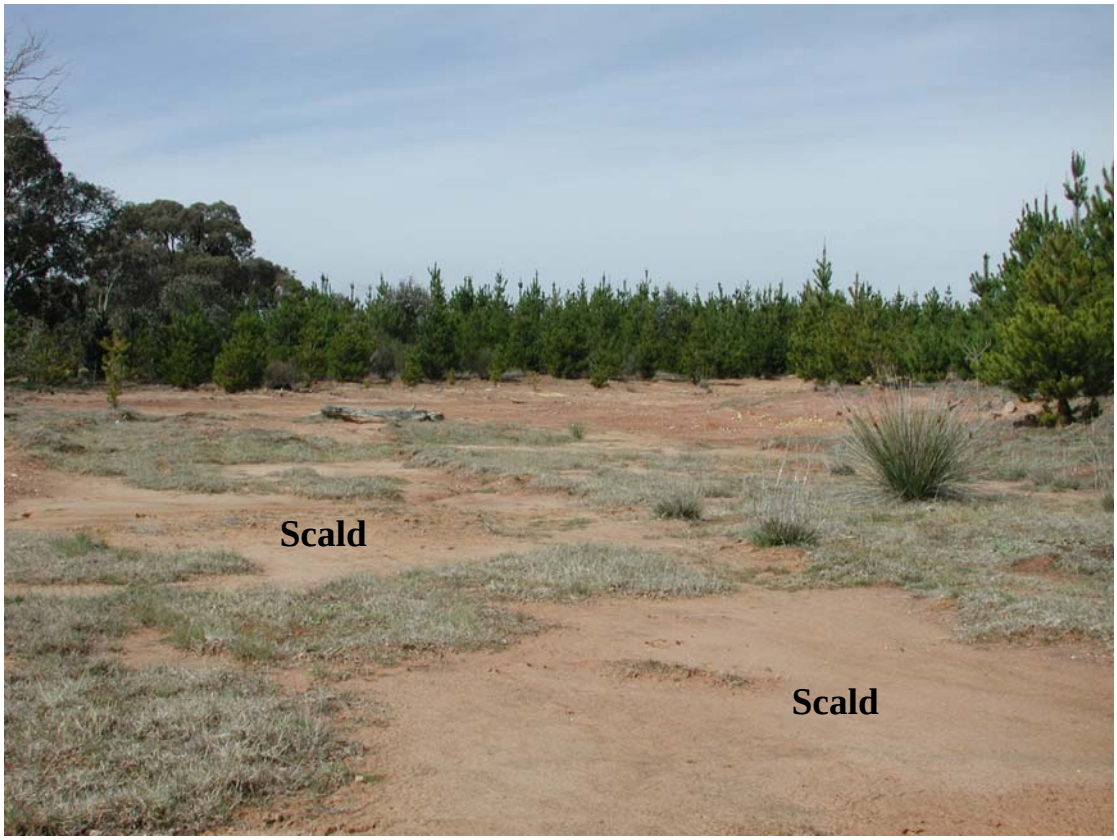


Plate 4.1 Extensive scalding at the "Tarcoola" site

Plate 4.2 Scalding at the "Avoca" site

4.2.2 Field Sampling

Samples were taken from 0-5, 5-10, 10-20, 20-30, 30-50 cm depths of each of the soil profiles, transported back to the laboratory in polyethylene bags, and stored at 4^{o}C prior to analysis. Soils were sampled with a shovel from a soil pit at each depth interval. Bulk density cores were also taken from each depth as described in Section A1.1 in Appendix A.

4.2.3 Sample Preparation and Soil Chemical Analyses

Bulk density cores were oven dried at 105^{o}C for 24 hours, and from the known soil core volume and oven dry weight contained in the soil core, bulk density was calculated, which is described in more detail in Section A1.1 in Appendix A. EC, pH and soluble cations were determined in 1:5 soil:water extracts. Soluble cations in the 1:5 soil:water extracts were analysed by ICP-AES. Exchangeable cations were extracted by using 1 M ammonium acetate extracts buffered to a pH of 7 and also determined by ICP-AES. A more detailed description of the analysis is found in Appendix A. The sodicity of the

samples were determined by calculating the SAR and ESP, described in Equations 2.1 and 2.2, respectively.

Organic C, total N and total S were determined by high temperature combustion on a CNS LECO-2000 analyser. Where the pH ≥ 7, samples were pre-treated with sulfurous acid prior to C analysis. Particle size analysis was undertaken using the hydrometer method (Bouyoucos 1936).

Samples used for soil biological analysis were stored at 4^{o}C prior to analysis. Soils from all depths were used for the measurement of soil respiration and SMB and were initially sieved at their field moisture contents through a 5 mm sieve. Sub-samples were then placed into 9.6 L buckets with holes drilled through the bottoms and covered with filter paper. The "unamended" soils were supersaturated to field capacity with water and allowed to equilibrate for 72 hours (termed *Tarcoola* and *Avoca*, respectively). The "amended" soils (termed *Amended Tarcoola* and *Amended Avoca*, respectively) were prepared with the incorporation of nursery grade gypsum ($CaSO_4.2H_2O$) at a rate of 10 t/ha in powder form and subjected to the same wetting conditions as for the unamended soils. The soils were then maintained in a constant temperature environment at 25^{o}C for the duration of the incubation, and analysed for respiration and SMB, as described below.

4.3.4 Soil Biological Analysis

Soil respiration was determined according to a modification of the method originally developed by Edwards (1982), as described in Section 3.3.4, using soda lime traps.

Soil microbial biomass was measured by the chloroform fumigation-extraction procedure described in Vance et al. (1987) and set out in Section 3.3.5.

All biological analyses were undertaken in quadruplicate.

4.2.5 Statistical Analysis

Data were statistically analysed using the GENSTAT Version 8.0 statistical analysis program (Payne 2005). Where respiration and SMB displayed negative values (negligible respiration and SMB), 0.01 was inserted; the data were then square-root transformed in order to satisfy the assumptions of ANOVA with back-transformed means presented. The block structure was given by week within replicate, within depth, within site, and the treatment structure was given by depth, gypsum, site, week and their interaction. Data were analysed by ANOVA and subjected to LSD testing at the 5 % level where significant differences were found.

4.2.6 Microbial Indices

The qCO_2 was calculated from the soil respiration rate and SMB according to Equation 3.5 in Section 3.2.4.3 at the end of the 12-week incubation period to provide an indication of the effects of gypsum on microbial activity. The C_{mic}:C_{org} was also calculated, as described in Section 3.2.4.3 at the end of the 12-week incubation period.

4.3 Results

4.3.1 Soil Properties

Soil bulk density, pH, EC, SAR, ESP, SOC, N and S were measured in the soils from the two sites, Tarcoola and Avoca. Soil bulk density at Tarcoola showed a general increase with depth to 30 cm, after which it decreased again; at Avoca, it generally remained constant (Table 4.1).

Table 4.1 Soil bulk density with depth

Depth (cm)	Tarcoola (Mg/m^3)	Avoca (Mg/m^3)
0-5	1.46	1.65
5-10	1.48	1.69
10-20	1.56	1.73
20-30	1.60	1.72
30-50	1.47	1.63

The particle size distribution and soil texture of the bulk soils from Tarcoola and Avoca is shown in Table 4.2. At Tarcoola, the soil texture changed gradually from a loamy sand at the surface to a sandy loam from 10 cm to the bottom of the profile. The decrease in bulk density from 30 cm at Tarcoola coincided with an increase in the sand

content and a decrease in the silt content relative to the 20-30 cm depth. At Avoca, the soil texture was a loamy sand at the surface, with an abrupt change at 30 cm to a heavy clay.

Table 4.2 Particle size distribution

Depth (cm)	*Tarcoola*				*Avoca*			
	Sand (%)	Silt (%)	Clay (%)	Soil Texture	Sand (%)	Silt (%)	Clay (%)	Soil Texture
0-5	82.9	10.6	16.7	Loamy sand	83.5	2.5	14.4	Loamy sand
5-10	82.3	9.7	17.1	Sandy loam	85.6	2.6	14.7	Loamy sand
10-20	61.2	19.8	33.2	Sandy loam	88.7	6.5	14.5	Loamy sand
20-30	66.4	19.4	28.8	Sandy loam	89.5	6.5	15.1	Loamy sand
30-50	70.0	5.9	29.2	Sandy loam	32.7	6.6	62.4	Heavy clay

Soil properties of the bulk samples from the two sites are shown in Table 4.3. The Tarcoola samples were highly alkaline (ie. pH ≥ 9.6), non-saline (EC ≤ 0.84 dS/m) and highly sodic (ie. ESP ≥ 12) at all depths. EC decreased with depth from 10 cm, and SOC, total N and total S were also very low throughout the profile, with SOC < 1% at all depths. The Avoca soils were highly acidic (pH ≤ 4.8), non-saline (EC ≤ 1.5dS/m) and highly sodic (ie. ESP ≥ 12) throughout the soil profile. EC decreased with depth to 30 cm, and then increased in the 30-50 cm layer while SOC, total N and total S were low throughout the profile. Whilst the soils were non-saline at the time of sampling, the sites were selected on the basis of a lack of vegetation, which indicated that salinity had occurred in the past. This is supported by a study of historical aerial photographs in the area by Wagner (2001), and discussed in more detail in Section 6.4.3.

Table 4.3 Soil properties of the bulk soil from Avoca and Tarcoola

Depth (cm)	pH $_{1:5(H2O)}$	$EC_{1:5}$ (dS/m)	ESP	SAR	SOC (%)	Total N (%)	Total S (%)
				Tarcoola			
0-5	10.22	0.70	86.4	3.6	0.39	0.018	0.205
5-10	10.31	0.84	67.2	4.6	0.47	0.020	0.246
10-20	10.12	0.74	52.0	3.7	0.18	<0.01	0.250
20-30	9.56	0.27	51.7	1.6	0.28	0.011	0.281
30-50	9.63	0.19	35.1	1.0	0.20	0.013	0.272
				Avoca			
0-5	4.81	1.50	11.6	1.2	1.06	0.075	0.010
5-10	4.58	0.60	72.7	1.0	0.23	0.014	0.003
10-20	4.32	0.86	62.1	1.1	0.11	<0.01	0.003
20-30	4.27	0.62	41.1	0.8	0.10	<0.01	0.002
30-50	4.36	1.20	37.7	2.7	0.29	0.016	0.023

Soil properties, pH, EC, SAR and ESP, of the amended and unamended soils from Tarcoola and Avoca are shown in Figures 4.2, 4.3, 4.4 and 4.5, respectively. The pH of the Tarcoola soil was highly alkaline, with a pH of 10 decreasing to a pH of 9.6 at depth. The addition of gypsum decreased the pH to approximately 8 at all depths. The pH of the Avoca soil was acidic, and showed a general decrease with depth to 30 cm. The addition of gypsum resulted in an increase in pH at all depths.

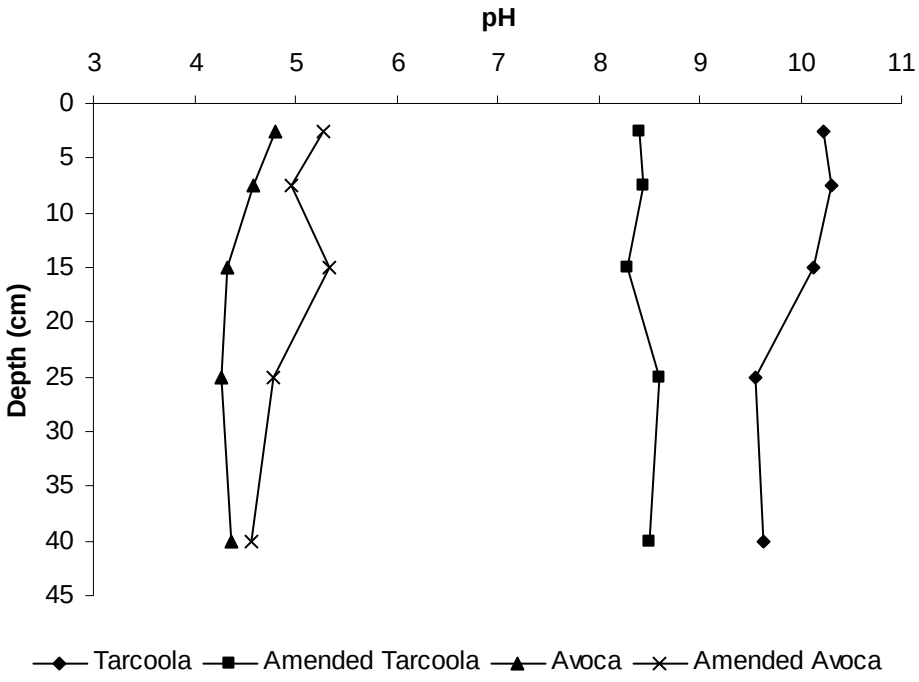


Figure 4.2 $pH_{1:5H2O}$ of the soils amended with gypsum and unamended soils from Tarcoola and Avoca

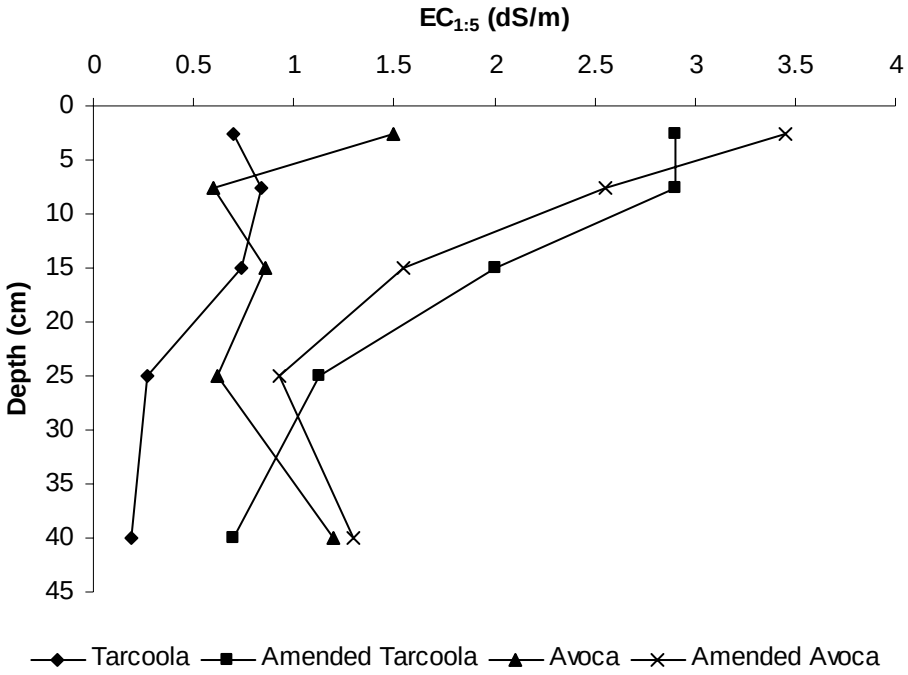


Figure 4.3 $EC_{1:5}$ of the soils amended with gypsum and unamended soils from Tarcoola and Avoca

The EC of the Tarcoola soil was moderately high in the top three layers (EC of 0.70, 0.84 and 0.74 dS/m respectively) before decreasing with depth (Figure 4.2). The EC of the Avoca soil was relatively high in the 0-5 cm layer (EC = 1.5 dS/m) but showed no pattern with depth. Following the incorporation of gypsum, the EC of both soils was increased at all depths.

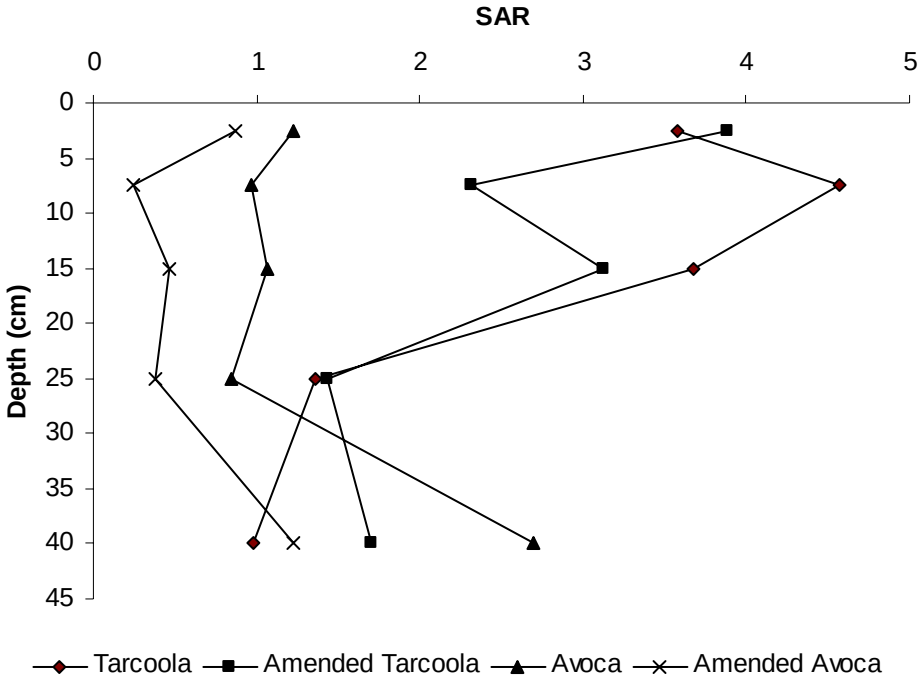


Figure 4.4 SAR of the soils amended with gypsum and unamended soils from Avoca and Tarcoola

The SAR of the soil solution from the Tarcoola profile generally showed an overall decline with depth (Figure 4.4). However, the Amended Tarcoola soil showed no clear pattern with depth. The SAR of the unamended Avoca soil was lower than the unamended Tarcoola soil at all depths with the exception of the 30-50 cm layer. With the addition of gypsum, the SAR of the Amended Avoca soil decreased at all depths relative to the unamended Avoca soil. In both the Avoca and Amended Avoca soils, the highest SAR was found in the 30-50 cm layer.

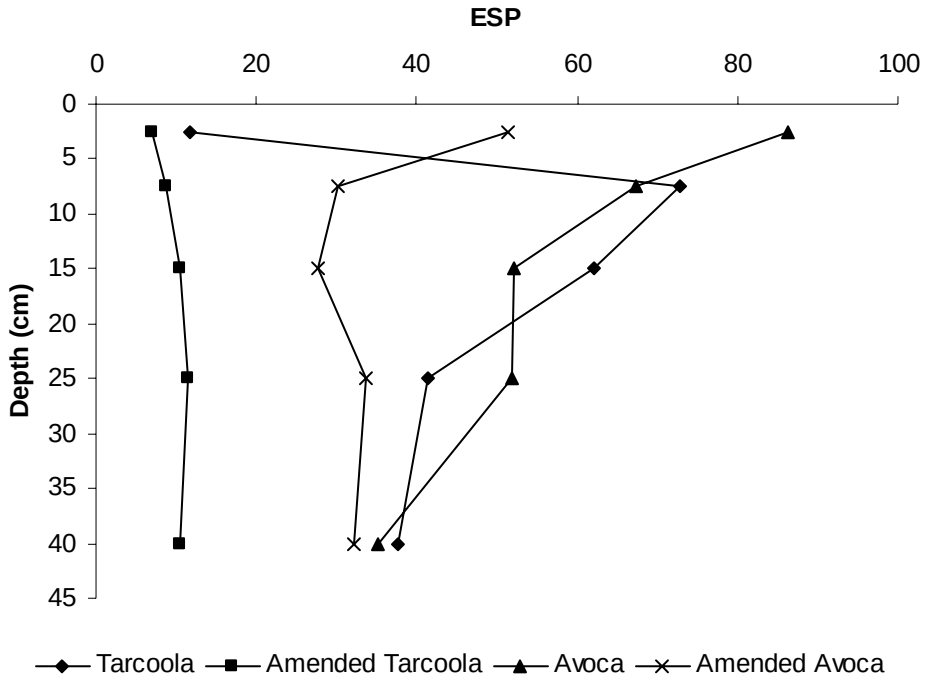


Figure 4.5 ESP of the unamended soils and soils amended with gypsum

The ESP showed the greatest increase between 0-5 cm and 5-10 cm depth before decreasing with depth in the Tarcoola soil (Figure 4.4). Following the incorporation of gypsum, the ESP decreased at all depths. Similarly, the ESP of the Amended Avoca soils decreased at all depths following incorporation of gypsum compared to the unamended soils.

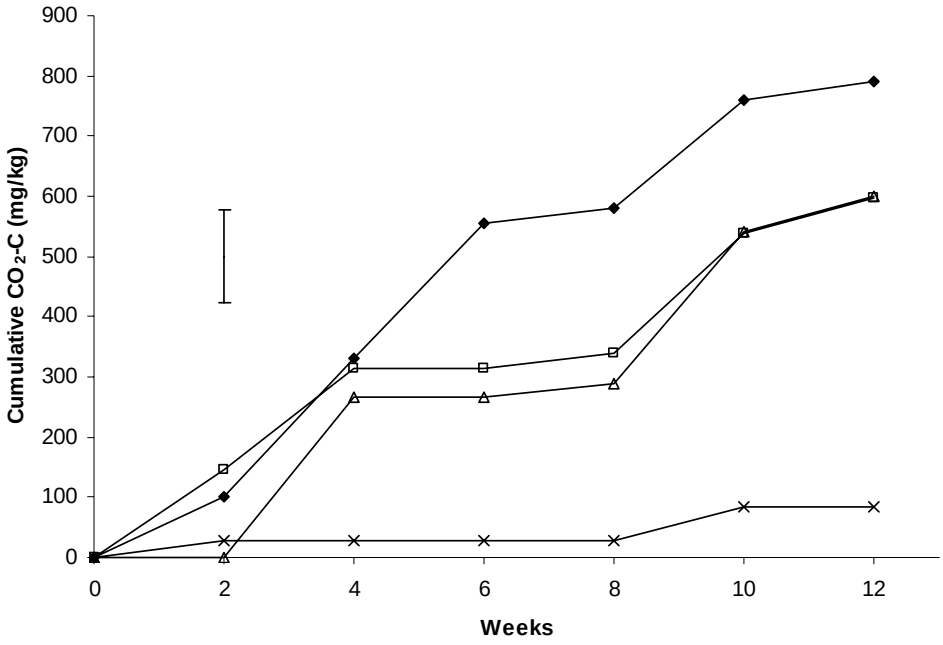
4.3.2 Soil Respiration

Figures 4.6a and 4.6b shows the cumulative soil respiration measured at two-week intervals for the six treatments at the 0-5 and 5-10 cm depths, respectively. Because respiration was minimal at depths below 10 cm (soda lime sample ≈ blank soda lime), it was below the detection limit of the soda lime method, and hence, data for the lower depths are not shown.

There was a significant interaction in respiration rates between site, depth, week and gypsum addition, as shown in Figure 4.6 (P<0.001). Respiration was higher in the 0-5 cm layer in the Avoca soils compared to the Tarcoola soils in the 0-5 cm layer, while the Amended Tarcoola soil had the lowest cumulative respiration rate. However, in the 5-10 cm layer the Tarcoola soils displayed the highest rates of respiration. The respiration rates in the Amended Tarcoola and

Amended Avoca soils were generally slightly lower than the respiration rates in the unamended counterparts over the 12-week experimental period.

a)



b)

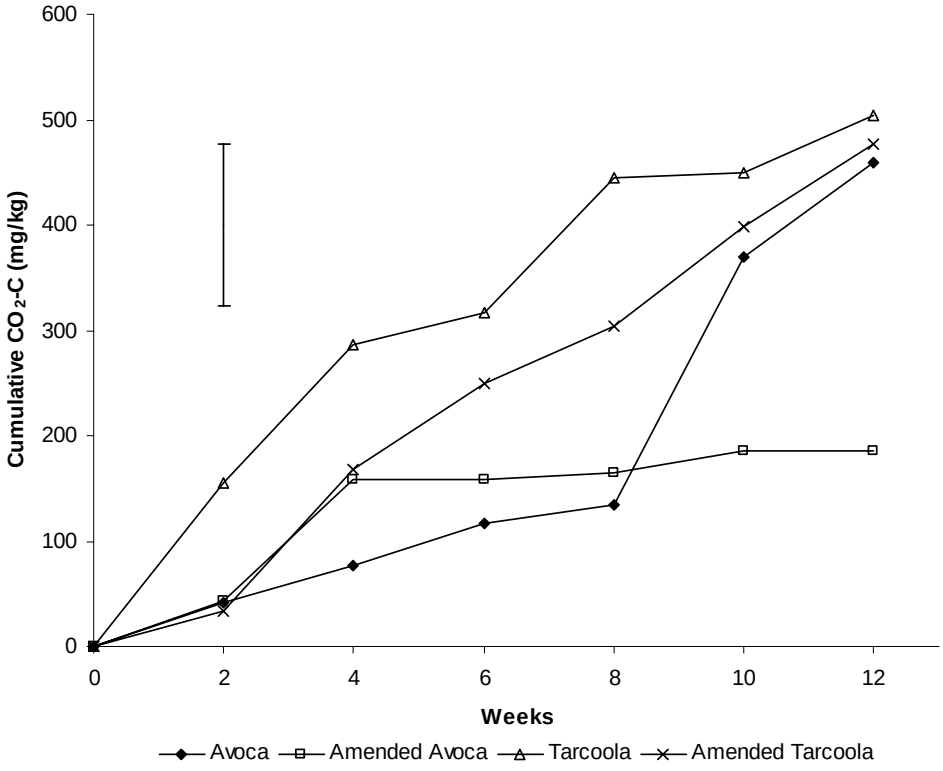


Figure 4.6 Gypsum effects on cumulative respiration rates over the 12 week incubation period from Avoca and Tarcoola at a) 0-5 cm and b) 5-10 cm

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Vertical bar represents the SED.

4.3.3. Soil Microbial Biomass

Data pooled for both soils over the 12 weeks showed the SMB decreased significantly with depth down to 30 cm but increased again at the 30-50 cm depth(P<0.001). The SMB did not differ significantly with the addition of gypsum nor with site (P>0.05; Table 4.4). There was a significant interaction between site and depth (P<0.01), as shown in Figure 4.7. The SMB decreased with depth at both sites to 30 cm, and increased in the 30-50 cm layer. Interactions were found to be highly significant between site, depth, week and gypsum addition (P<0.001), as shown in Figure 4.8.

Table 4.4 Effects of depth, gypsum addition and site on the SMB.

Depth (cm)	**0-5**	**5-10**	**10-20**	**20-30**	**30-50**	***P***
SMB mg/kg	50.55[a]	23.52[b]	12.53[c]	7.08[d]	18.40[bc]	P<0.001
Gypsum (t/ha)	**0**	**10**				
SMB mg/kg	20.25[a]	20.07[a]				NS
Site	**Avoca**	**Tarcoola**				
SMB mg/kg	19.18[a]	21.16[a]				NS

Note: Different letters within a row indicate a significant difference. NS indicates result is not significantly different. Data were square-root transformed for statistical analysis with back-transformed means presented.

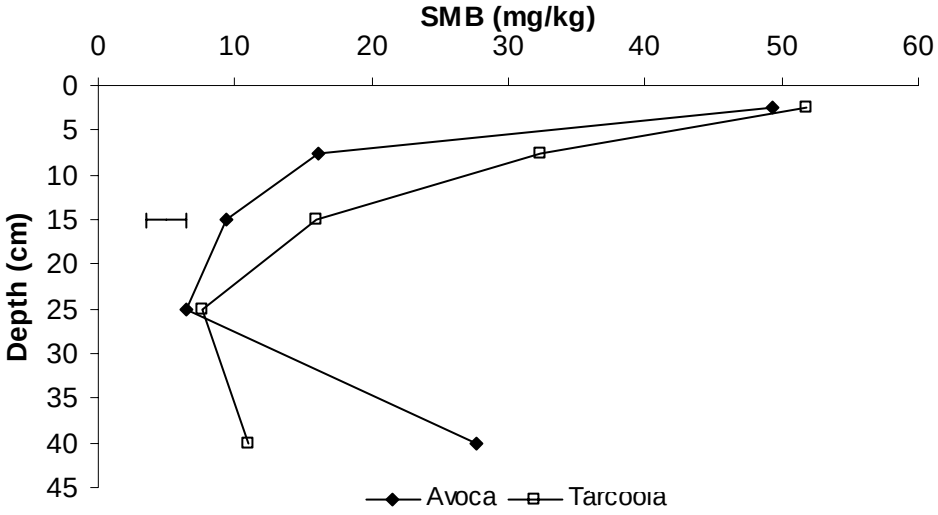


Figure 4.7 Pooled SMB data from Avoca and Tarcoola.

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Horizontal bar represents the SED.

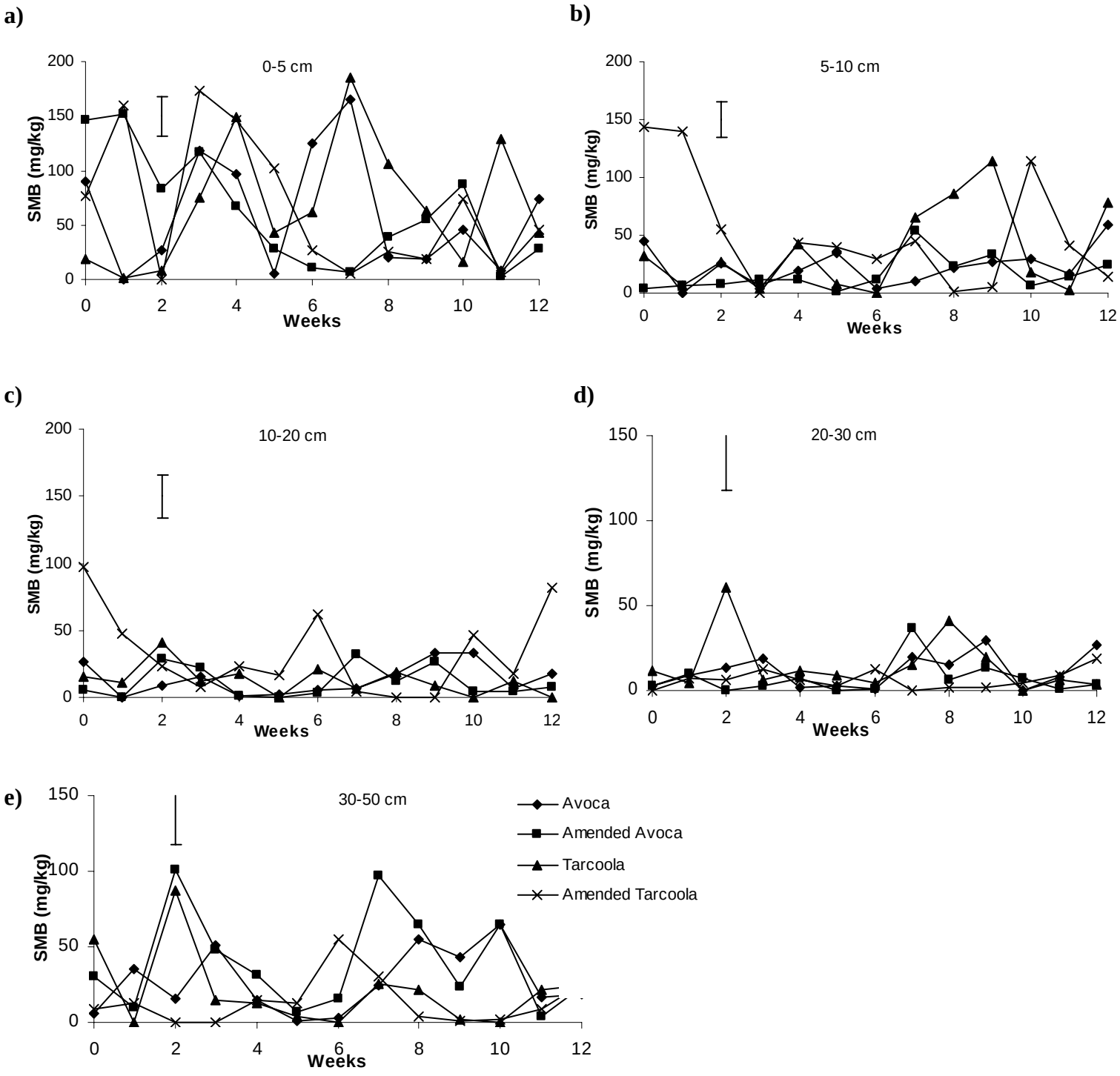


Figure 4.8 Gypsum effects on SMB over the 12 week incubation period from Avoca and Tarcoola at a) 0-5 cm, b) 5-10 cm, c) 10-20 cm, d) 20-30 cm and e) 30-50 cm.

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Vertical bar represents the SED.

No clear differences could be discerned in the SMB between sites in the amended and unamended soils.

4.3.4 Microbial Indices

Table 4.5 shows the effects of gypsum incorporation (0 and 10 t/ha) on qCO_2 and C_{mic}:C_{org} in the 0-5 and 5-10 cm layers at the end of the 12-week incubation period. The qCO_2 was higher in the Avoca soils compared to the Tarcoola soils in the 0-5 cm layer, while no consistent effect of gypum or site was seen in the 5-10 cm layer. The addition of gypsum did not have a consistent effect on the qCO_2. The C_{mic}:C_{org} was lower in the Avoca soils compared to the Tarcoola soils in the 0-5 cm layer, while there were no apparent differences in the 5-10 cm layer. The addition of gypsum at both sites appeared to decrease the C_{mic}:C_{org} compared to the unamended counterpart, primarily due to lower respiration rates in the former treatment.

Table 4.5 Effects on qCO_2 and C_{mic}:C_{org} due to gypsum incorporation and site.

Depth (cm)	Site	Gypsum (t/ha)	qCO_2 (mg CO_2-C/d/mg SMB-C)	C_{mic}:C_{org}
0-5	Avoca	0	0.109	0.74
	Avoca	10	0.185	0.34
	Tarcoola	0	0.078	2.26
	Tarcoola	10	0.060	1.50
5-10	Avoca	0	0.068	2.83
	Avoca	10	0.051	1.57
	Tarcoola	0	0.046	2.38
	Tarcoola	10	0.063	1.44

4.4 Discussion

4.4.1 Effects of gypsum addition

The decrease in pH at all depths in the *Amended Tarcoola* soils was due to the incorporation and dissolution of gypsum. The decrease in pH from 10.3 in the Tarcoola soils to 8.3 in the *Amended Tarcoola* samples with the addition of gypsum is due to reactions of CO_3^{2-} and HCO_3^- in the original soil solution with Ca^{2+} from the gypsum (Equations 4.1a and 4.1b). Due to the high pH of the soil as a result of Na_2CO_3, which dissociates to Na^+ and CO_3^{2-} ions (Equations 4. 2a and 4.2b), the addition of gypsum provides a source of Ca^{2+} ions which precipitates as $CaCO_3$ and $Ca(HCO_3)_2$, resulting in a decrease in soil pH. This may have also caused the slight increase in SAR in the 0-5 cm layer of the *Amended Tarcoola* soil, due to lower concentrations of Ca^{2+} ions in solution. Alternatively, the increase in the SAR may have been caused by increases in Na^+ ions in solution, due to exchange with Ca^{2+}, which is most likely. This is confirmed

in the decrease in ESP at all depths, including the 0-5 cm layer, indicating that the exchangeable Ca^{2+} concentration had increased, and the exchangeable Na^{+} concentration had decreased. In addition to decreasing soil pH due to its role in the precipitation of $CaCO_3$ and $Ca(HCO_3)_2$, the addition of gypsum also leads to proton generation and further reductions in pH (Chorom and Rengasamy 1997).

$Ca^{2+} + CO_3^{2-} \leftrightarrow CaCO_3$ **Equation 4.1a**

$Ca^{2+} + 2HCO_3^{-} \leftrightarrow Ca(HCO_3)_2$ **Equation 4.1b**

$Na_2CO_3 \leftrightarrow 2Na^{+} + CO_3^{2-}$ **Equation 4.2a**

$CO_3^{2-} + H_2O \leftrightarrow OH^{-} + HCO_3^{-}$ **Equation 4.2b**

In the *Amended Avoca* soils, however, the pH increased at all depths with the addition of gypsum. It is possible that the increase in pH was the result of SO_4^{2-} exchange from gypsum with hydroxyl groups on the clay particles, increasing the concentration of OH^{-} in solution which results in an increase in pH. However, it is more likely that the increase in pH occurred due to the pH of a saturated gypsum solution from the nursery grade gypsum used, which was 6.6. Because the pH of the gypsum solution was higher than that of the *Avoca* soil solutions, the pH of the *Amended Avoca* soil solutions increased. This effect is further compounded by the lower buffering capacity of the soil due to the low clay content and high sand content throughout the profile to a depth of 30 cm. The decrease in the ESP in the *Amended Avoca* soils was the result of Na-Ca exchange processes similar to those which occurred in the *Amended Tarcoola* soils.

In both soils, because the amended soils were not subjected to leaching, the EC of the soil solution was increased compared to the unamended soils. Similarly, because the soils were not subjected to leaching, the SAR of the soil solutions showed a smaller decrease than expected in the field. While exchange reactions took place, indicated in the decrease in ESP, Na^{+} was not leached from the soil.

4.4.2 Soil respiration and microbial biomass in salt-scalded soils

In general, gypsum-amended soils exhibited lower respiration rates than the unamended soils, thus supporting the salinity effects on respiration, as shown in Table 3.4 and

Figure 3.7. The higher respiration rate in the Avoca soil from the 0-5 cm depth compared to the Tarcoola soil is related to the higher SOC concentration found in the Avoca soil (1.06% at Avoca compared to 0.39% at Tarcoola). Previous studies have found a correlation between organic matter content and the size of the SMB, with higher levels of organic matter resulting in a larger SMB (Schnurer et al. 1985). Low SOC levels measured at both sites are attributed to the absence of vegetation growth in the scalded areas. Therefore, any C inputs into the soil are external, and most likely related to depositional processes. In this study, SOC levels were less than 1 % at both sites, and at all depths with the exception of the 0-5 cm layer at Avoca. Slightly higher SOC levels were found at Avoca most likely due to the difference in the length of time each site had been subjected to scalding, with the Avoca site having been scalded for approximately 10 years compared to the Tarcoola site which has been scalded for approximately 60 years. In addition, the sizes of the scalds at Avoca were smaller than those found at Tarcoola, and probably had SOC contributions from the lateral distribution of roots from nearby areas. Because scalded areas are more susceptible to erosion, losses of SOC are increased as SOC is usually concentrated in the surface layer and is relatively unconsolidated (Lal 2001). As the topsoil is eroded, SMB is also lost as it is also concentrated in the upper layers of the soil profile (Murphy et al. 1998), while the soil's fertility and microbial resilience are decreased (Mabuhay et al. 2006). Low levels of SMB are further compounded in salt-scalded areas because the SMB is intimately associated with the rhizosphere, with microbial population densities up to an order of magnitude higher than in the bulk soil (Toal et al. 2000). Therefore, areas with little or no vegetation will have very little C deposited in the form of root exudates and root turnover, and hence will exhibit low levels of SMB. In this study, low levels of SMB were found in the profiles of both sites, and as a result, any treatment effects are negligible despite optimal soil moisture and temperature conditions.

Under acidic conditions, the SMB can be further stressed by salinity (Rasul et al. 2006), which may be the case in the Avoca soils. Similarly, pH stress may also be occurring in the Tarcoola soils under alkaline conditions. However, because the Tarcoola site has been estimated to have been scalded for 60 years, the lack of C input due to absence of vegetation is the most likely cause of the low levels of SOC, and hence SMB and the lower respiration rates, which is likely to be at a minimum level, particularly in the 0-5 cm layer. In stabilised conditions, the size of the microbial biomass will reach

equilibrium with substrate supply in the soil (Liu et al. 2006); therefore, where biomass is low for a significant amount of time, the SMB and soil respiration rate will also follow a similar pattern. Similarly, the SMB and cumulative respiration rates from the Avoca soils will also be lower than that of a vegetated soil, such as that described in Chapter 3, due to scalding which has occurred over a period of 10 years, as turnover of the SMB occurs in the order of weeks to months (Jenkinson and Raynor 1977). Because microorganisms are generally more salt-tolerant than plants, the availability of substrate in the form of litterfall or root exudates is the limiting factor (Sarig and Steinberger 1994) with the substrate that is available, of a lower quality in terms of biochemistry (Tejada et al. 2006).

4.4.3 Gypsum and soil biological activity

Soil microbial activity and the size of the microbial population are also dependent on soil environmental conditions (Conant et al. 2000). Gypsum has been found to increase the SMB by Carter (1986) and Chorom and Rengasamy (1997), who have attributed this to improvements in the physicochemical environment caused by the addition of gypsum. Improvement in soil physical properties can contribute to increased biological activity indirectly by improving soil structure which allows water and air to pass through pores. In this study, samples were subjected to optimal soil moisture and temperature conditions. While the addition of gypsum resulted in a pH change towards neutral in both soils, it may be argued that the high salinity levels as a result of gypsum addition can deleteriously affect microbial activity and the size of the microbial population. However, both amended and unamended soils at both sites exhibited low levels of SMB and lower levels of cumulative respiration compared to that found in Chapter 3 from a soil that is vegetated, which suggests that the microbial biomass in scalded soils is the result of low SOC stocks and consequently low C substrates in both acidic and alkaline conditions. Because soil respiration is a function of the decomposition of SOM, root respiration, and root associated respiration, it is not surprising that scalded soils show low rates of respiration as there is likely to be little substrate to decompose. The readily decomposable C pool is concentrated in the short-lived fraction of the larger vegetative fraction, with plant life cycles the most important factor defining C supply (Buyanovsky and Wagner 1995). It is likely that the fluctuating levels of the SMB and low levels of soil respiration, compared to that found in Chapter 3, are most likely a result of the

microbial population turning over on itself, and decomposing the dead population and its metabolites.

4.4.4 Microbial Indices

It is of interest to note that the qCO_2 in the unamended Tarcoola soil (0.078 mg CO_2-C/d/mg SMB-C; Table 4.5) is comparable to that of the control treatment described in Chapter 3 (0.080 mg CO_2-C/d/mg SMB-C; Table 3.7) in the 0-5 cm layer. While the CO_2-C evolved at the end of the 12 weeks from the Tarcoola soil without gypsum amendment was approximately 25 % of the CO_2-C evolved from the control treatment in Chapter 3 (2246 CO_2-C mg/kg and 598 CO_2-C mg/kg, respectively), there was 10 times less SOC found in the scalded soil compared to that found in a vegetated soil (Table 4.3; 0.39 % SOC and Table 3.2; 3.87 % SOC, respectively). It is possible that over long periods of time, such as the time taken for scalding to occur as a result of salinisation and sodication, the SMB can adapt to soil environmental conditions. Adaptation of the microbial biomass has implications for C stocks during the degradation process, resulting in the efficient use of the available substrate while C inputs decrease. Over long periods of time, it is therefore conceivable that SOC becomes depleted.

The C_{mic}:C_{org} was lower than the 1-5 % suggested in Section 2.2.3 in the 0-5 cm layer of the Avoca (Table 4.5), most likely due to the scalding of the soil surface. It is possible the microbial biomass is still in the process of adapting to the current degradation processes, as the qCO_2 was higher and C_{mic}:C_{org} was lower compared to the Tarcoola site. The qCO_2 was also higher in the Avoca soil which had been amended gypsum compared to the Avoca soil which had not been amended. This may have been due to an increase in stress following the addition of gypsum due to the increase in EC. Similarly, in a study by Usman *et al.* (2004), an increase in EC following the addition of sewage sludge resulted in an increase in the qCO_2 to 0.130 mg CO_2-C/d/mg SMB-C, which was attributed to an increase in stress. In the current study, the addition of gypsum in both soils decreased the C_{mic}:C_{org} at both depths. This was due to the low levels of SMB, and may be related to the osmotic effect from salt addition. Because the addition of gypsum increases the electrolyte concentration of the soil solution, it is suggested that gysum addition resulted in flocculation of soil particles and the formation of aggregates due to

processes described in Section 2.2.4. These aggregates physically protect substrate, and therefore, the availability of substrate for the microbial biomass decreases, resulting in a decrease in the C_{mic}:C_{org}. However, given the large variability in the SMB-C over the 12-week incubation period (Figure 4.8), it is likely that the qCO_2 and C_{mic}:C_{org} determined at the end of the incubation period only merely provides an indication of trends rather than an accurate measure of biological activity.

4.5 Summary and Conclusion

In comparison to the results found in Chapter 3, C stocks and fluxes in salt-scalded soils in this series of experiments were found to be low. The lower levels of SMB and respiration rates are attributed to the low SOC levels found at both Avoca and Tarcoola, which are most likely the result of low plant biomass C inputs. It is suggested that as C inputs slow, or cease, native SOM is decomposed until the biochemically recalcitrant or physically protected C remains. While it was beyond the scope of this project, further research in fractionation of SOM under such conditions would confirm this hypothesis. It is likely that a lack of substrate restricts the SMB and respiration rates. Chapter 5 will determine whether decomposition processes can be restored in salt-scalded soils if substrate is available under controlled conditions. Therefore, the effects of salinity and sodicity on decomposition in a salt-scalded soil can be ascertained.

CHAPTER 5: DECOMPOSITION OF ADDED ORGANIC MATERIAL IN SALT-AFFECTED SOILS

5.1 Introduction

The addition of organic materials to soil has frequently been used in the past to aid in the rehabilitation of degraded lands. The importance of maintaining high levels of SOM, and hence, high levels of SOC, has been well established. SOM can improve soil structure and aggregation (Oades 1988; Tisdall and Oades 1982), increase hydraulic conductivity (Baldock *et al.* 1994), and promote higher nutrient levels and greater cation exchange capacity (von Lutzow *et al.* 2002). Incorporation of organic material, notably in the form of crop residues has been shown to improve soil aggregation and increase SOC stocks (Lal *et al.* 1999), while retaining stubble increases SOM and soil faunal activity (Valzano *et al.* 2001a).

Gypsum ($CaSO_4.2H_2O$) is the most commonly used ameliorant to reduce soil sodicity or to treat saline-sodic soils (Chapter 4). However, limited studies have been undertaken on the effects of gypsum on microbial processes in soils. One such study found that in the short term, the addition of gypsum caused a decrease in microbial activity, but tended to increase SMB; this was attributed to changes in the soil chemical environment (Carter 1986). However, the findings are far from conclusive.

Chapter 4 showed that very low levels of SMB and soil respiration rates occurred under controlled conditions in soils sampled from scalded areas in south-eastern Australia, both with and without gypsum ameliorant compared to SMB and soil respiration rates found in a vegetated soil as discussed in Chapter 3. In Chapter 4, these low rates were attributed to low levels of SOC, which provide little substrate for decomposition and hence, low levels of microbial activity. This current chapter aims to determine the behaviour of the labile C pool in scalded soils when treated with gypsum and organic material amendment under controlled temperature and moisture conditions. By providing additional organic material which the microbial biomass can decompose, C fluxes in degraded soils where substrate is available can be elucidated.

5.2 Materials and Methods

5.2.1 Site Descriptions

The soil used for analysis was collected from the same two salt-scalded sites as described in Chapter 4. The first profile was a Yellow Sodosol (Isbell 1996) from a property, "Tarcoola" in Bevendale, while the second profile was a Red Kurosol (Isbell 1996), located on a property "Avoca." Descriptions of these two sites are given in Section 4.2.1.

5.2.2 Field Sampling

Samples were taken from 0-5, 5-10, 10-20, 20-30 and 30-50 cm depths of the soil profile and subjected to the same treatment as described in Section 3.2.2.

5.2.3 Sample Preparation and Soil Chemical Analyses

Samples used for soil biological analysis were stored at 4^{o}C prior to analysis. Organic material was added in the form of kangaroo grass (*Themeda australis*). Following collection, the plant material was air dried for 72 hours, and coarsely ground with the use of a coffee grinder until the plant material was approximately 10-20 mm in length. Due to the amount of plant material required, grinding was undertaken in batches, with all the batches of plant material bulked prior to weighing out the required amount for incorporation into the soils. The kangaroo grass had a total C, N, C:N ratio and S content of 40.4 %, 0.531 %, 76 and 0.062 %, respectively.

Soils that were used for the measurement of microbial biomass and respiration were initially sieved at their field moisture contents through a 5 mm sieve. It should be noted that the organic material was incorporated into the soils after being passed through the 5 mm sieve. Subsamples were then placed into 9.6L buckets with holes drilled through the bottoms and covered with filter paper. Plant material was then incorporated into the soils of each separate depth interval in the laboratory at a rate of 10 t/ha (termed *Tarcoola+OM* and *Avoca+OM* soils, respectively) according to Equations 5.1 and 5.2.

$S = 10\,000 * 10\,000 * d * BD$ **Equation 5.1**

Where S = mass of soil (g) in one ha

d = depth interval (cm)

BD = oven-dried bulk density (g/cm^3)

10 000 * 10 000 is the area of one ha in cm^2

$OM = i/S * 10$ **Equation 5.2**

Where OM = weight of organic material required for incorporation (g)

i = weight of soil sampled from a depth interval

S = mass of soil in one ha from a particular depth interval from Equation 5.1

10 refers to incorporation rate of 10 t/ha

The soils amended with both gypsum ($CaSO_4.2H_2O$) and organic material (termed *Amended Tarcoola+OM* and *Amended Avoca+OM* soils, respectively) were prepared by applying nursery grade gypsum at a rate of 10 t/ha in addition to the organic material.

Water was then added to field capacity to all the samples, with the samples allowed to equilibrate for 72 hours. The soils were then maintained in a constant temperature environment at 25^oC for the duration of the incubation, and analysed for respiration and SMB, as described below.

Following the equilibration period of 72 hours after the incorporation of organic material and gypsum, approximately 10 g of soil was sub-sampled for soil chemical analysis. The same chemical analyses were undertaken as described in Section 3.2.3. Bulk density cores were oven dried at 105^oC for 24 hours. pH and EC were analysed in 1:5 soil:water extracts. Soluble cations were analysed by ICP-AES in the 1:5 soil:water extracts. Exchangeable cations were extracted using 1:5 soil:1 M ammonium acetate (CH_3COONH_4) solutions buffered to a pH of 7. Exchangeable cations were determined by ICP-AES.

Organic C, total N and total S were determined by high temperature combustion on a LECO CNS-2000 analyser. Inorganic C was removed with sulphurous acid where the pH $\geq$ 7. Samples analysed for organic C, N and S were air-dried and crushed with a

mortar and pestle to pass through a 2 mm sieve to remove gravel sized particles. It should be noted however, that when the sub-sampled air-dried soil was passed through the 2mm sieve for chemical analysis, some plant material did not pass through the sieve. Where organic material was incorporated into the soil, as described below, inorganic C was removed prior to incorporation. Organic C, total N and total S of the soil samples were determined after incorporation of organic material and prior to the incubation of the samples. Particle size analysis was undertaken using the hydrometer method (Bouyoucos 1936).

5.2.4 Soil Biological Analysis

Soil respiration was determined using soda lime traps according to the method described in Edwards (1982) and Section 3.2.4.1.

Soil microbial biomass was measured by the chloroform fumigation-extraction procedure described by Vance *et al.* (1987) and in Section 3.3.5.

The results were then compared with those found in Chapter 4, which did not have organic material incorporated, as described in Section 5.2.6.

All biological analyses were done in quadruplicate.

5.2.5 Microbial Indices

The qCO_2 was calculated according to Equation 3.5 in Section 3.2.4.3 at the end of the 12-week incubation period to provide an indication of the effects of gypsum on microbial activity. The C_{mic}:C_{org} was also calculated, as described in Section 3.2.4.3 at the end of the 12-week incubation period.

5.2.6 Statistical Analysis

Data were analysed using the GENSTAT 8.0 statistical analysis program (Payne 2005). Differences occurring in the different treatments were subjected to an ANOVA. The block structure was given by week within replicate, within depth, within site, and the treatment structure was given by depth, gypsum, site, week and their interaction.

Differences found in respiration rates over the 12-week incubation period were analysed by REML, as the respiration data were found to be significantly correlated over time (P<0.05), with fixed effects for site, depth, week and their interaction, and random effects for the interaction of site, depth, gypsum addition and week. Where significant differences were found (P<0.05), data were subjected to LSD testing at the 5 % level. The SMB data were square-root transformed to satisfy the assumptions of normal distribution for ANOVA with back-transformed means presented.

Differences occurring in the SMB and soil respiration due to the different treatments described in Section 5.2.3 and those described in Chapter 4 were analysed by REML. Fixed effects were given by site, depth, week and their interaction, and random effects for the interaction of site, depth, gypsum addition, organic material addition and week. Where significant differences were found (P<0.05), data were subjected to LSD testing at the 5 % level.

5.3 Results

5.3.1 Soil Properties

Soil bulk density values were high in both the Tarcoola and Avoca soils, but did not show a clear pattern with depth (Table 5.1).

Table 5.1 Soil bulk density at five depths of the soil profile

Depth (cm)	Avoca (Mg/m^3)	Tarcoola (Mg/m^3)
0-5	1.40	1.61
5-10	1.61	1.47
10-20	1.47	1.72
20-30	1.43	1.69
30-50	1.61	1.84

The particle size distribution of the bulked soils is shown in Table 5.2. The soil texture at Avoca was a sandy clay loam at the surface, with a distinct change to a medium clay at 10 cm. The increase in clay content was not reflected in the bulk density values. At Tarcoola, the soil texture was also a sandy clay loam at the surface to 10 cm, a sandy clay between 10-30 cm, and a sandy clay loam at 30-50 cm.

Table 5.2 Particle size distribution of the Avoca and Tarcoola soil profiles

Depth (cm)	*Avoca*				*Tarcoola*			
	Sand (%)	Silt (%)	Clay (%)	Soil Texture	Sand (%)	Silt (%)	Clay (%)	Soil Texture
0-5	73.5	5.1	22.3	Sandy clay loam	55.9	14.3	32.1	Sandy clay loam
5-10	70.1	8.6	25.3	Sandy clay loam	47.1	12.3	39.8	Sandy clay loam
10-20	44.5	6.6	54.9	Medium clay	52.3	16.5	42.3	Sandy clay
20-30	45.9	4.6	49.2	Medium clay	47.4	16.6	48.6	Sandy clay loam
30-50	45.1	4.2	46.7	Medium clay	58.6	12.6	32.6	Sandy clay loam

Soil chemical properties of the untreated soil are shown in Table 5.3. The 0-5 cm layer from Tarcoola was saline ($EC_{1:5} \geq 1.5$ dS/m), while non-saline EC values were found from 5–50 cm depths. At the surface, the profile was highly sodic (ESP > 6) and alkaline (pH > 7) but these properties generally decreased with depth. The SOC concentration was very low (< 0.2 %), and with N, displayed a general decrease with depth, while S did not show any pattern. The Avoca soils were acidic (pH < 7) and saline ($EC_{1:5} \geq 1.5$ dS/m) throughout the profile with the exception of the 5-10 cm layer. The profile was sodic (ESP > 6) at all depths and did not display a clear pattern. The SOC and N concentrations showed a general decrease with depth, while total S did not show a clear pattern.

Table 5.3 Soil properties of the untreated soil from Avoca and Tarcoola

Depth (cm)	$pH_{1:5(H2O)}$	$EC_{1:5}$ (dS/m)	ESP	SAR	SOC (%)	Total N (%)	Total S (%)
				Tarcoola			
0-5	10.42	2.65	89.1	21.9	0.14	0.022	0.376
5-10	9.95	0.60	62.2	2.4	0.13	0.023	0.355
10-20	9.81	0.22	48.3	1.0	0.12	0.017	0.407
20-30	9.43	0.22	51.0	0.8	0.12	0.015	0.384
30-50	9.32	0.11	38.6	0.7	0.08	0.015	0.321
				Avoca			
0-5	4.83	2.00	35.9	2.4	1.19	0.088	0.010
5-10	4.70	1.10	28.3	1.9	0.87	0.055	0.009
10-20	4.45	1.55	30.8	2.6	0.69	0.051	0.017
20-30	4.61	1.75	18.9	2.7	0.50	0.040	0.024
30-50	4.37	1.72	20.9	2.7	0.57	0.063	0.024

The pH of the soil solutions of the untreated and treated soils from both sites are shown in Figure 5.1. The soil sampled from Tarcoola was highly alkaline, with pH values between 9.32 and 10.42. The pH of the soil solution decreased slightly with the addition of organic material alone, but decreased markedly with the addition of organic material and gypsum, with the largest decrease occurring in the 5-10 cm layer. The soil sampled from Avoca was acidic, with little change with depth and showed very little change with the addition of organic material or with the combined addition of gypsum and organic material.

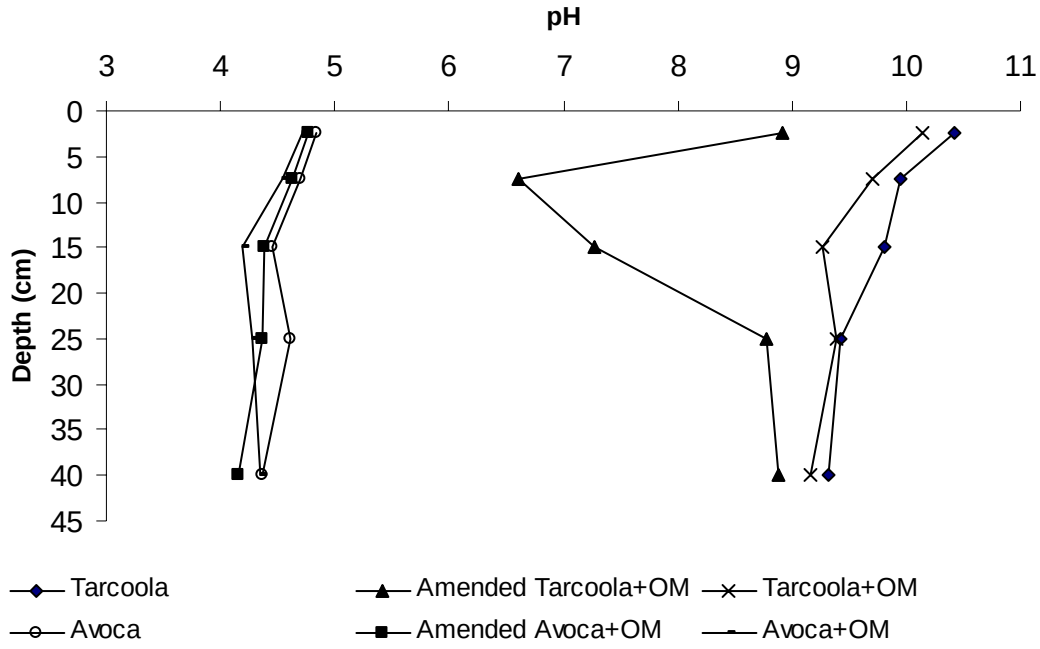


Figure 5.1 $pH_{1:5(H2O)}$ profiles of the untreated Tarcoola and Avoca soils without organic material addition, and the treated Tarcoola and Avoca soils with organic material addition (OM).

NB. $pH_{1:5(H2O)}$ profiles were determined following incorporation of organic material and gypsum at the end of the equilibration period.

The EC profiles of the treated soils are shown in Figure 5.2. Following the addition of organic material, both soils showed a general increase in EC at all depths. The *Amended Tarcoola* and *Amended Avoca* soils had the highest EC values at all depths compared to their respective soils amended with organic material only, and with the untreated soil.

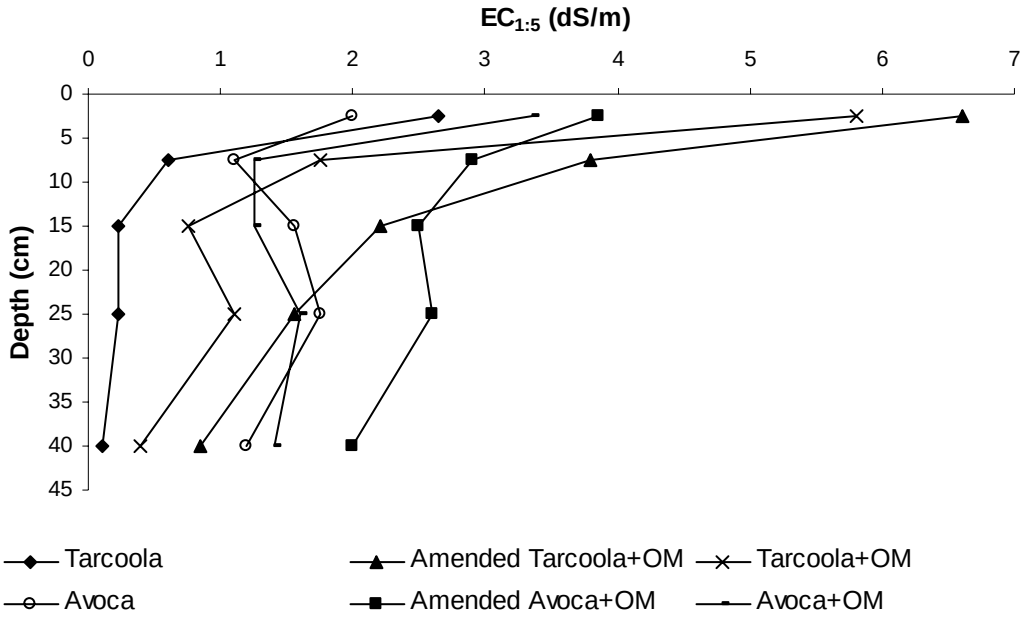


Figure 5.2 $EC_{1:5}$ profiles of the untreated Tarcoola and Avoca soils without organic material addition, and the treated Tarcoola and Avoca soils with organic material addition (OM).

The SAR of the untreated *Tarcoola* soil was highest at the surface, and showed a general decrease with depth (Figure 5.3). The addition of organic material resulted in an increase in SAR at all depths compared to the untreated soil, while the addition of organic material and gypsum caused the SAR to decrease in the 0-5 cm layer only, with no clear pattern shown with depth (Figure 5.3). The SAR of the untreated *Avoca* did not display a clear pattern with depth (Table 5.3). The *Amended Avoca+OM* soil and *Avoca+OM* soil had slightly lower SAR at all depths compared to the untreated soil (Figure 5.3).

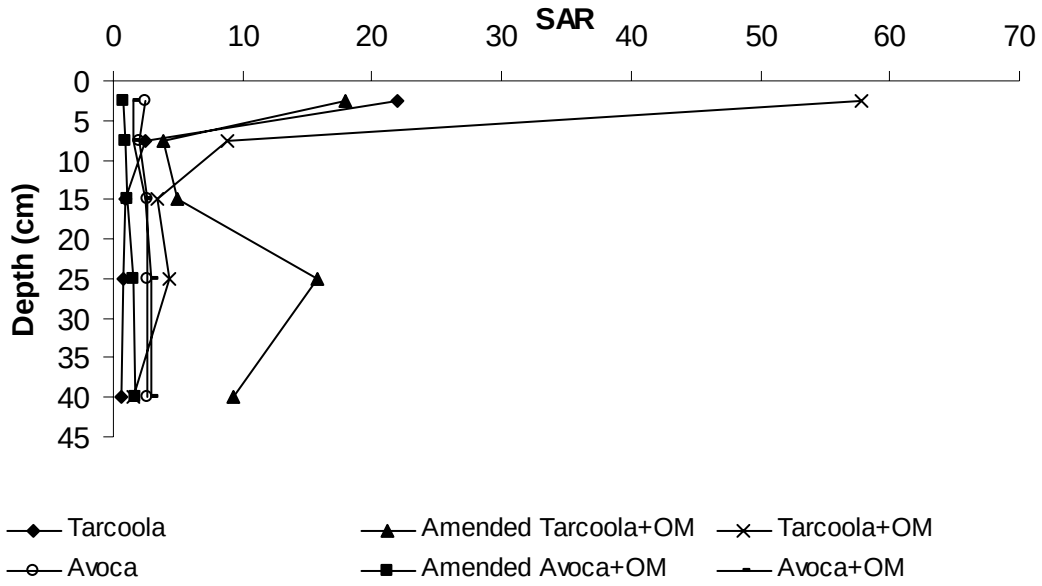


Figure 5.3 SAR profiles of the untreated Tarcoola and Avoca soils without organic material addition, and the treated Tarcoola and Avoca soils with organic material addition (OM).

The untreated soils from both *Avoca* and *Tarcoola* were highly sodic, with very high ESP, particularly in the 0-5 cm layer (Table 5.3). Following the addition of organic material, and gypsum in conjunction with organic material, the ESP decreased in the soils from both sites (Figure 5.4). At both sites and at all depths, the ESP decreased with the addition of organic material alone, but showed very little additional decrease in ESP where organic material and gypsum were added together, with the exception of the *Amended Avoca+OM* soil in the 30-50 cm layer.

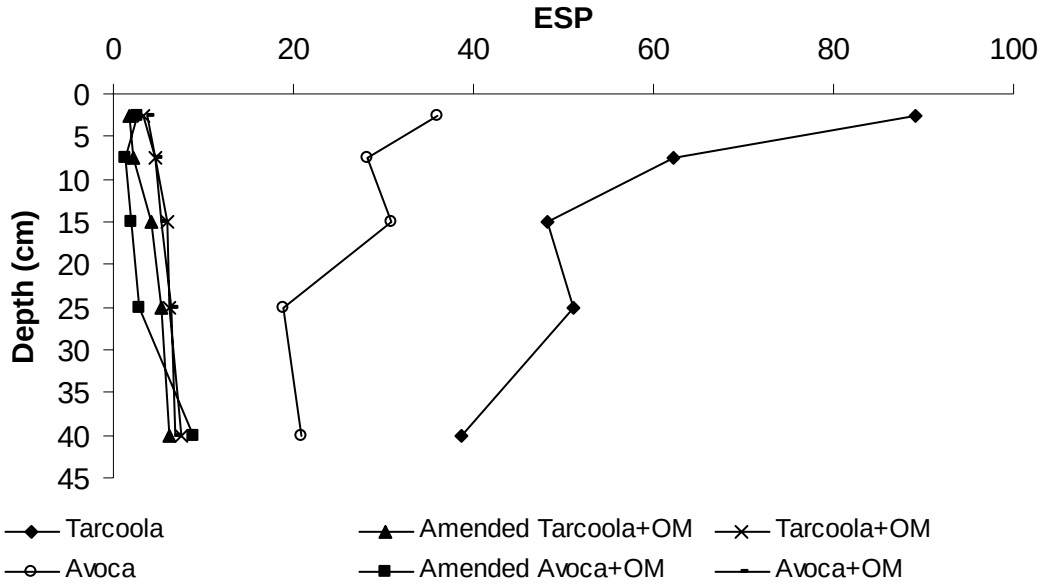


Figure 5.4 ESP profiles of the untreated Tarcoola and Avoca soils without organic material addition, and the treated Tarcoola and Avoca soils with organic material addition (OM).

SOC was relatively low in the untreated soils from both sites at all depths, with the Tarcoola soil containing < 0.5 % SOC at all depths (Figure 5.5). Following the addition of organic material, SOC increased in both soils at all depths.

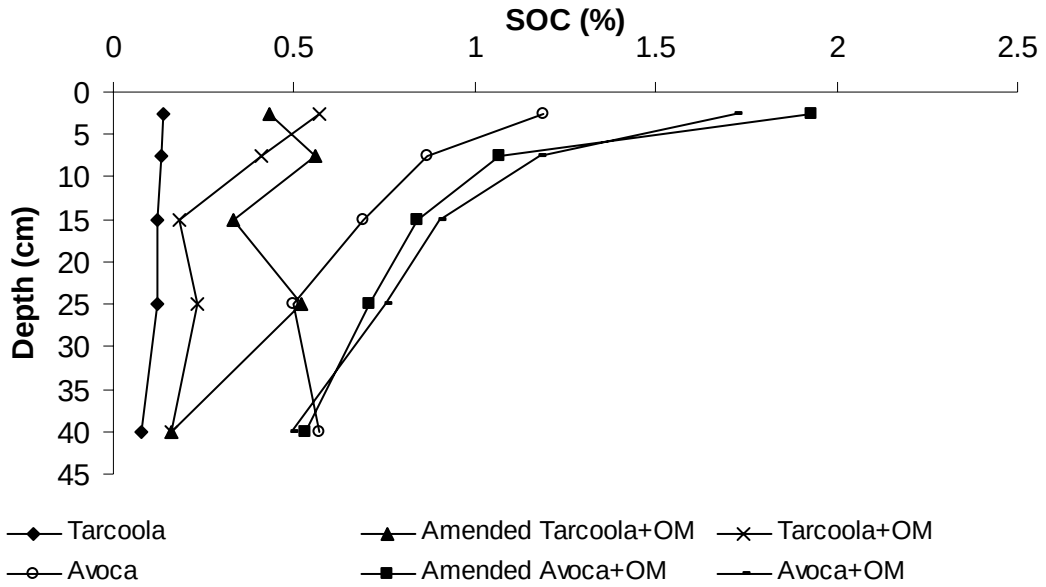


Figure 5.5 SOC profiles of the untreated Tarcoola and Avoca soils without organic material addition, and the treated Tarcoola and Avoca soils with organic material addition (OM).

No clear pattern in total N could be discerned from both sites and at all depths (Figure 5.6). The addition of organic material did not appear to influence the Tarcoola soils; however, total N generally increased in the Avoca soils throughout the profile.

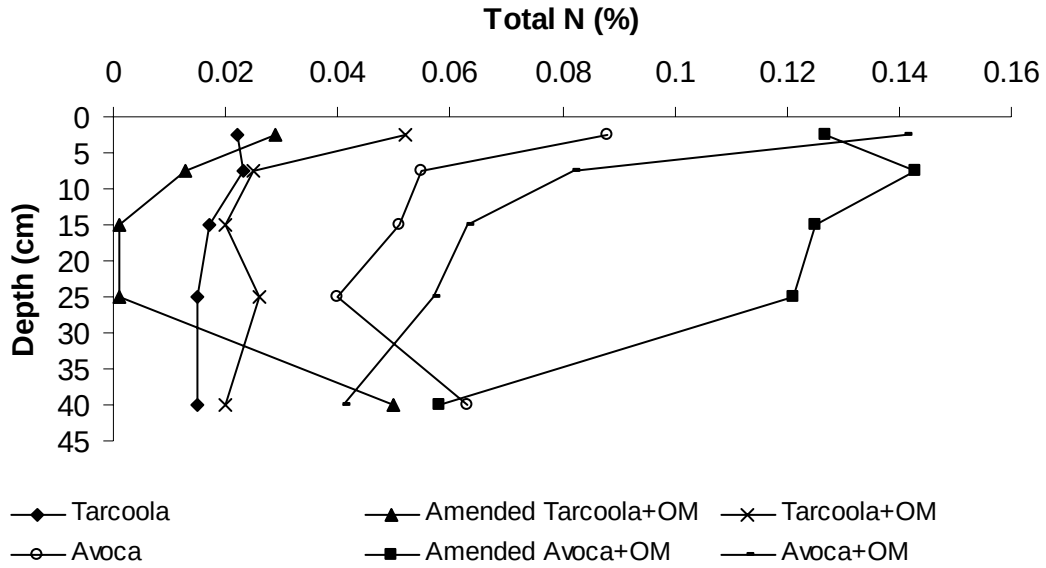


Figure 5.6 Total N profiles of the untreated Tarcoola and Avoca soils without organic material addition, and the treated Tarcoola and Avoca soils with organic material addition (OM).

Total S profiles are shown in Figure 5.7. The *Amended Tarcoola+OM* soils showed an increase in total S compared to the untreated soil at all depths except the 0-5 cm layer, while there was a decrease in the 10-20 cm layer in the *Tarcoola+OM* compared to the untreated soil. Similarly, the *Avoca+OM* soils showed a negligible difference in S compared to the untreated soil, while the *Amended Avoca+OM* showed a distinct increase.

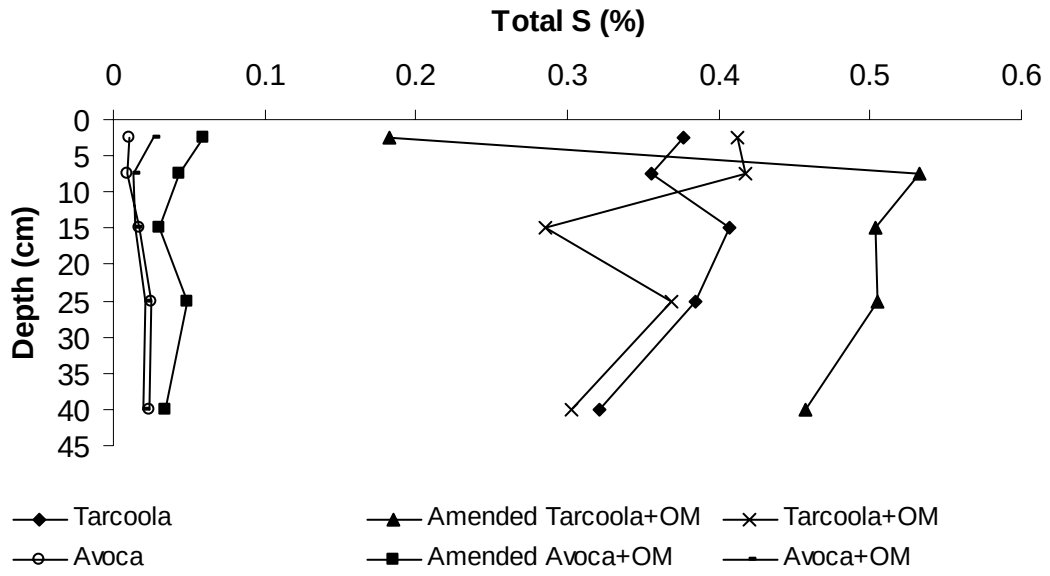


Figure 5.7 Total S profiles of the untreated Tarcoola and Avoca soils without organic material addition, and the treated Tarcoola and Avoca soils with organic material addition (OM).

5.3.2 Soil Respiration

Data pooled over the 12 weeks from both sites showed that differences in respiration rates were highly significant ($P<0.001$) with the addition of gypsum (Table 5.4), while site effects were not significantly different ($P>0.05$).

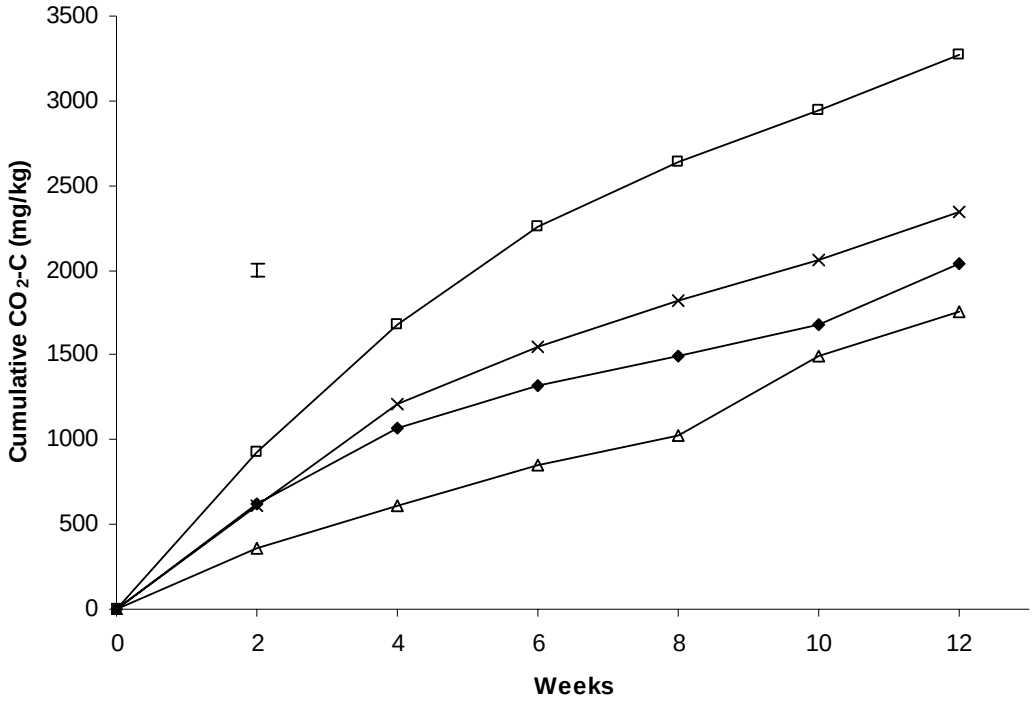
Table 5.4 Effects on respiration following organic material addition with gypsum addition and site.

Gypsum (t/ha)	**0**	**10**	***P***
Respiration rate (CO_2-C mg/kg/week)	229.36[a]	278.73[b]	$P<0.001$
Site	**Avoca**	**Tarcoola**	
Respiration rate (CO_2-C mg/kg/week)	260.45[a]	247.64[a]	NS

Note: Different letters within a row indicate a significant difference. NS indicates that no significant difference was found.

Differences in cumulative respiration rates were highly significant with gypsum addition ($P<0.001$). However, there was also a highly significant interaction between site, week and gypsum addition, as shown in Figure 5.8 ($P<0.001$). In the surface layer (0-5 cm), the *Tarcoola+OM* soil showed the lowest rates of respiration. The *Amended Avoca+OM* soils showed the highest rate of respiration over the 12 week incubation period in the surface layer (0-5 cm). At both sites, the soils amended with both organic material and gypsum showed higher rates of respiration compared to the respective soils amended with organic material alone (Figure 5.9a). Similarly, in the 5-10 cm layer, the soils amended with both gypsum and organic material displayed higher rates of respiration than their counterparts amended with organic material only (Figure 5.9b).

a)



b)

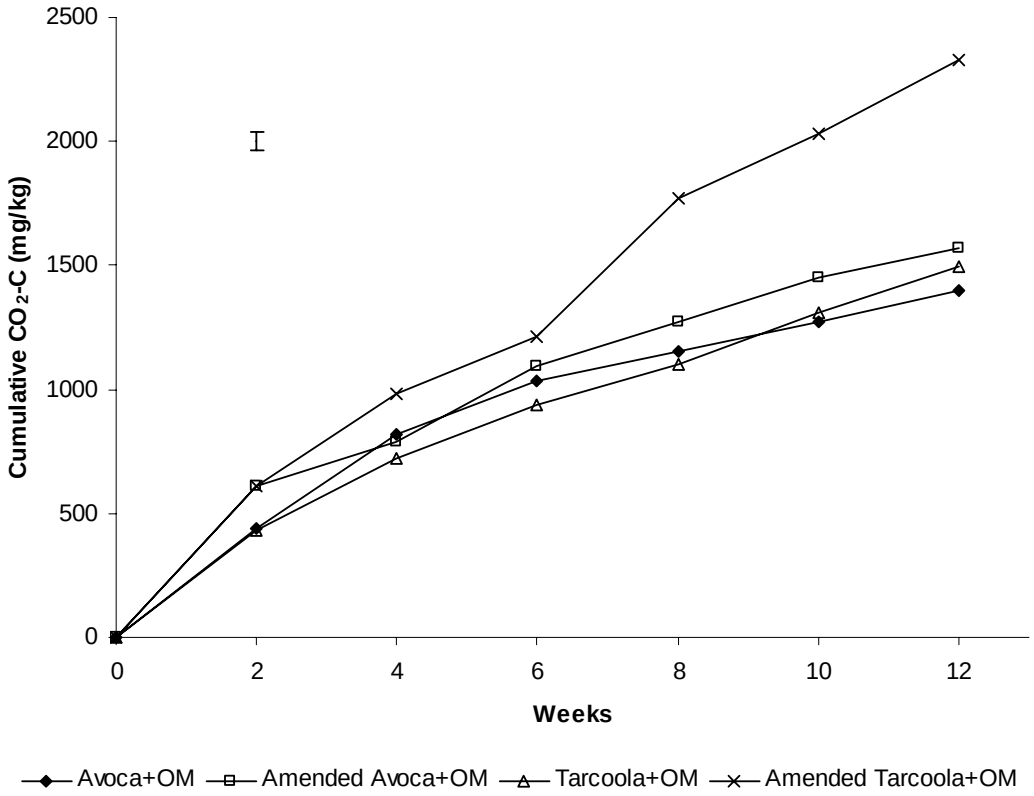


Figure 5.8 Cumulative respiration rates following organic material addition over the 12 week period with and without gypsum amendment from Avoca and Tarcoola at a) 0-5 cm and b) 5-10 cm.

Note: Vertical bar represents the SED

Differences in respiration rates due to organic material addition are shown in Table 5.5. Data pooled over the 12 weeks and from both sites showed that respiration was significantly higher following the addition of organic material (P<0.001; Table 5.5). Respiration was also significantly higher following addition of organic material when compared to the respective gypsum treatments (P<0.001; Table 5.5). Those samples with organic material incorporated showed higher respiration rates than those samples which did not have organic material incorporated. Those samples which had gypsum and organic material incorporated showed higher levels of respiration than those with gypsum incorporated alone.

Table 5.5 Effects in respiration due to organic material addition and interactions with gypsum addition

Organic material (t/ha)	**0**	**10**		
Respiration (CO_2-C mg/kg/week)	185.18[a]	256.09[b]		
Organic material (t/ha)-Gypsum (t/ha)	**0-0**	**0-10**	**10-0**	**10-10**
Respiration (CO_2-C mg/kg/week)	187.09[a]	183.00[a]	231.00[b]	281.18[c]

Note: Different letters within a row indicate a significant difference

5.3.3 Soil Microbial Biomass

Data pooled over the 12 weeks from both sites following incorporation of organic material showed highly significant differences in SMB (P<0.001) with depth and by site (Table 5.6). SMB showed a general decrease with depth; however, there was a small increase at the 20-30 cm depth. While differences with gypsum addition were not significant (P>0.05), overall SMB was significantly lower (P<0.001) at the Tarcoola site compared to the Avoca site. SMB data pooled over the 12 weeks indicated that SMB levels were similar at both sites to a depth of 10 cm, with the Tarcoola soils decreasing more with depth than the Avoca soils, below 10 cm depth (Figure 5.9).

Table 5.6 Effects in the SMB with depth, gypsum addition and site following incorporation of organic material.

Depth (cm)	**0-5**	**5-10**	**10-20**	**20-30**	**30-50**	***P***
SMB-C mg/kg	671.33[a]	337.82[b]	175.83[d]	240.25[c]	149.82[e]	*P*<0.001
Gypsum (t/ha)	**0**	**10**				
SMB-C mg/kg	287.64[a]	294.12[a]				NS
Site	**Avoca**	**Tarcoola**				
SMB-C (mg/kg)	361.38[a]	228.31[b]				*P*<0.001

Note: Different letters within a row indicate a significant difference. Data have been square-root transformed, with back-transformed means presented. NS indicates that no significant difference was found

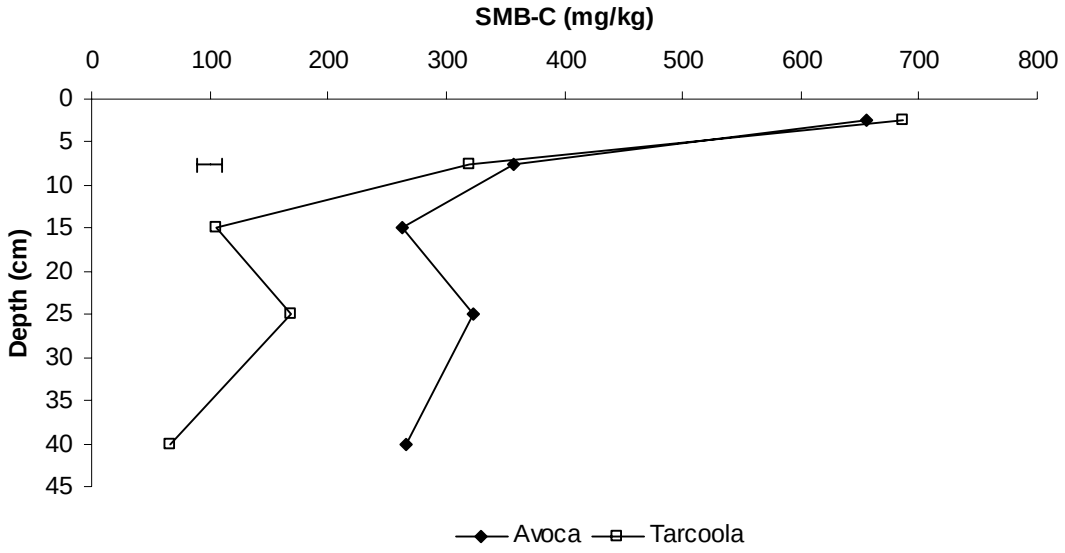


Figure 5.9 Pooled SMB data per week from Avoca and Tarcoola

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Horizontal bar represents the SED.

Apart from the 0-5 and 5-10 cm depths, there were significant interactions in the SMB between site, depth, week and gypsum addition ($P<0.01$), shown in Figure 5.10. Below the 10 cm depth, site effects become apparent, with the *Avoca+OM* and *Amended Avoca+OM* soils displaying higher levels of SMB than those from Tarcoola. In the 30-50 cm layer, the soils amended with both gypsum and organic material at the two sites showed higher levels of SMB than the respective soils amended with organic material alone from Week 2.

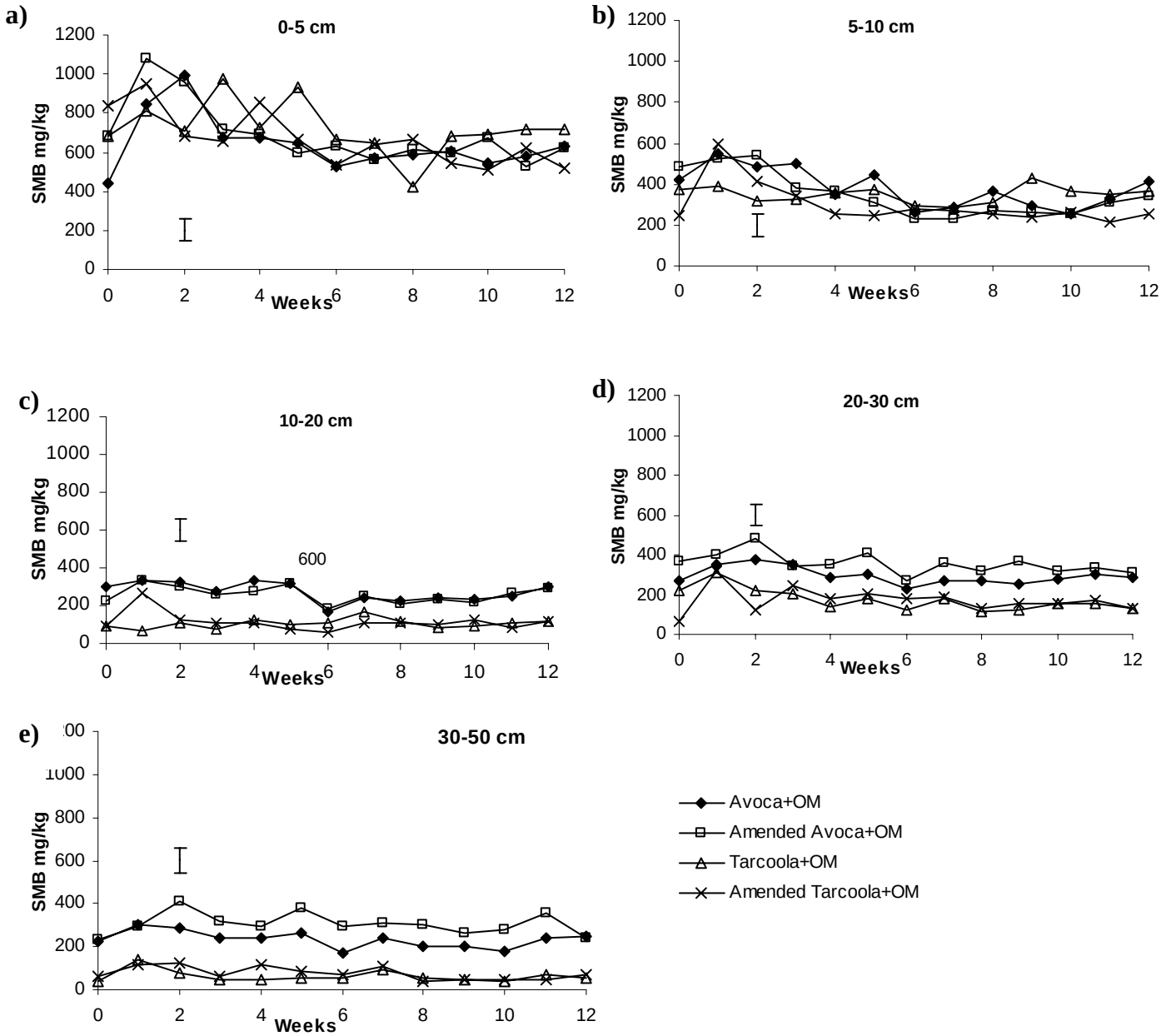


Figure 5.10 SMB-C over the 12 week period with and without gypsum amendment from Avoca and Tarcoola following organic material incorporation at a) 0-5 cm, b) 5-10 cm, c) 10-20 cm, d) 20-30 cm and e) 30-50 cm.

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Vertical bar represents the SED.

Differences in the SMB due to the addition of organic material are shown in Table 5.7. Data pooled over the 12-week experimental period showed that the SMB was significantly higher with the addition of organic material (P<0.001; Table 5.7). The SMB was significantly higher at each site following the incorporation of organic material, compared to those samples which did not have organic material incorporated (P<.001; Table 5.7). Those samples from *Avoca* displayed higher levels of SMB than those from *Tarcoola* following addition of organic material. While the addition of organic material resulted in significantly higher levels of SMB compared to those without organic material, there were no significant differences when gypsum was also incorporated.

Table 5.7 Effects due to organic material addition and interactions with gypsum addition in the SMB-C

Organic material (t/ha) **SMB-C (mg/kg)**	**0** 20.16^{a}	**10** 292.07^{b}		
Organic material (t/ha)-Site	**0-Avoca**	**10-Avoca**	**0-Tarcoola**	**10-Tarcoola**
SMB-C (mg/kg)	19.36^{a}	361.00^{c}	20.88^{a}	230.43^{b}
Organic material (t/ha)-Gypsum (t/ha) **SMB-C (mg/kg)**	**0-0** 20.07^{a}	**0-10** 20.16^{a}	**10-0** 289.00^{b}	**10-10** 293.44^{b}

Note: Data have been square-root transformed with back-transformed means presented. Different letters within a row indicate a significant difference

Data pooled over the 12 weeks from both sites showed differences in the SMB following organic material addition with depth, as shown in Table 5.8. The SMB was significantly higher at all depths following incorporation of organic material compared to those samples which had no organic material incorporated at each respective depth (P<0.001, Table 5.9).

Table 5.8 Effects due to organic material addition and interactions with depth in the SMB-C (mg/kg).

Organic material	**Depth (cm)**				
(t/ha)	**0-5**	**5-10**	**10-20**	**20-30**	**30-50**
0	44.76^{a}	24.50^{b}	12.89^{c}	6.66^{d}	19.10^{bc}
10	672.36^{e}	337.46^{f}	14.27^{g}	203.63^{h}	152.52^{i}

Note: Data have been square-root transformed with back transformed means presented. Different letters indicate a significant difference.

5.3.4 Microbial Indices

Table 5.9 shows the effects of gypsum incorporation on qCO_2 and C_{mic}:C_{org} in the 0-5 and 5-10 cm layers following organic material addition at the end of the 12-week incubation period. The qCO_2 was higher in the soils with gypsum incorporation in the 0-5 cm layer. There were no trends in the qCO_2 between sites in both the 0-5 and 5-10 cm layers. The C_{mic}:C_{org} was slightly lower in the soils which had gypsum incorporated compared to their unamended counterparts in the 0-5 cm layer.

Table 5.9 Effects of gypsum addition and site on qCO_2 and C_{mic}:C_{org} following incorporation of organic material.

Depth (cm)	Site	Gypsum (t/ha)	qCO_2 (mg CO_2-C/d/mg SMB-C)	C_{mic}:C_{org}
0-5	Avoca	0	0.036	3.66
	Avoca	10	0.062	3.24
	Tarcoola	0	0.029	11.73
	Tarcoola	10	0.053	9.84
5-10	Avoca	0	0.040	3.57
	Avoca	10	0.048	3.22
	Tarcoola	0	0.049	7.93
	Tarcoola	10	0.110	4.52

5.4 Discussion

5.4.1 The effects of organic material and gypsum on soil properties

The decrease in pH in the Tarcoola soils following the addition of organic material (Figure 5.1) results from a number of processes. With the addition of organic material, microbial respiration is increased, described in Section 5.4.2. The pH is lowered as a result of: i) organic acid produced during the decomposition of organic material; and ii) the H^+ is increased (Equation 5.3; Nelson and Oades 1998) as a result of increased P_{CO2}.

$$2CO_{2(gas)} + H_2O \leftrightarrow H_2CO_3 + CO_{2(aq)} \leftrightarrow 2HCO_3^- + 2H^+ \qquad \textbf{Equation 5.3}$$

This effect was greatest where organic material was added in conjunction with gypsum; the effects due to gypsum addition are described in Section 4.4.1. Similarly, Chorom and Rengasamy (1997) found a greater decrease in pH in a highly alkaline soil with the combined addition of gypsum and green manure, compared with the addition of green manure or gypsum alone, through the additional production of protons from fatty acids.

Malik and Haider (1977) also found pH and ESP decreased following the addition of plant material due to increased CO_2 evolution and humic acid formation. Where the pH was less than 5, as was the case in the Avoca soils (Figure 5.1), the addition of organic material and the concomitant production of organic and carbonic acids is likely to have had a negligible effect on pH (Nelson and Oades 1998), and may have buffered any potential increase in pH. Although this was not confirmed in this study, additional plant residues also have the potential to increase pH through processes such as the microbial decomposition of organic anions, which results in a release of alkalinity and ammonification of N in the added plant material (Equation 5.4; Xu *et al.* 2006). It may be that the soil pH in the untreated Avoca soils was not low enough for this process to be apparent, or alternatively, the pH was low enough to inhibit the micro-organisms responsible for this process.

$$NH_3 + H_2O \leftrightarrow NH_4^+ + OH^- \qquad \textbf{Equation 5.4}$$

EC was increased with the incorporation of organic material in both the soils amended with gypsum and those without (Figure 5.2). Increasing EC due to gypsum addition was due to the dissolution of gypsum and the electrolyte effect, as described in Section 4.4.1. The increase in EC in the soils amended with organic material alone at both sites was most likely due to an increase of ions in solution, which may have resulted from mineral dissolution caused by the increase in P_{CO2} (Sekhon and Bajwa 1993), or the formation of organic acids.

Not surprisingly, the addition of organic material resulted in an increase in SOC (Figure 5.5) due to the addition of C contained in the plant material. The incorporation of kangaroo grass at a rate of 10 t/ha is equivalent to the addition of approximately 4 t/ha of SOC. However, the measured SOC levels following incorporation of kangaroo grass were lower as the kangaroo grass was coarsely ground prior to incorporation. Due to the method of sample preparation used for the determination SOC, some of the kangaroo grass was removed by sieving prior to analysis. Subsamples of approximately 10 g were passed through a 2 mm sieve after the incorporation of kangaroo grass to remove gravel sized particles. However, following sieving, some organic material which was too long to pass through the 2 mm mesh remained in the sieve, resulting in an underestimation of SOC levels, and may be why the curves in Figure 5.5 do not track parallel to the unamended soils. Similarly, the addition of gypsum increased total S at all depths

except in the 0-5 cm layer of the *Amended Tarcoola+OM* sample. This is most likely due to the heterogeneity of the soil sample, and the small amount of subsample required for analysis (< 0.5 g). Total N content did not increase at any depths in either the Tarcoola or Avoca soils following the addition of organic material. This was most likely due to the low N content of the kangaroo grass that was incorporated (≈ 0.5 %; Section 5.2.3), and N additions within the error of measurements.

5.4.2 Microbial activity and the soil microbial biomass

Just as increased salt concentration increased substrate availability for the SMB through the dissolution of SOM, as described in Chapter 4, the addition of organic material in the short term provides additional substrates for the microbial population. This also resulted in an increase in the level of respiration (Table 5.5) and SMB (Tables 5.6 and 5.7). The increase in respiration and SMB occurred despite the large C:N ratio of 76 found in the kangaroo grass used in this study, indicating that plant C was still available to microorganisms for both biomass increase and respiration activity. Also, in a process similar to that found in Chapter 3, this additional substrate may still relieve osmotic and pH stress on the microorganisms, while improving soil physical and chemical conditions (Chander *et al.* 1994). The presence of SOM can provide a buffer to the soil solution and to soil microorganisms and their activity, particularly where salinity or sodicity increases (McCormick and Wolf 1980), or as in this study, under adverse soil pH conditions and where salinity and sodicity levels are already high.

The higher levels of soil respiration and SMB due to increased substrate in this chapter compared to a scalded soil which did not have organic material added, as discussed in Chapter 4, may have also been aided by the partial breakdown of the added organic material in the case of the high pH Tarcoola soils. Laura (1973) found that higher alkalinity increases mineralisation of organic matter as a result of dissolution of plant C compounds, thus increasing its susceptibility to decomposition. Nelson *et al*. (1996) have suggested, in the case of sodic soils, that Na^+ dissolves readily decomposable plant components and microbial metabolites following the initial addition of organic material, while a small portion of the native organic matter is constantly available due to the solubilisation processes by Na^+.

However, while high pH increases mineralisation, low soil pH levels have been shown to depress microbial activity as a result of reduced substrate utilisation efficiency

(Sadinha *et al.* 2003). Substrate availability can also be restricted under acidic conditions due to the effects of acidity on substrate availability, and hence, SMB and respiration. The effects are largely attributable to interactions at low pH that result in the formation of Al-organic matter or Fe-organic matter complexes (Brunner and Blaser 1989; Xu *et al.* 2006). However, soluble Al and Fe concentrations in the Avoca soils were low, with exchangeable Al and Fe below the detection limit of the ICP-AES (see Tables C1 and C2, Appendix C). Therefore, the higher levels of SMB and soil respiration of the *Avoca+OM* and *Amended Avoca+OM* soils compared to the respective counterparts without organic material, as discussed in Chapter 4, are predominantly the result of additional, easily accessible and decomposable substrate.

In the absence of pH or aeration effects, sodicity has been found to increase, and salinity decrease, the decomposition of plant material indicated by an increase in the respiration rate (Nelson *et al.* 1996). In this study, however, the addition of gypsum and organic matter caused large changes in EC and pH in the *Tarcoola* soils, while the EC increased without the concomitant pH changes in the *Avoca* soils. Despite changes in EC and pH, the SMB did not appear to be altered, but respiration rates were increased. The higher levels of sodicity in the *Tarcoola* soils may have contributed to the differences found in the SMB, particularly at depth, as the substrate may have become coated with dispersed clay at high sodicity levels (Nelson *et al.* 1997), although this could not be confirmed.

It is suggested that a dormant population of salt-tolerant SMB is present in the salt-scalded soils in this study, which has become adapted to such environmental conditions over time. Following the addition of organic material, the population multiplies rapidly due to the availability of substrate. Similarly, Sarig and Steinberger (1994) found, in saline conditions, the highest amount of SMB occurred under a desert halophyte following the addition of litterfall resulting in a newly available substrate to microorganisms. Enzymatic activity can also increase following the addition of organic material, which aids in microbial activity while improving nutrient availability in the untreated soil (Liang *et al.* 2005; Tejada *et al.* 2006). Microbial and enzymatic activity is most likely stimulated due to the increase in readily utilisable energy sources. McCormick and Wolf (1980) found that the addition of organic material can act as a buffer to salinisation processes. The deleterious effects of NaCl were reduced following the addition of a readily available substrate while respiration was less affected in soils amended with alfalfa than those that were left unamended. However, Rasul *et al.* (2006)

suggested that the effect of the addition of substrate on the subsequent SMB is dependent on the ratio of substrate to the initial microbial biomass, which may have also been the case at depth in this experiment. Higher levels of SMB in the *Avoca* soils compared to the *Tarcoola* soils at depth are most likely due to a higher initial microbial biomass because of higher SOC levels in the former.

Following the incorporation of organic material, the qCO_2 was generally lower (Table 5.9) compared to the qCO_2 from a scalded soil that did not have organic material incorporated (Table 4.5). It is likely that the addition of organic material for decomposition alleviates stress on the microbial biomass by providing additional substrate, as described previously. Therefore, with the addition of organic material, the qCO_2 decreased as the microbial population increased due to the additional substrate of high C:N ratio, despite hostile environmental conditions, in both alkaline and acidic soils.

The C_{mic}:C_{org} was higher in those soils with organic material addition (Table 5.9) compared to those without (Table 4.6). Similarly, a previous study showed that salinity intensifies stress on the microbial community under acidic conditions, which was indicated by a reduction in the SMB to SOC ratio and an increase in the specific respiration rate (Rasul *et al.* 2006). In the same study by Rasul *et al*. (2006), it was found that the SMB and CO_2 production were similar in both soils after the addition of a complex organic amendment in the form of sugar cane filter cake, with the SMB and CO_2 production linearly related to the amount of filter cake added and was not affected by the differences in the initial SMB content. However, in the current experiment, it is likely that the C_{mic}:C_{org} has been overestimated due to the level of SOC being underestimated, as described previously. This may have been the case particularly in the Tarcoola soils in the 0-5 cm layer, where the C_{mic}:C_{org} was approximately 12. Moreover, it is likely to be transitory, as C substrate for microbial biomass synthesis becomes limiting over time. However, it should be noted that the $qCO2$ and C_{mic}:C_{org} provide an indication of trends only, as the indices were only calculated at the end of the experimental period, with further research required to determine whether the differences found in Section 5.3.4 and Section 4.3.4 are significant.

5.4.3 The effects of gypsum

While the addition of gypsum improved the soil environment, as shown in this chapter by the decrease in pH in the *Tarcoola* soils, it also caused an increase in EC in soils that were already saline. It has been noted that Cl^- salts are more toxic to microbial activity, in terms of nitrification, than the corresponding sulfate salt (McCormick and Wolf 1980), and may also apply to C mineralisation. Most of the salinity in Australia is due to Cl^- (Naidu and Rengasamy 1993). At equal molar concentrations, Cl^- salts have the potential to be more toxic to biological activity compared to the SO_4^{2-} counterpart, due to the higher activity of Cl^- ions, and the potential for SO_4^{2-} to precipitate with Ca^{2+} (Garcia and Hernandez 1996). Similarly, Baldock and Oades (1989) found that at equal EC levels but different Ca concentrations, Ca^{2+} did not influence microbial activity.

Following the addition of gypsum, respiration rates increased, which may be attributed to the decline in pH or more amenable environmental conditions as in the case of the *Tarcoola* soils. Batra and Manna (1997) found that microbial activity is linked to soil pH and levels of SOC. Despite up to a five-fold increase in EC following the addition of gypsum in this study, there were no distinct differences in trends in the SMB from either the *Avoca* or the *Tarcoola* soils, both with and without gypsum addition. This indicates that osmotic stresses were not great enough to affect the microbial population in the short term. Where soils are saline, osmotic stress usually limits microbial growth and activity, while under sodic conditions, ion toxicities and adverse pH conditions may also inhibit microbial growth (Rietz and Haynes 2003). However, the results in Chapter 3 indicate that microbial activity need not be suppressed by high salt concentration nor high pH conditions, as the microbial population may be well-adapted to such environmental conditions (Beltran-Hernandez *et al.* 1999).

Previous studies have also shown an interaction between the addition of gypsum and the incorporation of organic matter. The addition of Ca compounds with organic materials can decrease spontaneous dispersion (Vance *et al*. 1998), and thus, have an additive effect of improving aggregate stability (Muneer and Oades 1989b). Furthermore, Muneer and Oades (1989b) found that C mineralisation rates were also decreased, indicating that losses of organic matter from soils can be decreased with the addition of gypsum. This may be due to increased aggregate stabilisation from the formation of Ca-organic linkages in the form of clay particle-Ca-organic molecule (Baldock et al. 1994). The biological stabilisation of substrate C is suggested to result from the organic

compounds synthesised by the biomass utilising the substrate, presumably carboxylic materials which are capable of forming Ca^{2+}-organic complexes (Baldock and Oades 1989). Loss of SOC decreases during the formation of soil aggregates, as SOM becomes physically protected and inaccessible for microbial decay. Similarly, concentrations of DOC in soils have decreased following gypsum application due to the inhibition of decomposition of organic matter by microorganisms and the reduced release of DOC by leaching (Suriadi et al. 2002). The addition of organic matter may also promote flocculation by increasing the EC and hence, improve soil structural stability (Tejada et al. 2006) while providing physical protection to SOM.

Furthermore, Muneer and Oades (1989a) found a reduction in the mineralisation of glucose with the addition of Ca compounds in the form of either lime or gypsum, with gypsum being more effective than lime. This can be attributed to stabilisation by microbial products leading to prolonged stabilisation of macroaggregates, in addition to flocculation of dispersive soils due to an electrolyte effect. It was assumed by Muneer and Oades (1989a) that glucose decomposition was not inhibited, with the reduction in mineralisation due to the stabilisation of the products of decomposition. Glucose is soluble and readily degradable, hence, is decomposed rapidly. In this study, kangaroo grass was used which is comparatively less easily decomposed. Therefore, it produces lower amounts of microbial residues, with the stabilisation effect of decomposition products, and hence physical protection, likely to be small.

In terms of reclamation, increasing SOM increases the re-establishment of soil nutrient cycles, and the retention and supply of these nutrients, especially N (Mummey *et al.* 2002). This highlights the importance of plant cover and the associated SOM inputs in the reclamation of bare soils. During periods of growth, roots provide substrate in the form of exudates, sloughed-off material and dead roots (Buyanovsky and Wagner 1995). This chapter has demonstrated that soil microbial activity can be restored following addition of organic material in highly degraded salt-scalded areas at both low and high pH levels. Despite hostile soil environmental conditions, it appears that these systems are limited by substrate supply rather than by adverse soil conditions. Hence, if rehabilitation efforts are successful in re-introducing plant growth into bare areas, through the initial addition of organic material or gypsum, or a combination of both, the production of organic material and, therefore, SOM, can become self-sustaining and aid in the restoration of soil ecosystem processes.

5.5 Summary and Conclusion

Chapter 4 demonstrated that those soils sampled from scalded areas without organic material addition showed low levels of SMB and respiration. The results from this chapter demonstrate that while SMB levels are low in scalded soils, where organic material is available as substrate for decomposition, the microbial population is still active and present. It is likely that a dormant population of salt-tolerant micro-organisms exists in soils that have been degraded and hence, scalded, for a significant period of time, which can multiply rapidly when substrate is readily available. Despite the large increases in EC caused by the addition of gypsum, microbial respiration does not appear to be adversely affected, while the increase in SMB may improve soil structure by increasing fungal hyphae, mucilages, and other decomposition products. Therefore, it is apparent from this study that decomposition processes are limited by substrate rather than by the deleterious soil conditions commonly found in salt-scalded areas. While biomass production is likely to be limited at the soil pH values found in this study, the potential still exists for these degraded areas to return to functioning soil ecosystems if rehabilitation is successful in remediating adverse soil pH and EC conditions, and plant production is re-established in both alkaline and acidic conditions. Chapters 3, 4 and 5 have demonstrated the effects of salinity and sodicity on C fluxes under controlled conditions in the laboratory. Chapter 6 will determine the level of SOC stocks in the field in salt-scalded, revegetated and unaffected soil profiles.

CHAPTER 6: CARBON STOCKS IN SALT-SCALDED AND NON-SCALDED LANDSCAPES

6.1 Introduction

Increasing soil salinity and sodicity currently cause significant impacts on agricultural production and native vegetation. These impacts are predicted to increase in the future. The processes associated with salinisation in terms of altered hydrology and its effects on plant health have been extensively reviewed and described in Section 2.2.1.

In salt-affected soils, plant growth is restricted by osmotic and specific ion effects, low availability of plant nutrients and indirect effects related to adverse soil physical properties. The degree of salinisation can range from slight salinisation with marginal impact on crop production to the development of extensive salt scalds. C accounting in saline and sodic areas is complicated by topographic factors. Because soil C efflux and stocks are dependent on clay content and soil moisture (Jobaggy and Jackson 2000), processes which commonly occur in saline and sodic areas such as waterlogging in lower parts of the landscape can enhance C sequestration. Increasing clay content with depth will also enhance SOC concentrations (Bird *et al.* 2001), while scalding increases susceptibility to erosion and hence, enhances SOC loss. Decomposition processes are slowed due to the formation of massive structure, commonly found in sodic soils, as substrate availability is limited to the microbial population (Nelson and Oades 1998). Difficulties can also arise when assessing C stocks in revegetated saline, sodic and saline-sodic areas due to high spatial and temporal variability.

This chapter aims to determine the level of SOC and the associated soil properties in saline-sodic scalds, eroded, revegetated and unaffected soil profiles. This will ascertain the amount of SOC lost due to salinisation and sodication relative to an unaffected soil profile, and the increase in the level of SOC following revegetation.

6.2 Materials and Methods

6.2.1 Site Descriptions and Field Sampling

Soil samples were taken from two sites in the Southern Tablelands region of NSW; "Tarcoola" located at Bevendale and "Gunyah" located at Rugby (Figure 6.1) with comparisons made between scalded and non-scalded soil profiles. A paired sites approach was undertaken to to estimate the loss of soil carbon as a consequence of salt-scalding, with the experimental site set-up shown in Table 6.1.

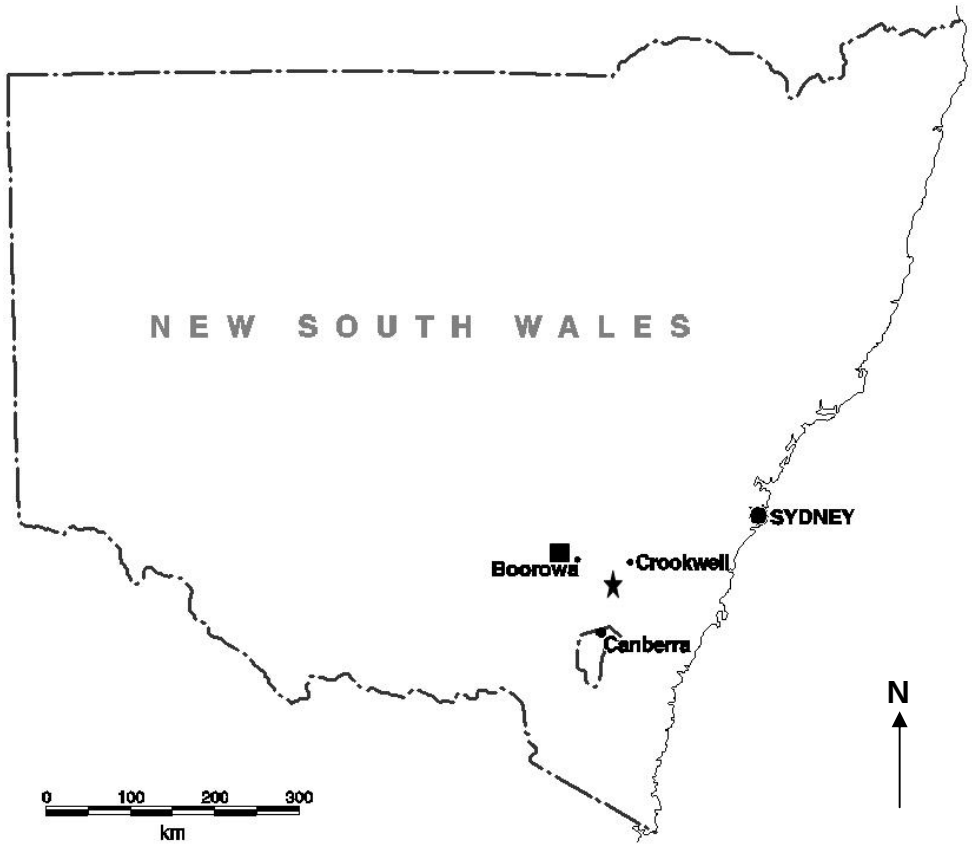


Figure 6.1 Location of fieldsites
Note: Square indicates Gunyah site, star indicates Tarcoola site

Table 6.1 Site set-up

Site	**Tarcoola**			**Gunyah**			
Microsite	Scald	Vegetated	Depression	Eroded	Scald	Revegetated	Vegetated

6.2.1.1 Tarcoola Site

The first site was located on a property, "Tarcoola" in Bevendale (34 30' 45" S, 149 05' 00" E), approximately 40 km south-west of Crookwell. An area on an adjacent property, "Riverview" was also used. The area has an average annual rainfall of 660 mm. In January, the average daily maximum temperature for the area is 29.2^{o}C. In July, the average minimum temperature is 1.3^{o}C, as determined from a nearby weather station.

The scalded and vegetated paired sites were located on the same landform element of a footslope, with a vegetated depression microsite added for comparison. The scalded microsite was located in a bare scalded patch (termed *Tarcoola Scald*) while the vegetated microsite was located on the same footslope approximately 100 m to the east of the scalded site (termed *Tarcoola Vegetated*). A further microsite was located on a non-scalded, vegetated patch in a drainage depression (termed *Tarcoola Depression*), along approximately the same contour approximately 100 m to the north of the *Tarcoola Scald* site.

At each microsite, soil pits were excavated to dimensions of at least 2 m wide * 5 m long * 1 m deep with a mini-excavator. Two soil profiles were described at the ends of each pit and bulk density cores were sampled using stainless steel cores of volume 209.81 cm^3 (Plate 6.1). Triplicate samples taken from depths 0-5, 5-10, 10-20, 20-30, 30-50, 50-70 and 70-100 cm of each profile, giving six replicates per soil pit in total. Due to the length of the soil pits, it is assumed that sampling at each end of each pit will address the heterogeneity common to salt-affected sites. The samples were transported back to the laboratory in polyethylene bags for analysis, as described in Section 6.2.2. At the *Tarcoola Vegetated* microsite, the soils were sampled from the side of a gully using profiles approximately 5 m apart. The gully walls were scraped back approximately 50 cm, and then sampled according to the methods described above to a depth of 50 cm. Due to difficulties encountered at the 50 cm depth from the presence of a pebble layer, samples were only taken to this depth.

The *Tarcoola Scald, Tarcool Depression and Tarcoola Vegetated* microsites are shown in Plates 6.2, 6.3 and 6.4, respectively. The vegetated areas (*Tarcoola Depression* and T*arcoola Vegetated*) were dominated by Red Grass (*Bothriochloa* spp) with minor occurrences of Couch (*Cynodon dactylon*). The soil types at the *Tarcoola Scald,*

Tarcoola Depression pit and *Tarcoola Vegetated* microsites were all Yellow Sodosols (Isbell 1996). The locality is underlain by undifferentiated Ordovician and Silurian metasediments (Hird 1991), with the property predominantly grazed by sheep.

6.2.1.2 Gunyah Site

The second site was located on a property, "Gunyah" in Rugby, approximately 35 km east of Boorowa (Figure 6.1; 34° 29' 0.32" S 149 ° 1' 27.99" E). Climate data have been taken from Boorowa, the nearest meteorological station, with average annual rainfall of 610 mm. The average maximum temperature for the area in January is 29.5^{o}C and average minimum temperature is 0^{o}C in July. All the microsites (soil pits) were located on a lower footslope position in the landscape, and were within 300 m of each other. Four soil pits (microsites) were excavated with a mini-excavator, with dimensions similar to those described in Section 6.2.1.1. The soil pits were located as follows: on a bare scalded patch which had been eroded and hence, had lost its A horizon (*Gunyah Eroded;* Plate 6.5); a bare scalded patch which had not been eroded (*Gunyah Scalded;* Plate 6.6); a vegetated patch of what is assumed to be the original vegetation (*Gunyah Vegetated;* Plate 6.6); and an area that had been reclaimed by revegetation (*Gunyah Pasture;* Plate 6.6). The area of the *Vegetated* soil pit was dominated by wallaby grass (*Austrodanthonia bipartita*) with minor occurrences of kangaroo grass (*Themeda australis*). Reclamation of saline patches, which highlight the spatial variability of salinity-issues, had been undertaken by the landholder with the use of salt-tolerant pasture, namely Tall Wheatgrass (*Thinopyron ponticum*) approximately 10 years ago. Reclamation of the *Gunyah Pasture* site involved fencing the area to exclude stock and revegetation with Tall Wheatgrass, with no additional treatment of any amendment. It is assumed that the *Pasture* microsite was very similar to the *Scald* and *Eroded* microsites prior to revegetation.

Soils were sampled according to the method described above with the exception of the *Pasture* profiles, which were sampled to a depth of 70 cm due to difficulties experienced in placing the cores into the profile at depth. The soil types were a Red Sodosol (Isbell 1996) at the *Gunyah Eroded, Gunyah Vegetated* and *Gunyah Scald* profiles and a Red Kurosol (Isbell 1996) at the *Gunyah Pasture* profile. The locality is underlain by undifferentiated Ordovician and Silurian metasediments (Hird 1991), with the property predominantly grazed by sheep.

Plate 6.1 The bulk density corer used to obtain bulk density cores

Plate 6.2 Location of the *Tarcoola Scalded* soil pit

Plate 6.3 Location of the *Tarcoola Depression* soil pit (foreground)

Plate 6.4 The *Tarcoola Vegetated* site

Plate 6.5 Location of the *Gunyah Eroded* soil pit. The red circle highlights the loss of topsoil

Plate 6.6 Location of the *Gunyah Scalded*, *Gunyah Pasture* and *Gunyah Vegetated* soil pits

6.2.2 Laboratory Analysis

Following transportation to the laboratory, the bulk density samples were weighed before subsamples of approximately 50 g were oven dried at 105^{o}C for 24 hours to determine the moisture content. Bulk density was determined according to the method described in Section A1.1 in Appendix 1. Samples were then subjected to the same chemical analysis as described in Section 3.2.3. The remainder of the sample was air dried. Soil pH, EC and soluble cations were determined on 1:5 soil:water extracts. Where the $EC_{1:5}$ > 0.3 dS/m, soluble salts were removed with an ethanediol/ethanol wash, described in more detail in Appendix A, according to the method of Rayment and Higginson (1992). Exchangeable cations were extracted with 1 M ammonium acetate (CH_3COONH_4) buffered to a pH of 7. Soluble and exchangeable cations were analysed by ICP-AES and were used to determine the SAR and ESP.

Organic carbon and total nitrogen (N) were determined by high temperature combustion on a LECO CNS-2000 analyser. Inorganic C was removed with sulphurous acid where the pH ≥ 7. SOC stocks were determined according to the Equation 6.1 and summed to 30 cm, which gives the numerical equivalent in t/ha.

SOC (t/ha) = D * BD * C **Equation 6.1**

Where D = thickness of soil layer (cm)
BD = bulk density (g/cm^3)
C = soil organic carbon (%)

The three samples from each soil profile were subsampled and bulked for particle size analysis. Particle size analysis was undertaken using the hydrometer method (Bouyoucos 1936). The mean values of the two profiles are presented.

6.2.3 Statistical Analysis

Data were analysed using the GENSTAT 8.0 statistical analysis program (Payne 2005). The sites were analysed as a split-plot design. Data were subjected to a non-orthogonal ANOVA, as the data were unbalanced. The SAR and SOC data were square-root transformed to satisfy the assumptions for normal distribution for ANOVA, with back

transformed means presented. Differences in soil properties (bulk density, pH, EC, SAR, ESP, SOC, N, and Ca) due to sites by microsites and depth were analysed. Due to the split-plot design and lack of replication, differences between sites (ie. Tarcoola and Gunyah) and between microsites (ie. Vegetated, Scald, Depression and Pasture) were not analysed. Where significant differences were found (P<0.05), data were subjected to LSD testing at the 5% level. Correlations between soil properties were undertaken using a correlation matrix. Where correlations were found, SOC was set as a dependent variate with soil properties fitted after site factors in a non-orthogonal ANOVA.

6.3 Results

Descriptions of the soil profiles sampled are given in Appendix D.

6.3.1 Soil Bulk Density and Particle Size Analysis

Soil bulk density profiles are shown in Figure 6.2 for a) Tarcoola, and b) Gunyah. There were significant interactions in bulk density between site, microsite and depth (P<0.05). The *Tarcoola Scald* profile had higher bulk density values at all depths compared to the *Tarcoola Depression* and *Tarcoola Vegetated* profiles. The lowest bulk density values at the Tarcoola site were found in the *Tarcoola Depression* profile at all depths. At the 0-5 cm depth bulk density was higher in the *Gunyah Eroded* profile than in the *Gunyah Scald*, *Pasture* and *Vegetated* profiles. From 5-50 cm, the bulk density was higher in the *Gunyah Scald* and *Gunyah Eroded* profiles compared to the *Gunyah Pasture* and *Gunyah Vegetated* profiles.

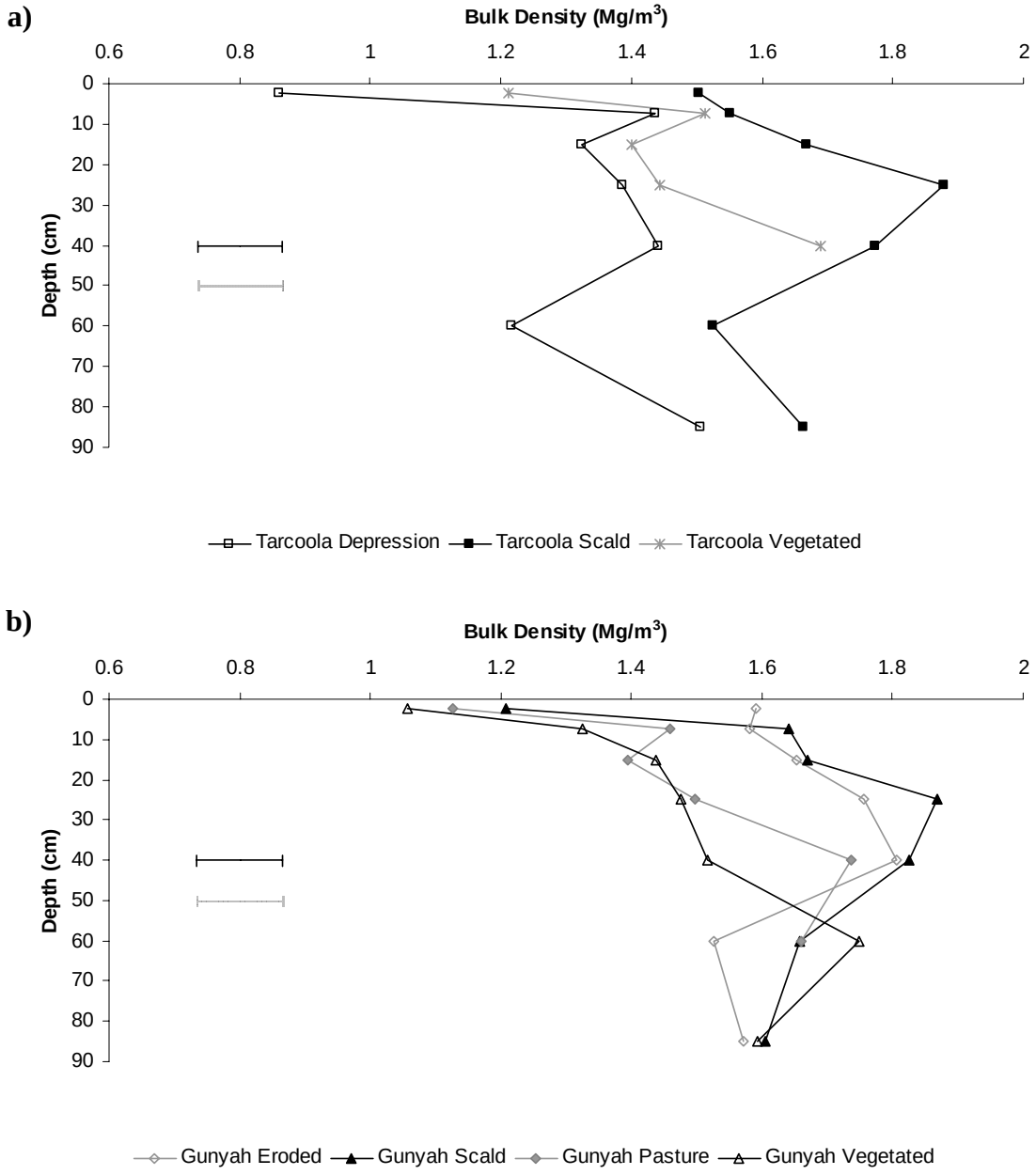


Figure 6.2 Oven-dried soil bulk density profiles from a) Tarcoola, and b) Gunyah

Note: Solid horizontal line indicates LSD for microsite effects; dashed line indicates the LSD for depth effects.

Particle size distribution at each microsite for both the Tarcoola and Gunyah sites are shown in Figures 6.3 and 6.4, respectively. A more detailed table of the particle size analysis is shown in Table D2 in Appendix D. The mean values of each depth from the two soil profiles in each pit are presented in Figures 6.3 and 6.4. The *Tarcoola Scald* profile increased in clay content with depth, while the *Tarcoola Depression* and *Tarcoola Vegetated* profiles generally had a uniform texture (Figures 6.3b and 6.3c, respectively). The *Gunyah* profiles all showed a general increase in clay content and

decrease in sand content with depth. At both sites, there was no obvious relationship between texture and bulk density values for all profiles.

The soil texture at the *Tarcoola Scald* profile was a loamy sand at the surface which graded with depth to a sandy loam at 30-50 cm. There was an abrupt change in texture at the 50-70 cm depth to a medium clay (Figure 6.3a). The *Tarcoola Depression* and *Tarcoola Vegetated* profiles were a sandy clay loam throughout. The texture at the surface in the *Gunyah* profiles was a sandy loam. At the 50-70 cm depth, there was an abrupt increase in soil texture in the *Gunyah Eroded* and *Gunyah Scald* profiles to a medium clay. In the *Gunyah Pasture* profile the soil texture was a sandy clay loam throughout. In the *Gunyah Vegetated* profile, there was a distinct change to a sandy clay at 50-70 cm, then to a light clay at 70-100 cm.

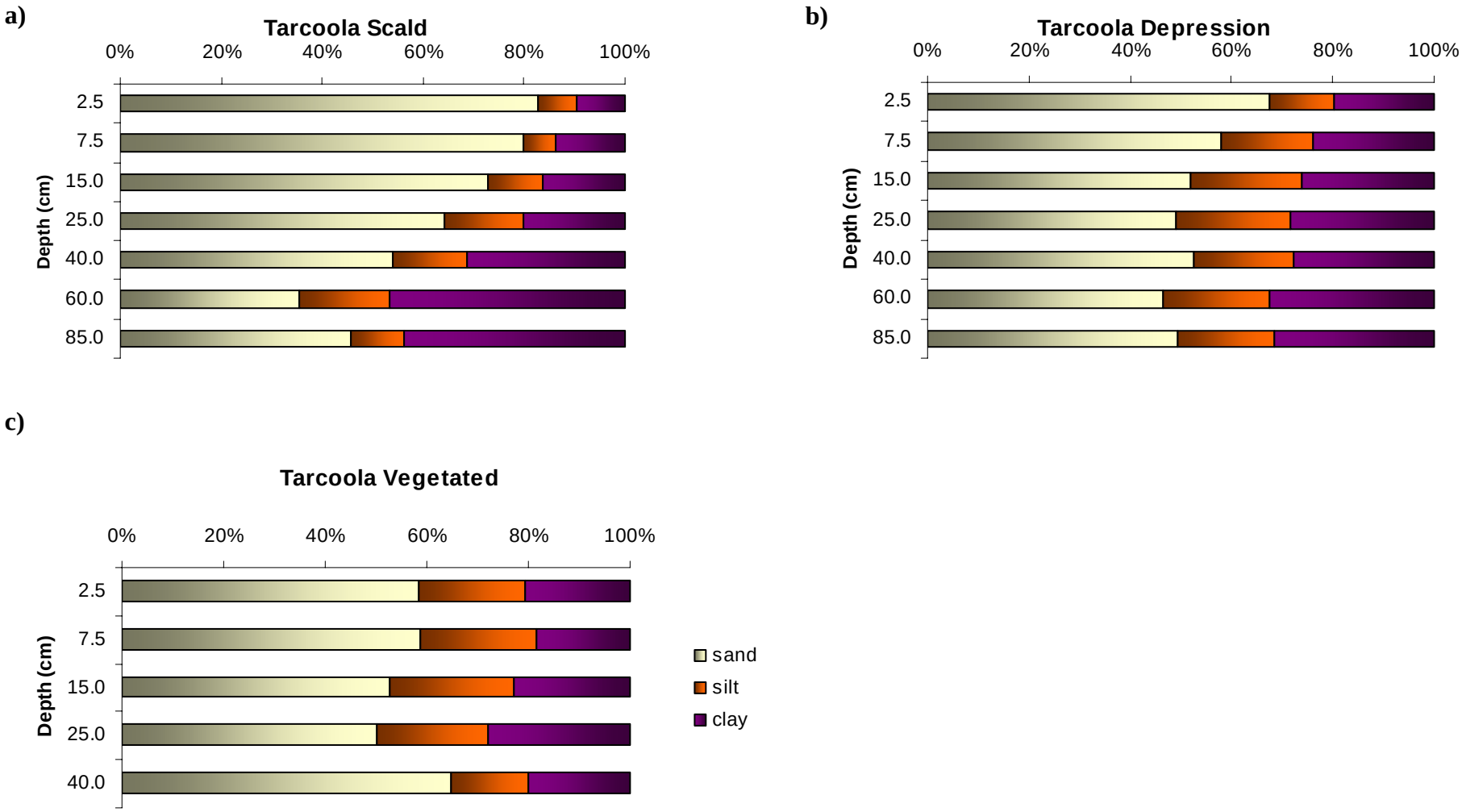


Figure 6.3 **Particle size distribution of the profiles from a) *Tarcoola Scald* b) *Tarcoola Depression* and c) *Tarcoola Vegetated* sites**

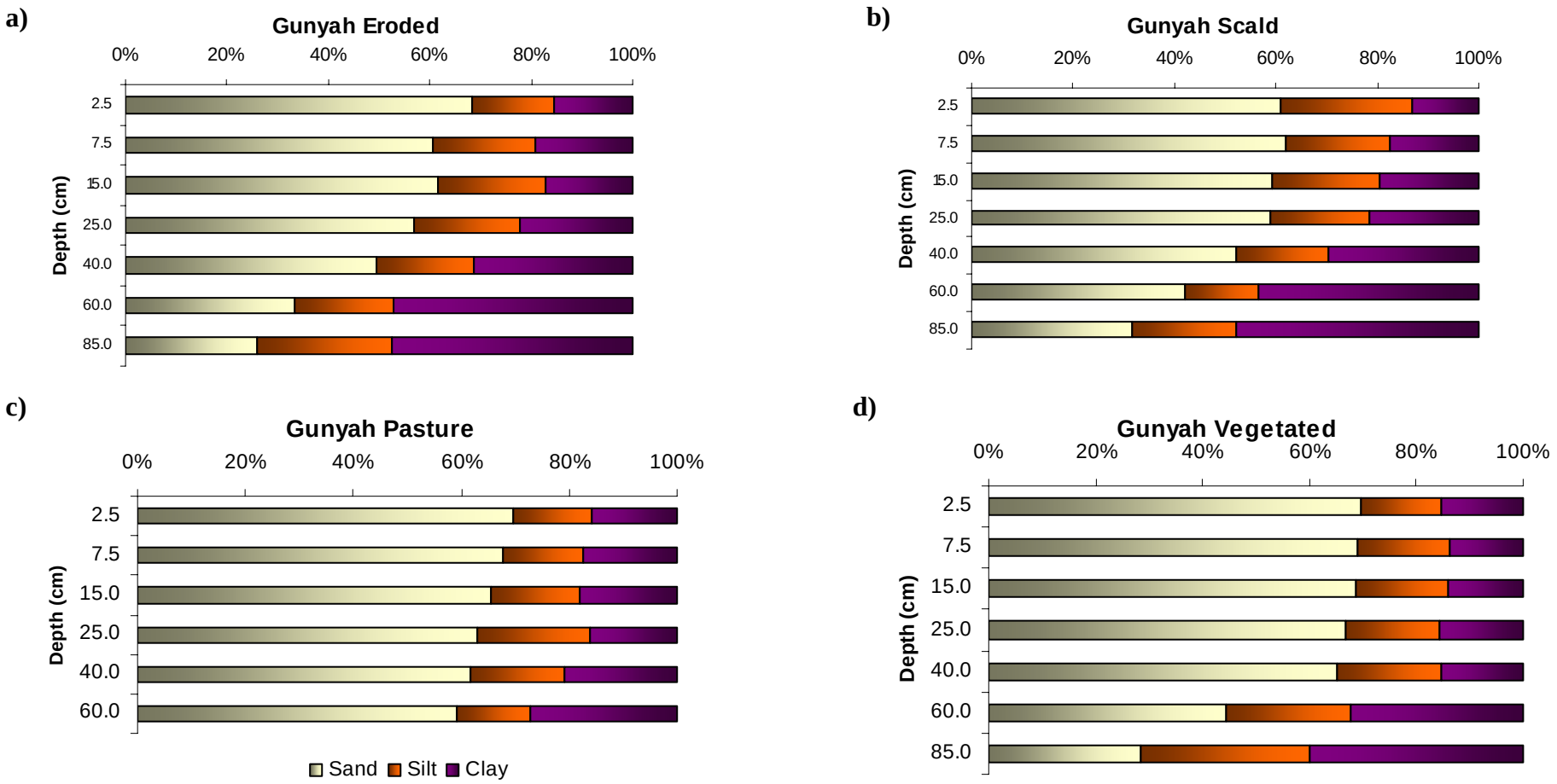


Figure 6.4 Particle size distribution of the profiles from a) *Gunyah Eroded*, b) *Gunyah Scald*, c) *Gunyah Pasture* and d) *Gunyah Vegetated* profiles

6.3.2 Soil pH and EC

Soil pH profiles for both the Tarcoola and Gunyah sites are shown in Figures 6.5a and 6.5b, respectively. There were highly significant differences in pH with microsite and depth ($P<0.001$), and significant interactions between microsite and depth ($P<0.01$), and microsite and site ($P<0.01$). At the Tarcoola site, soil pH was highest at all depths in the *Tarcoola Scald* profile, while the *Tarcoola Depression* and *Tarcoola Vegetated* profiles were both acidic, with pH values significantly less than that of the *Tarcoola Scald* profile at all depths (Figure 6.5a). At the Gunyah site (Figure 6.5b), the *Gunyah Eroded* and *Gunyah Scald* microsites had uniform pH profiles with values near neutral. The *Gunyah Pasture* and *Gunyah Vegetated* microsites were both acidic profiles, with the *Gunyah Pasture* profile showing a large decrease in pH between the surface (0-5 cm) and 20 cm, and hence, had pH values significantly less than the respective *Eroded* and *Scald* profiles. However, below 20 cm, the pH of the *Gunyah Pasture* profile increased.

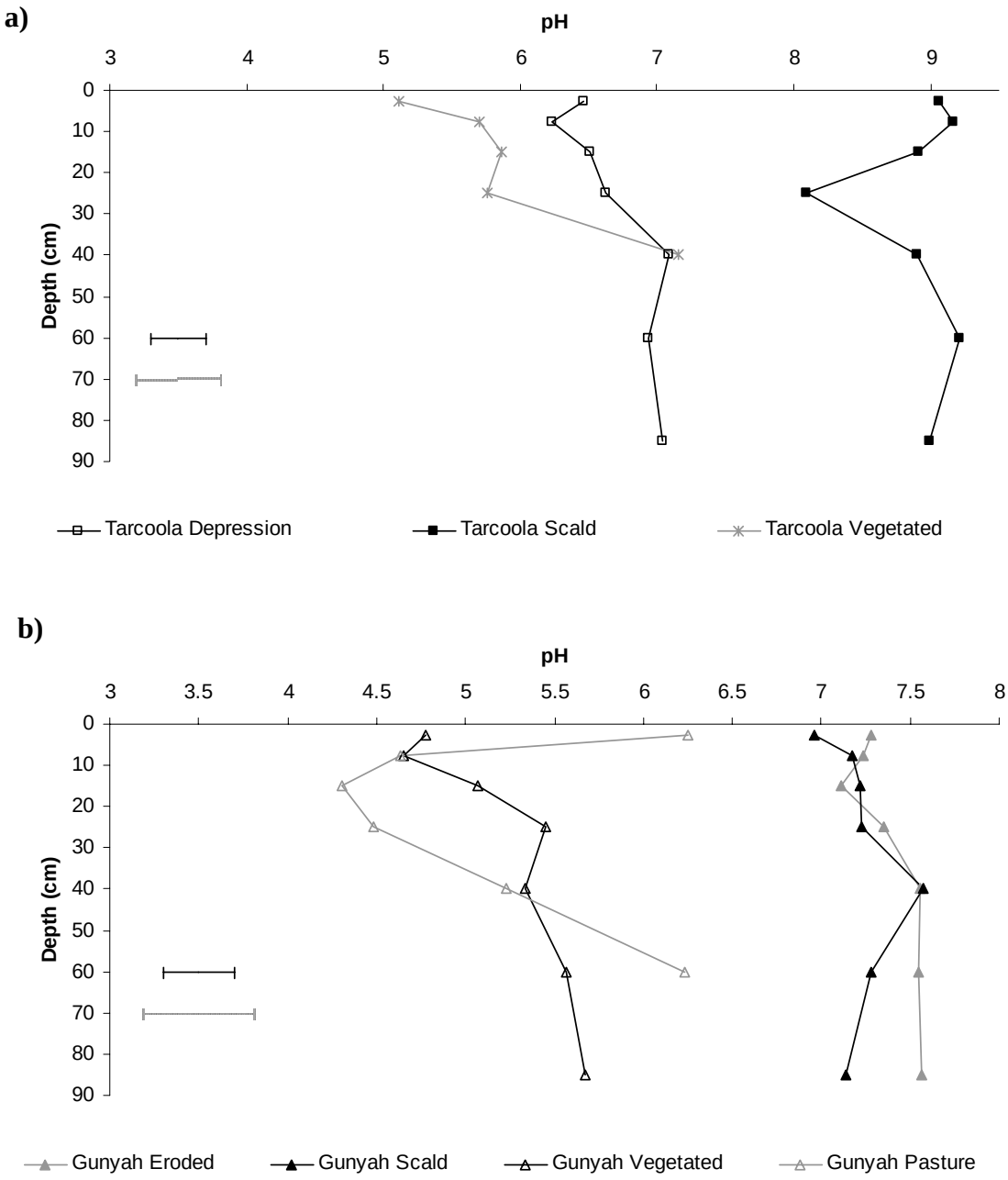


Figure 6.5 Soil $pH_{1:5(H2O)}$ profiles from a) Tarcoola and b) Gunyah
Note: Solid horizontal line indicates LSD for microsite effects; dashed line indicates the LSD for depth effects.

Soil EC profiles for both the Tarcoola and Gunyah sites are shown in Figures 6.6a and 6.6b, respectively. Differences in soil EC were highly significant with microsite (P<0.001), and with depth (P<0.01), with significant interactions between microsite and depth (P<0.05), and microsite and site (P<0.01). In general, the *Tarcoola Depression* and *Tarcoola Vegetated* profiles had lower EC levels compared to the *Tarcoola Scald* profile (Figure 6.6a). Similarly, the *Gunyah Vegetated* and *Gunyah Pasture* profiles had lower EC levels compared to the *Gunyah Scald* or *Gunyah Eroded* profiles. From 40-

100 cm, the *Gunyah Vegetated* profile showed a large increase in EC, which coincides with an increase in soil moisture in the field (refer to soil profile descriptions in Appendix D), indicating soil water not used by the vegetation from the highly saline layer. The *Gunyah Eroded* profile had the highest EC levels of all profiles to a depth of 20 cm, before generally decreasing.

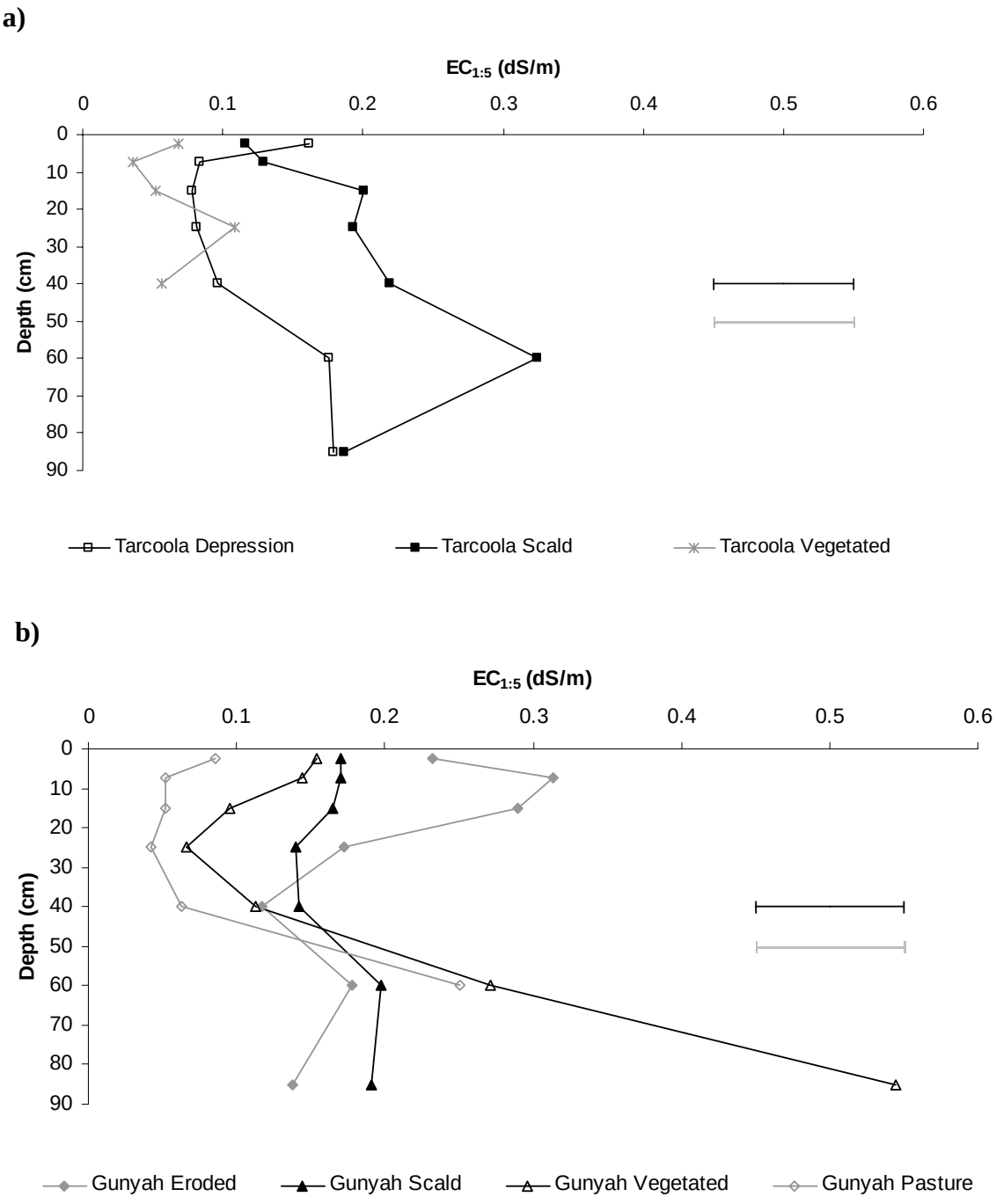


Figure 6.6 Soil $EC_{1:5}$ profiles from a) Tarcoola and b) Gunyah
Note: Solid horizontal line indicates LSD for microsite effects; dashed line indicates the LSD for depth effects.

6.3.3 SAR and ESP

SAR was square-root transformed to satisfy the assumptions of ANOVA, with the back-transformed means presented. Raw means are shown in Table D3 in Appendix D. Concentrations of the soluble and exchangeable cations are shown in Tables D2 and D3 in Appendix D for profiles at both sites, respectively. SAR was highly significantly different with microsite and depth ($P<0.001$), with significant interactions between microsite and site ($P<0.01$), and microsite and depth ($P<0.001$). The *Tarcoola Scald* profile did not show a clear pattern with depth. The *Tarcoola Depression* and the *Tarcoola Vegetated* profiles had lower SAR values than the *Tarcoola Scald* profile. The *Gunyah Eroded* profile had the highest SAR at the surface and generally decreased with depth. The *Gunyah Scald* profile showed the opposite pattern; increasing in SAR with depth to 30 cm, decreasing to 50 cm and increasing again from 50 cm to the bottom of the profile. The *Gunyah Pasture* profile had the lowest SAR at the surface, which increased with depth to values greater that those found in the corresponding *Gunyah Scald* and *Gunyah Eroded* profiles.

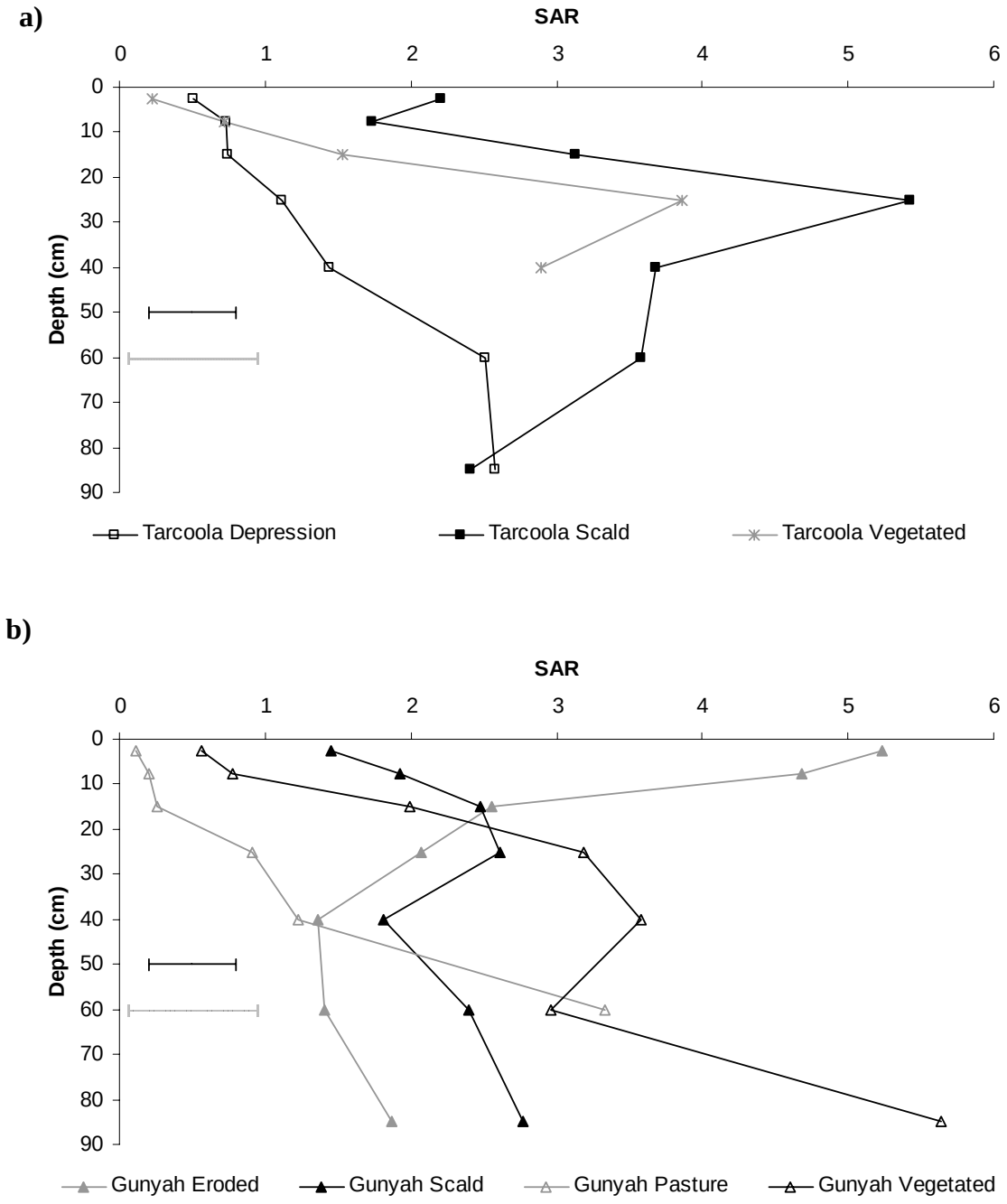


Figure 6.7 SAR Profiles from a) Tarcoola and b) Gunyah; note that data have been square-root transformed

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Solid horizontal line indicates SED for microsite effects; dashed line indicates the SED for depth effects.

The ESP profiles for both the Tarcoola and Gunyah sites are shown in Figure 6.8a and 6.8b, respectively. Differences in ESP were highly significant with microsite (P<0.001), and with depth (P<0.01), with significant interactions between site and microsite (P<0.001) and microsite and depth (P<0.05). The *Tarcoola Scald* profile was highly sodic and had the highest ESP at all depths. The *Tarcoola Depression* and *Tarcoola Vegetated* profiles showed a general increase in ESP to a depth of 30 cm. Below 30 cm, the ESP of the *Tarcoola Vegetated* profile increased sharply with depth, while the *Tarcoola Depression* profile showed a slight increase with depth. The *Gunyah Vegetated* and *Gunyah Pasture* profiles showed a general increase in ESP to a depth of 30 cm, with the *Gunyah Vegetated* and *Gunyah Pasture* profiles reaching a maximum at 20-30 cm and 30-40 cm, respectively. The ESP of the *Gunyah Scald* profile followed a similar pattern to that of the *Gunyah Vegetated* profile to 20 cm, while the ESP was higher in the *Gunyah Eroded* profile compared to the *Gunyah Scald* profile to a depth of 30 cm, and then both profiles displayed a similar pattern with depth.

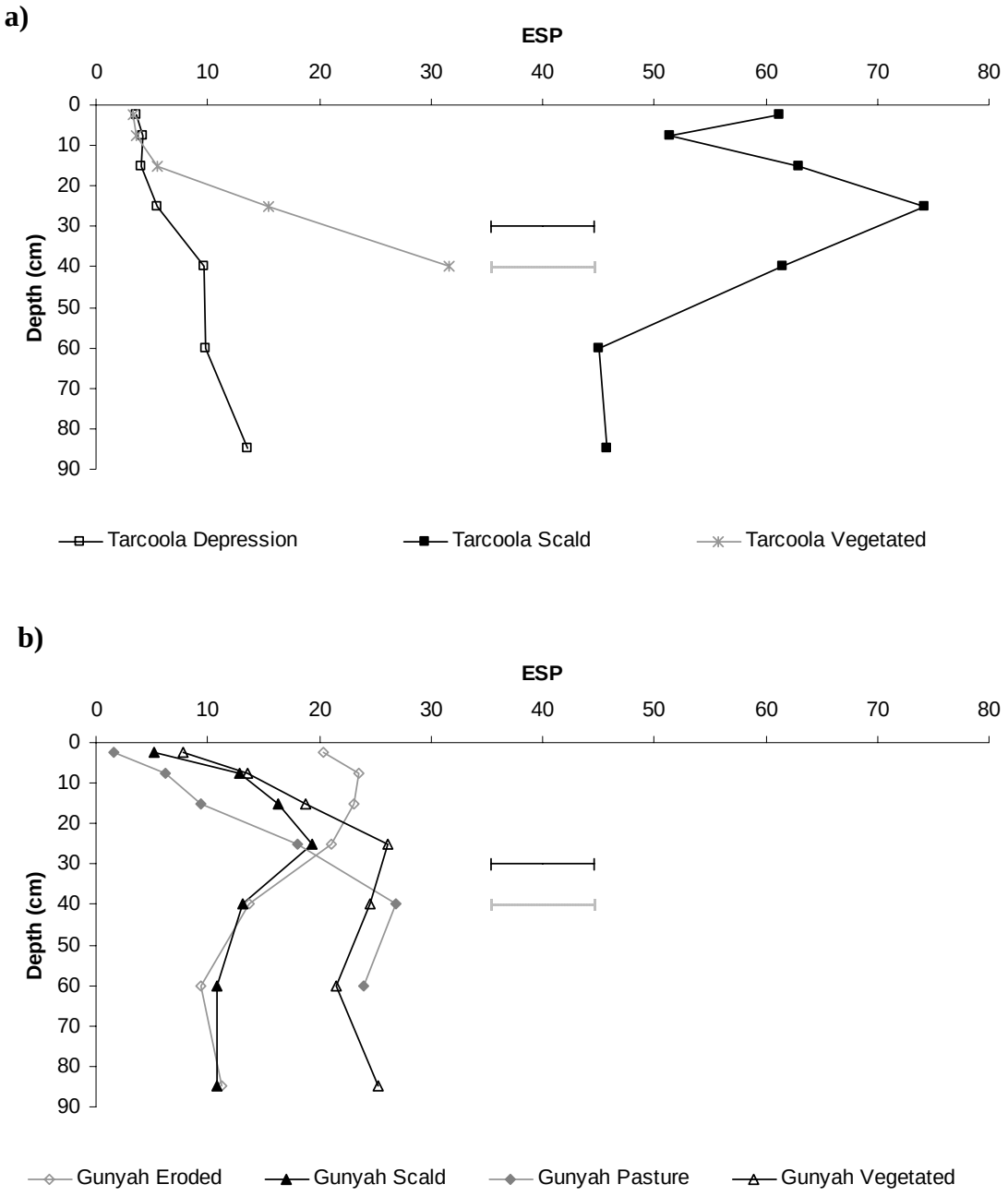


Figure 6.8 ESP profiles from a) Tarcoola and b) Gunyah
Note: Solid horizontal line indicates LSD for microsite effects; dashed line indicates the LSD for depth effects.

6.3.4 Soluble and Exchangeable Ca

Soluble and exchangeable Ca values were square-root transformed to satisfy the assumptions of ANOVA, with the back-transformed means presented in Figures 6.9 and 6.10, respectively. Differences in soluble Ca concentration were highly significant between sites and with depth (P<0.001), with significant interactions occurring between sites, microsites and depth (Figure 6.9). At the *Tarcoola Vegetated* profile, soluble Ca decreased with depth to 30 cm and then increased to 50 cm. The *Tarcoola Scald* profile did not show any pattern with depth. The *Gunyah Pasture* profile had the highest soluble Ca concentration at the surface, which decreased with depth to 30 cm. Similarly, the soluble Ca concentration in the *Gunyah Vegetated* profile also decreased with depth to 30 cm, while the *Gunyah Scald* profile decreased to 10 cm and then showed very little change with depth. The *Gunyah Eroded* profile did not show any pattern with depth.

Differences in exchangeable Ca were highly significant with site and microsite (P<0.001), with highly significant interactions occurring between site, microsite and depth (P<0.001; Figure 6.10). At the Tarcoola site, the *Scald* profile had the lowest exchangeable Ca values at all depths. The *Tarcoola Depression* profile decreased to 50 cm, increased in the 50-70 cm layer, and decreased with depth, while the *Tarcoola Vegetated* and *Tarcoola Scald* profiles had the lowest exchangeable Ca values at the lowest depth sampled from each profile. At the Gunyah site, the *Pasture* profile had the highest exchangeable Ca concentrations at the surface, while the *Eroded* profile had the lowest concentration. Exchangeable Ca concentration displayed a general decrease with depth in the *Gunyah Pasture* profile, while the *Gunyah Eroded* and *Gunyah Scald* profiles displayed similar patterns, decreasing with depth to 30 cm and then increasing to 70 cm. The *Gunyah Vegetated* exchangeable Ca profile decreased to 30 cm, but did not display a clear pattern to 100 cm.

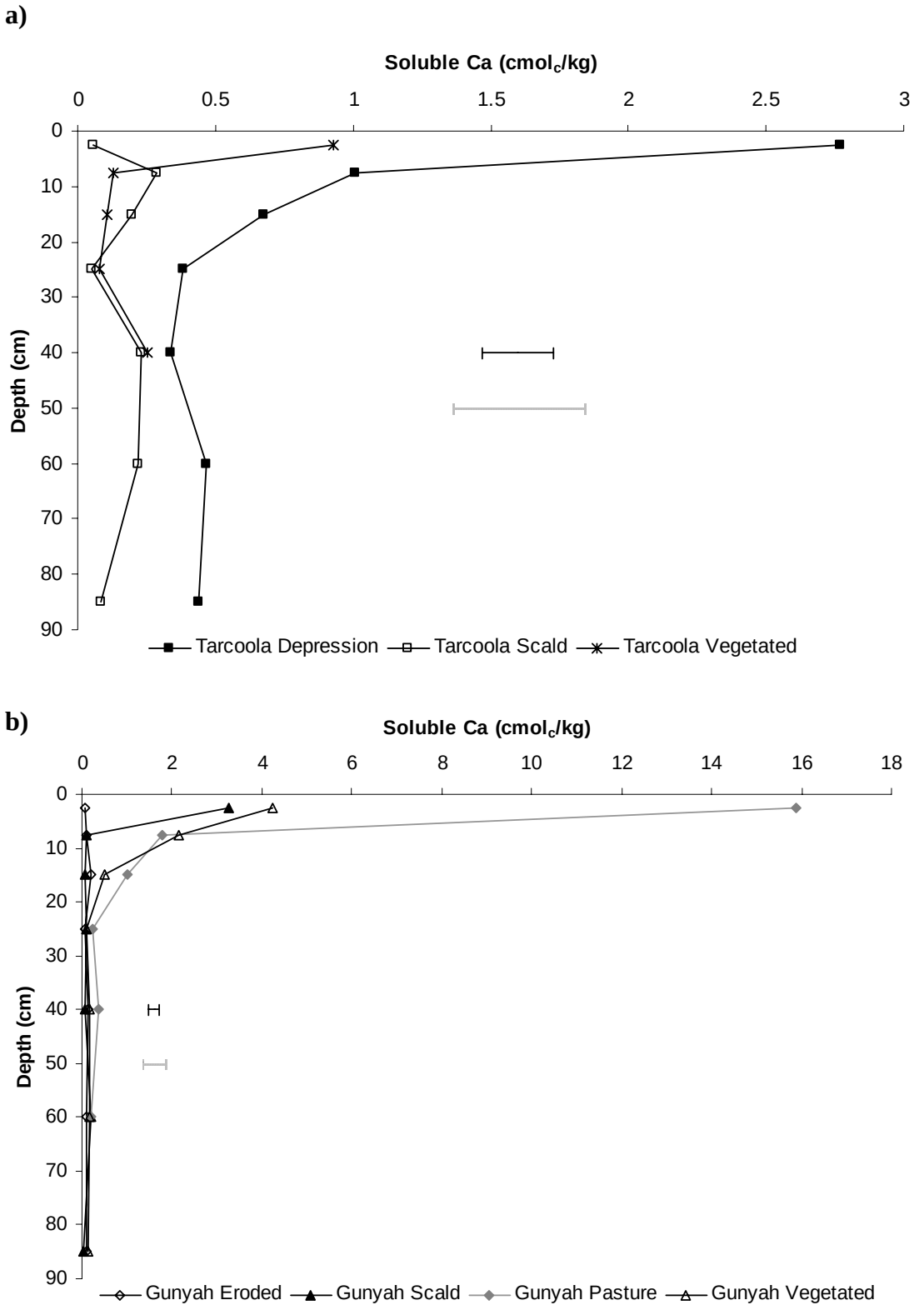


Figure 6.9 Soluble Ca profiles from a) Tarcoola and b) Gunyah

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Solid horizontal line indicates SED for microsite effects; dashed line indicates the LSD for depth effects.

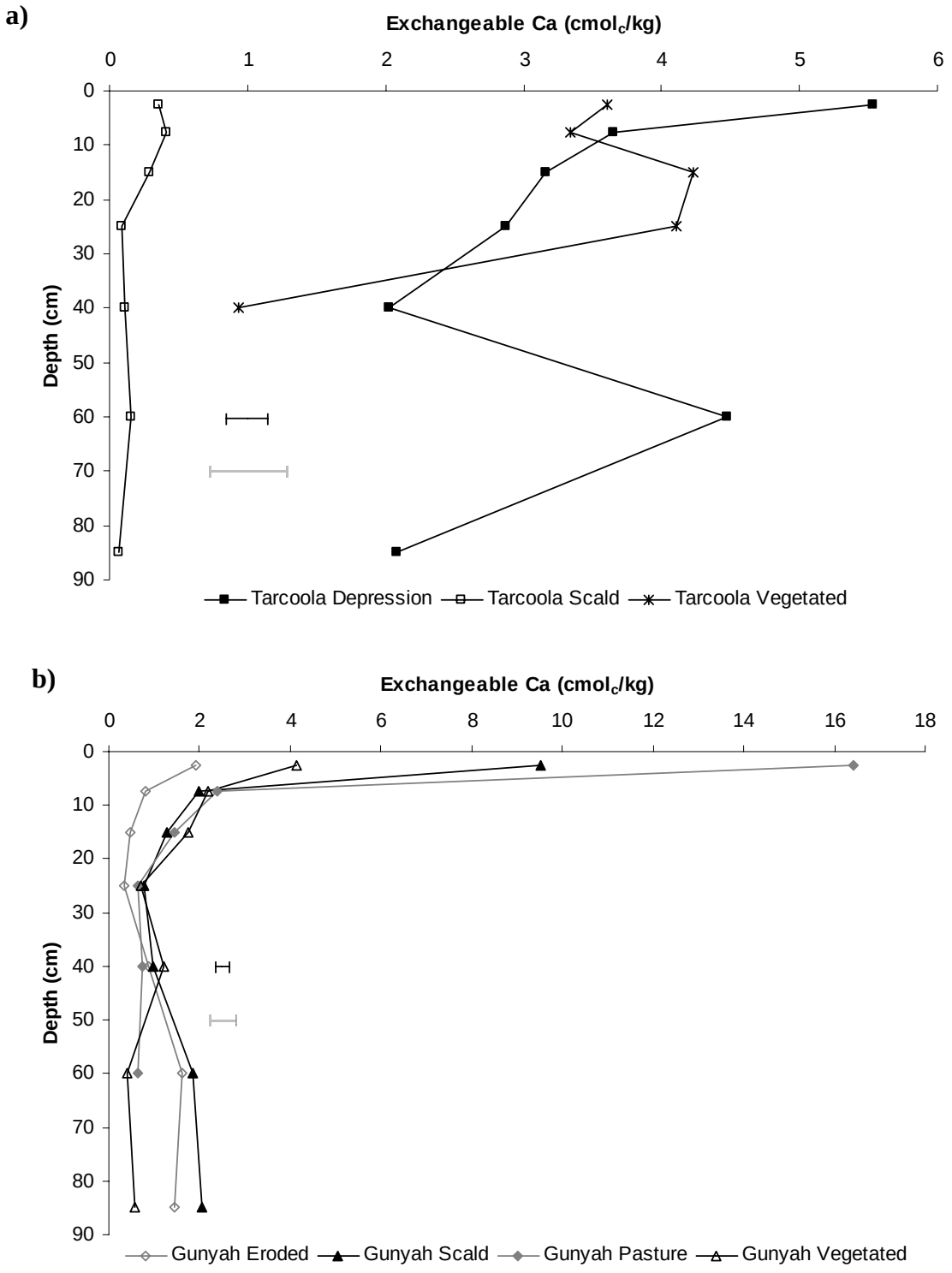


Figure 6.9 Exchangeable Ca profiles from a) Tarcoola and b) Gunyah

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Solid horizontal line indicates SED for microsite effects; dashed line indicates the SED for depth effects.

6.3.5 Soil Organic Carbon and Total Nitrogen

Back-transformed means of SOC for both sites are shown in Figure 6.11, with raw means shown in Table D5 in Appendix D. Differences were highly significant with microsite and depth (P<0.001), with significant interactions occurring between site, microsite and depth (P<0.001). SOC in the *Tarcoola Depression* profile decreased to 50 cm, increased in the 50-70 cm depth, before decreasing again. The *Tarcoola Scald* profile did not appear to follow any patterns, while SOC in the *Tarcoola Vegetated* profile decreased to 10 cm, increased to 30 cm, and decreased to 50 cm. All the *Gunyah* profiles showed a general decrease of SOC with depth. In general, the *Vegetated* and *Pasture* profiles had higher SOC values than the *Eroded* and *Scald* profiles at all depths.

The cumulative SOC stocks profiles for both sites are shown in Figure 6.12. The *Tarcoola Scald* profile had the lowest SOC stocks at all depths compared to the *Tarcoola Depression* and *Tarcoola Vegetated* profiles (Figure 6.12a). There was a notable increase in SOC stocks in the *Tarcoola Depression* profile occurring at the 50-70 cm depth. At the Gunyah site, the *Gunyah Eroded* profile had the lowest SOC stocks. The *Gunyah Vegetated* and *Gunyah Pasture* profiles showed similar levels of SOC stocks to a depth of 30 cm. From 30 cm, the *Gunyah Vegetated* profile had the highest SOC stocks. The increase in SOC stocks with depth was not as apparent in the *Scald* and *Eroded* profiles as it was in the *Vegetated* and *Pasture* profiles at both sites.

SOC stocks were highly significantly different between sites and microsite to a depth of 30 cm *(*P<0.001; Figure 6.13). SOC stocks were significantly higher in the *Tarcoola Depression* and *Tarcoola Vegetated* sites compared to the *Tarcoola Scald* site. Similarly, at the Gunyah site, SOC stocks were significantly higher in the *Vegetated* and *Pasture* microsites compared to the respective *Scald* and *Eroded* microsites, with at least 2.5 times more C in the *Vegetated* profiles. There was significantly more SOC in the *Gunyah Scald* profile compared to the *Gunyah Eroded* profile, while the differences in SOC between the *Gunyah Vegetated* and the *Gunyah Pasture* profiles were not significantly different.

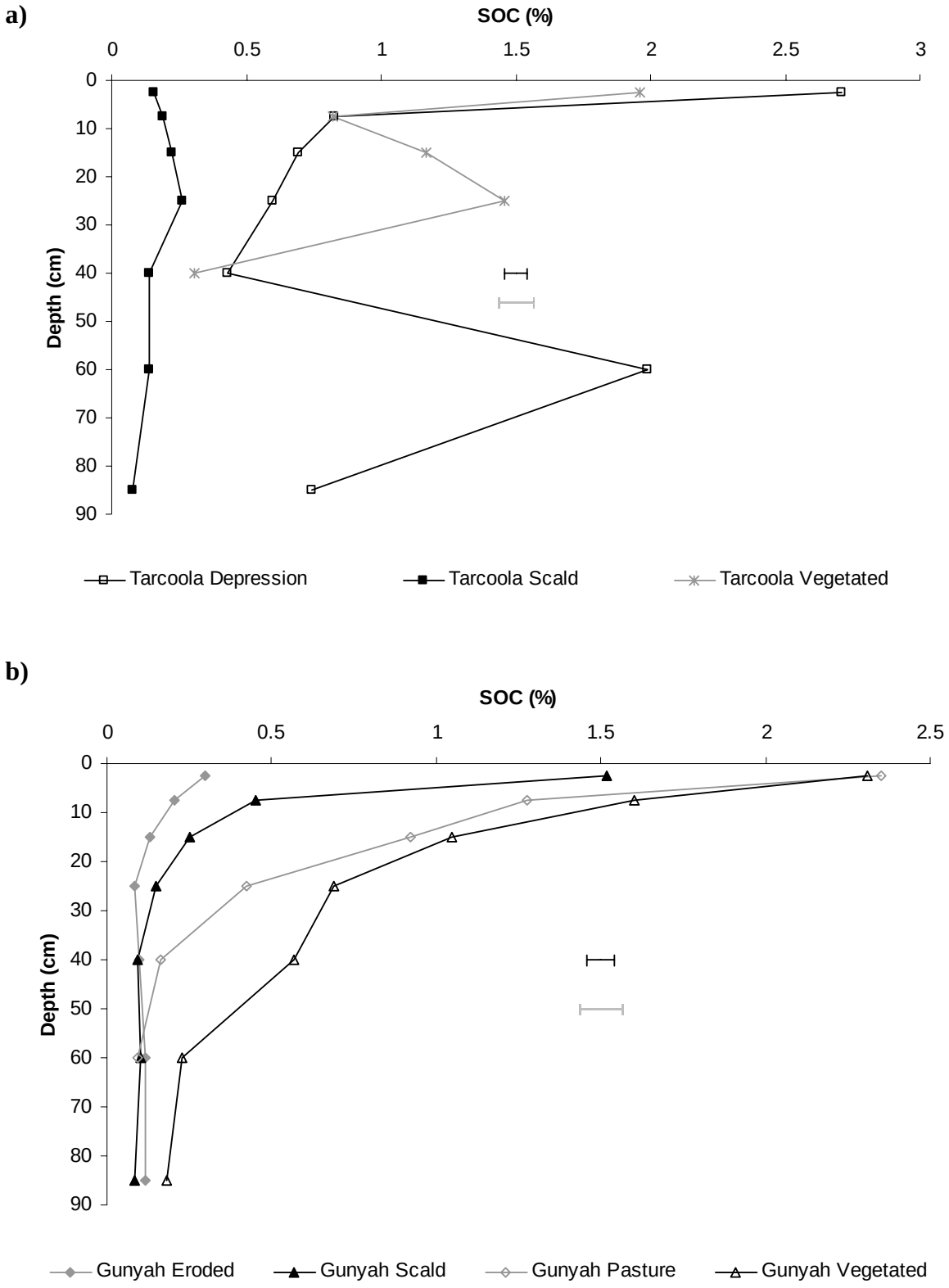


Figure 6.11 SOC profiles from a) Tarcoola and b) Gunyah

Note: Data were square-root transformed for statistical analysis with back-transformed means presented. Solid horizontal line indicates SED for microsite effects; dashed line indicates the SED for depth effects.

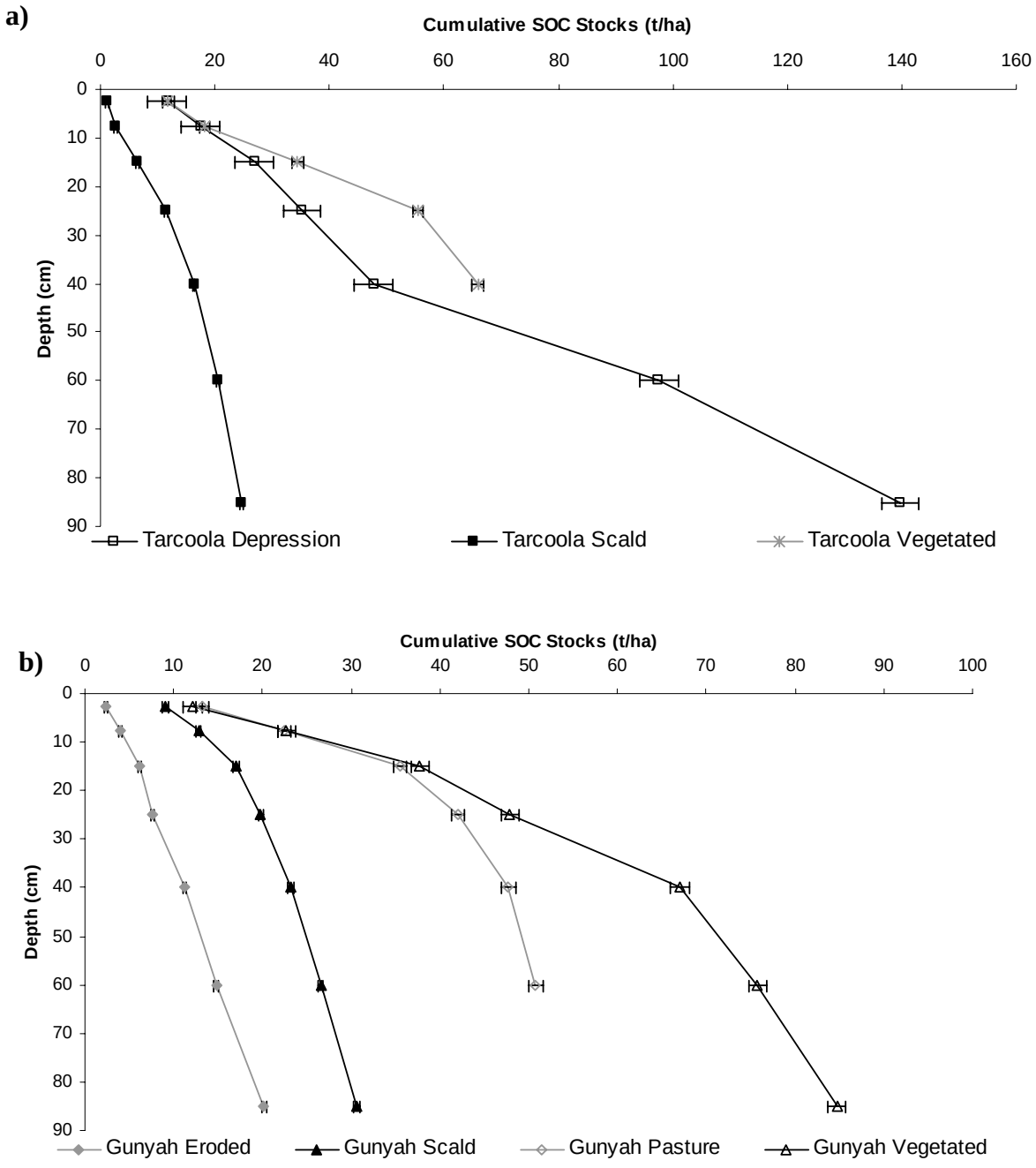


Figure 6.12 Cumulative SOC stocks with depth at a) Tarcoola and b) Gunyah
Note: Error bars indicate the standard error of the mean.

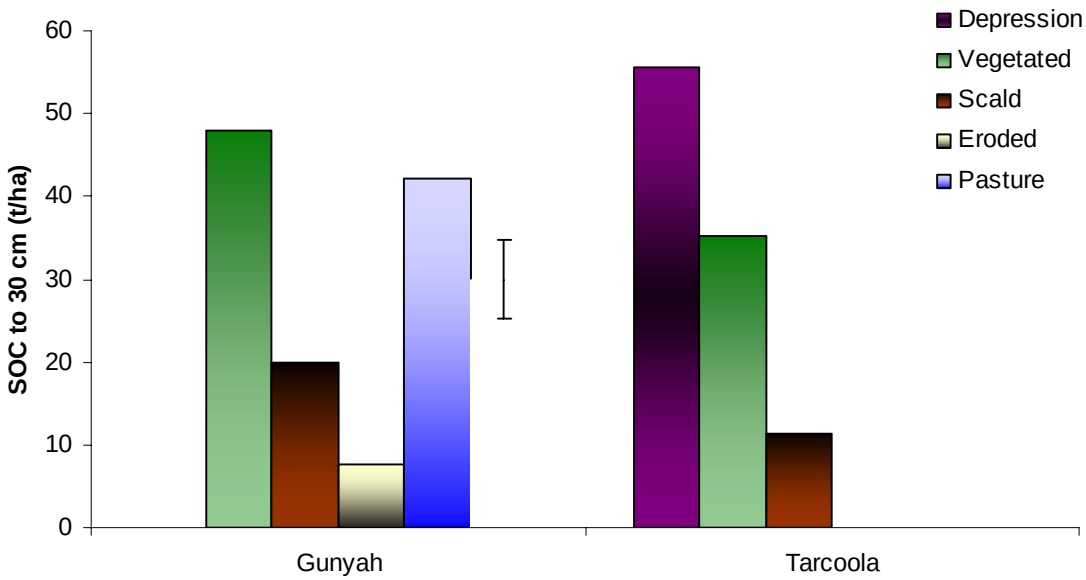
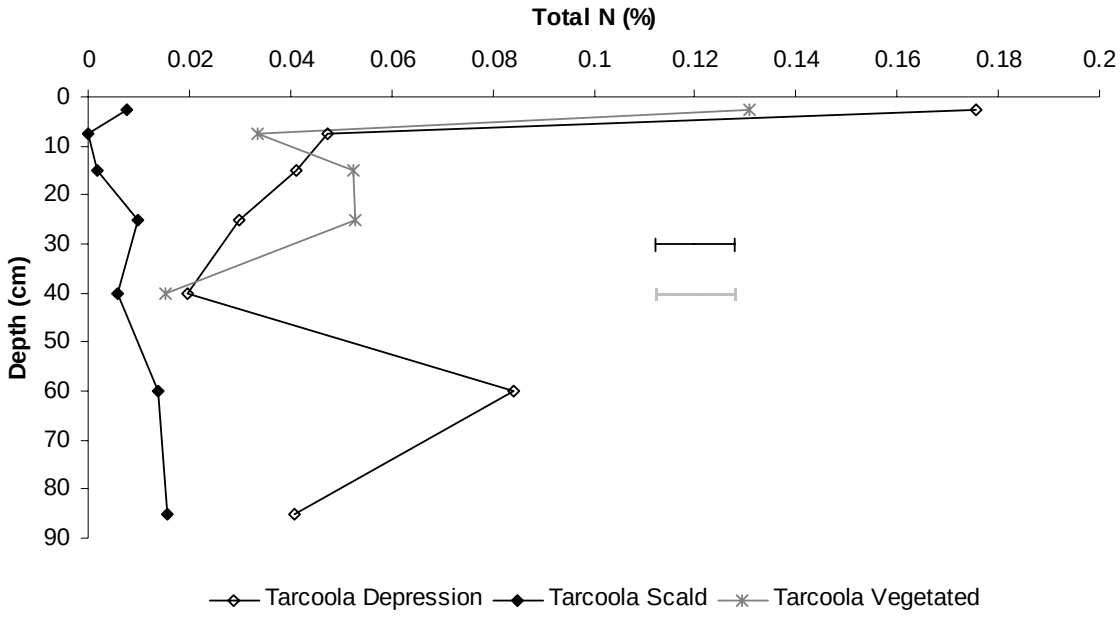


Figure 6.13 SOC stocks to a depth of 30 cm from each site and microsite
Note: Vertical bar indicates the LSD.

Total N profiles at both sites are shown in Figure 6.14. Total N followed similar patterns to that of the SOC profiles. Nitrogen was highly significantly different with microsite and depth (P<0.001), and showed significant interactions between microsite, site and depth (P<0.001).

a)



b)

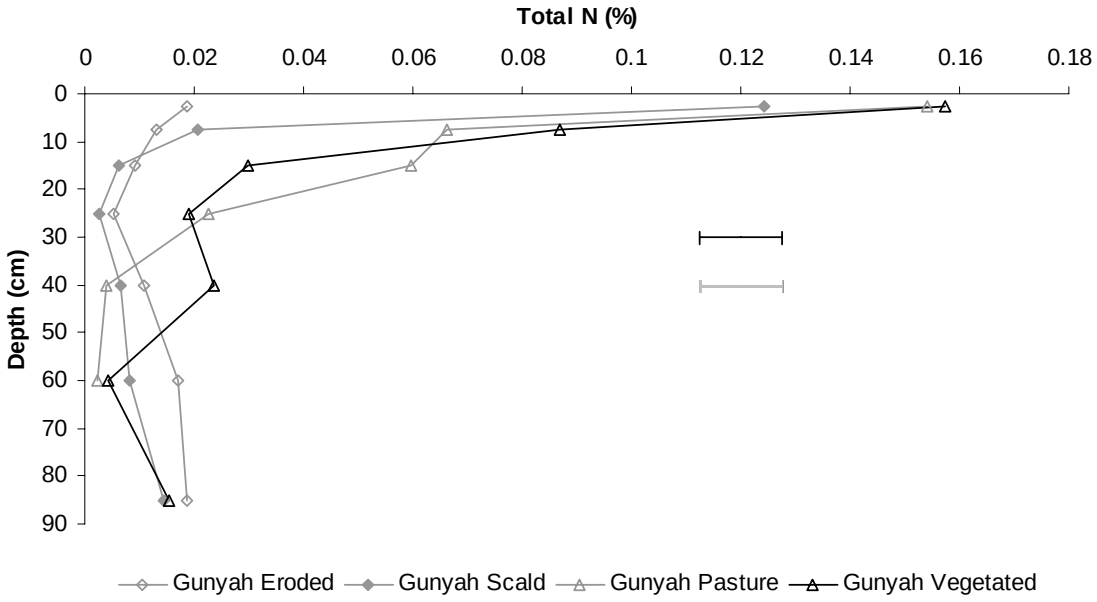


Figure 6.14 Total N profiles from a) Tarcoola and b) Gunyah
Note: Solid horizontal line indicates LSD for microsite effects; dashed line indicates the LSD for depth effects.

6.3.6 Correlations Between Soil Properties

The correlation matrix between the soil properties is shown in Table 6.2. Fitting pH, EC, ESP, SAR and S after site factors did not significantly affect SOC. However, N was strongly correlated with SOC (ie. 0.935; $P<0.001$), as shown in the correlation matrix (Table 6.2).

Table 6.2 Correlation matrix of soil properties

pH	1.000						
EC	0.276	1.000					
SAR	0.259	0.511	1.000				
ESP	0.597	0.275	0.455	1.000			
SOC	-0.492	-0.269	-0.335	-0.462	1.000		
N	-0.395	-0.195	-0.366	-0.440	**0.935**	1.000	
Bulk Density	0.386	0.191	0.315	0.454	-0.793	-0.790	1.000
	pH	**EC**	**SAR**	**ESP**	**SOC**	**N**	**Bulk Density**

6.4 Discussion

6.4.1 Soil Properties: Bulk Density, pH, EC, ESP and SAR

These results demonstrate that the presence of existing vegetation aids in maintaining a number of important soil properties, while the revegetation of formerly degraded sites can aid in the mediation of adverse soil conditions, such as those commonly found in saline and sodic landscapes. Processes associated with vegetation growth can improve soil fertility in general, in addition to a number of soil properties. This occurs in alkaline soils by increasing the partial pressure of CO_2 in the soil environment, as discussed in Section 2.2.4, and through increased inputs of litter, which further promotes vegetation growth in a process of positive feedback.

At both sites, the presence of a scalded or eroded profile resulted in high bulk density values relative to the vegetated profiles. The lower bulk density at the surface in the *Gunyah Scald* profile (1.2 Mg/m^3) compared to the *Gunyah Eroded* profile (1.6 Mg/m^3) is probably due to unconsolidated material present at the surface of the scald due to deposition processes. High bulk density values were apparent in the *Gunyah Eroded* and *Tarcoola Scald* (1.50 Mg/m^3) profiles due to the exposure of the subsoil, which results

in higher bulk density values at the surface, as bulk density tends to be higher in subsoils than surface soils.

Crusting and hardsetting commonly occur in sodic soils (Levy *et al.* 1998), which can increase bulk density and impact upon root growth by reducing root penetration. The lack of plant growth on the scalded areas can also result in a lower number of pores due to a decrease in root channels, both horizontally and vertically. With plant growth, bulk density in the surface layers decreases due to the presence of root channels (Bruand and Gilkes 2002) and an increase in the build-up of organic material on the soil surface which is unconsolidated, as noted in the *Vegetated* and *Pasture* profiles. The input of organic matter from vegetation improves aggregation, resulting in an improvement in soil structure, and hence, an increase in pore space. At depth, however, there are few roots, and hence, very little improvement in soil structure. This is also reflected in the bulk density values at depth, which increase with increasing clay content. No such pattern in bulk density occurred in the *Scald* and *Eroded* profiles due to the dominance of sodic processes throughout the entire profile.

Soil pH at the Tarcoola site showed similar values to those previously described, with high pH in the scalded areas, in Chapters 4 and 5, and near neutral pH in the vegetated areas, in Chapter 3. Similarly, pH decreased with the presence of native vegetation and planted pasture in the Gunyah profiles. The SAR and ESP were also generally higher in the *Scald* and *Eroded* profiles compared to the *Vegetated* and *Pasture* profiles. Interestingly, the ESP in the *Gunyah Scald* profile is lower than that found in the *Gunyah Vegetated* profile at the surface, and is not comparable to the *Gunyah Eroded* profiles until the 20-30 cm layer. The *Gunyah Eroded* profile is estimated to have lost the top 5 cm of its original profile; however, the ESP of the 0-5 cm layer is still higher than the ESP of the 5-10 cm layer from the *Gunyah Scald* profile. It may be that the loss of the top 5 cm of the *Gunyah Eroded* profile has decreased the soil's buffering capacity against degradation processes due to increased losses of SOM associated with the top layer of soil, as described in more detail in Section 6.4.2.

Vegetation has been noted to reduce pH and ESP in sodic soils in previous studies (eg. Garg 1998; Mishra and Sharma 2003). Those processes described in the laboratory experiment in Section 5.4 are likely to also occur in the field, whereby production of CO_2 from decomposition of organic material results in a decrease in soil pH due to an

increase in P_{CO2}. While decomposition processes produce CO_2, as described in Section 5.4.1, the growth of plants also results in CO_2 production from root respiration, increasing P_{CO2} in the root-zone (Figure 2.4; Qadir *et al.* 2003). Concurrently, protons are excreted from plant roots in the form of organic acids, while mineralisation of organic N, P and S also produces acidity, which contributes to lowering of pH in vegetated areas (Nelson and Oades 1998), indicated in the *Tarcoola Depression* and *Tarcoola Vegetated* microsites.

At both sites, EC was generally lower in vegetated profiles. At Tarcoola, the EC was consistently lower in the *Tarcoola Vegetated* profile than in the *Tarcoola Scald* profile, except in the 0-5 cm layer. Similarly, at Gunyah, the EC values of the *Vegetated* and *Pasture* profiles were lower than those of the *Eroded* and *Scald* profiles, notably in the upper parts of the profiles where roots are likely to be concentrated. It could be argued that either the presence of vegetation decreases soil EC by enhancing the leaching of salts, or that vegetation growth occurs as a result of the lower salt concentrations already present. However, evidence indicates that the vegetation was present prior to the development of salinity, as described in more detail in Section 6.4.3, with subsequent outbreaks of salinity resulting in the death of vegetation and the establishment of the scalds. Increasing EC with depth under the *Vegetated* and *Pasture* profiles is most likely due to salt exclusion by the plants present and its subsequent translocation down the profile by leaching, which is enhanced by improved soil properties under vegetation. Similarly, soluble Na^+ may also be excluded by plants and also translocated down the profile, as evidenced in the general increase in SAR with depth in the *Vegetated* and *Pasture* profiles. The high EC values in the *Gunyah Vegetated* and *Gunyah Pasture* profiles below 40 cm support the occurrence of leaching.

However, it is unlikely that the scalds at either Gunyah or Tarcoola have developed as a result of salinity alone, as the EC profiles indicate that it is not considered to be of high enough salinity for plant growth to cease (EC(1:5) < 1.5 dS/m; Murphy and Eldridge 1998).. The scalds at Gunyah and Tarcoola are most likely the result of both salinity and sodicity, with the alkaline nature of Tarcoola also playing a role in the lack of plant growth and hence, increasing dispersion and erosion of top soil. However, it is likely that the EC will be high enough in certain microsites at times of moisture stress to negatively impact on plant growth.

6.4.2 Soil Organic Carbon and Total Nitrogen

At both Tarcoola and Gunyah, SOC was higher in the *Vegetated* and *Pasture* profiles compared to the corresponding *Scald* profile in the surface 30 cm; SOC was also higher in the *Gunyah Scald* profile compared to the *Gunyah Eroded* profile in the same depth interval. Vegetation is a major determinant in the relative distribution of SOC as a result of patterns of C input (Jobaggy and Jackson 2000). Hence, if little or no vegetation occurs on the surface, as is the case in the scalded soils, then very little C input is occurring, as reflected in low SOC concentrations.

The large increase in SOC in the 50-70 cm layer of the *Tarcoola Depression* profile coincides with an increase in soil moisture and clay content. It is likely that this layer at represents a buried soil surface layer, which is supported by a change in soil colour from dull yellowish brown in the layer above to brownish grey (see profile description in Appendix D). Where soils are waterlogged, which commonly occurs in saline and sodic landscapes, decomposition processes are slowed. Clay content also increases with depth in duplex soils, commonly found in the Southern Tablelands region (Murphy and Eldridge 1998), which is also linked to higher SOC contents (Bird *et al.* 2001). Similarly, poor drainage conditions and high clay contents favoured C sequestration in the upper 20 cm of the soil profile in a range of land use types in Ohio in the United States of America (Tan *et al.* 2004). Because SOC is generally highest at the surface, a buried surface layer of soil will also exhibit very high concentrations of SOC due to limited decomposition at depth, particularly where conditions are anoxic or sub-oxic. Similarly, Fang *et al*. (2006) found SOC to increase to levels greater than that found in the topsoil, where an original surface soil layer had been buried at depth. Leaching processes may also translocate DOC to lower layers which accumulate where there is an increase in soil texture, and may also lead to a build up in SOC stocks at depth. These processes are likely to affect the SOC concentration at depth of the *Tarcoola Depression* profile.

Effects due to land use, and hence land management, on SOC are usually only observed in the topsoil, or surface layers, with SOC profiles usually approaching similar values at depth (Jinbo *et al.* 2006). However, in this study, differences were apparent even in the 70-100 cm layer, with concentrations of SOC significantly higher in the *Gunyah Vegetated* and *Tarcoola Depression* and *Vegetated* profiles compared to their respective *Scald* and *Eroded* profiles. While the *Gunyah Pasture* site had significantly higher SOC

levels than the *Gunyah Eroded* and *Gunyah Scald* profiles to 30 cm, the concentration of SOC was similar at depth (Figure 6.11b). These patterns indicate that where SOC is lost as a result of scalding, these losses continue to occur throughout the soil profile, including at depth. Through revegetation with introduced pasture, it is possible to restore SOC concentrations to levels similar to that of native pasture. However, in this study, these effects are only evident at one site (Gunyah) in the top 30 cm, and did not occur at depth over the 10 years since revegetation.

The loss of SOC in the *Gunyah Eroded* profile, particularly the top layers, is clearly evident in both concentrations of SOC (Figure 6.11) and SOC stocks (Figure 6.13). The loss of SOC highlights the importance of preserving the upper layers of soil and the potential for SOC loss, with over 10 t/ha less SOC in the top 30 cm of the *Gunyah Eroded* profile compared to the *Gunyah Scald* profile. Loss of topsoil also results in a decrease in soil fertility and resilience, and hence, increases its susceptibility to further erosion (Mabuhay *et al.* 2006). As erosion increases, loss of SOC also increases since SOM is concentrated near the soil surface, as the SOM at the soil surface is of relatively low density and contains the most labile fractions (Lal 2001). The loss of SOM further decreases the soil's buffering capacity against degradation processes such as high alkalinity, sodicity or salinity, as SOM contributes a significant proportion of a soil's CEC (Nelson and Oades 1998) and nutrients. Further losses of SOC can occur as the loss of the upper layers exposes subsoil layers, resulting in increased accessibility of SOM in the lower layers for decomposition.

Whilst the presence of vegetation can mediate adverse soil conditions, the difficulty lies in establishing and maintaining vegetative production on salt-affected sites over time. It is likely that the successful revegetation strategy evident at Gunyah is due to the soils being of moderate salinity, neutral pH with adequate Ca concentrations. The high levels of soluble and exchangeable Ca found in the *Gunyah Pasture* site compared to the *Tarcoola Scald* site probably played a role in the re-establishment of vegetation at Gunyah, as Ca can aid in mediating against the toxic offects of Na (Reid and Smith 2000). Similarly, the neutral pH values found in the *Gunyah Scald* and *Gunyah Eroded* sites compared to the high pH conditions of the *Tarcoola Scald* site probably also played a role in the successful establishment of vegetation at Gunyah, as soil pH was already within the limits for plant growth, with no further remediation required. Previous revegetation strategies at Tarcoola have failed, most likely due to the high

alkalinity and ESP evident in the *Tarcoola Scald* profile which occurred at levels which prevented seedling establishment and plant growth (B. Murphy pers. comm). As vegetation health declines in such environments, there is the potential for a positive feedback system to establish, whereby alkalinity and sodicity increase as root respiration decreases. Vegetation growth slows, then ceases over time as alkalinity and sodicity increase to levels above plant tolerance limits. Unfortunately, in saline and sodic landscapes, a simple solution for rehabilitation, which was found at Gunyah, usually does not exist, with revegetation strategies generally site specific. It is likely that multiple remediation strategies, which include deep-ripping and addition of soil ameliorants, will need to be employed in hostile soil environments, such as that found at Tarcoola where plant establishment is difficult. It is also possible in extreme cases that vegetation will only re-establish at a great financial and labour cost. Therefore, the best solution in such cases may be to fence scalded areas to remove from production to prevent further degradation.

SOC stocks in the top 30 cm were increased to a level comparable to that under native vegetation (*Gunyah Vegetated*; Figure 6.12) following revegetation with pasture (*Gunyah Pasture*), with no significant differences found between the *Gunyah Vegetated* and *Gunyah Pasture* profiles. It has been noted in a meta-analysis by Conant *et al*. (2001) that an improvement in land management practices, such as the revegetation practices used in this study, can increase SOC stocks, with these net increases in SOC persisting for at least 40 years. Similarly, Young *et al*. (2005) have shown that after a period of 15 years or more, SOC concentrations in the upper soil layers under perennial pasture were approaching equilibrium conditions characteristic of a perennial system such as a grassy woodland. As pasture age increases, more SOC is physically protected in microaggregates due to continual development of the root system (Conant *et al.* 2004). As SOC increases, so too does the stability of the C pool (Rutigliano *et al.* 2004), as SOM can become increasingly protected as soil structure improves. Many macroaggregates form around new root derived POM, such as sloughed-off root material, during periods of vegetative growth and senescence (Gale *et al.* 2000). Mucilages are also produced *in situ* by roots which aid in aggregation and physically protect C (Oades 1984). After the death of plants, macroaggregates continue to form around new root-derived POM. As the roots decompose, microbial binding agents are produced resulting in an increase in macroaggregate stability and the formation of microaggregates over time. Concurrently, microbial products and SMB are adsorbed to

mineral particles, aiding in the maintenance of stable soil structure (Golchin *et al.* 1994) and the physical protection of SOM.

It has been suggested that SOC concentrations and SOC stocks near the surface can be poor predictors of the amounts of C at depth, particularly as land-use, topography and vegetation type all influence C distribution down the profile (Young *et al.* 2005). In this study, SOC stocks were calculated to a depth of 30 cm because it is an internationally recommended practice in C accounting to express C stocks to a depth of 30 cm (IPCC 1997). However, SOC stocks in areas where groundwater tables are high may be underestimated. Where watertables are high or waterlogging occurs, decomposition may be slowed resulting in an accumulation of SOC in the wetter parts of the soil profile, as described in Section 2.2.3. SOC stocks may also be underestimated where SOC stocks are assessed to a depth of 30 cm as a result of increasing bulk density and clay content with depths, where there are likely to be significant stores of SOC. Similarly, in areas where buried soil horizons are found, which is not uncommon in salt-affected landscapes, SOC can also be underestimated when assessed to a depth of 30 cm, as seen in the *Tarcoola Depression* site. Similarly, SOC can accumulate in areas of deposition, or display a sharp increase at depth due to the burial of SOC from continued deposition of eroded material (Fang *et al.* 2006; VandenBygaart 2001). It is likely that this process occurred in the *Tarcoola Depression* profile, which was located in a drainage depression position, with approximately 60 cm of material deposited over the original soil surface, which was darker in colour and contained a higher content of SOC compared to the layer above of recently deposited materials.

Soil C profiles can provide information on the pedological history and soil formation in the landscape. Under equilibrium soil conditions, SOC profiles generally follow a depth function if uninterrupted by geomorphological or pedological events, decreasing with depth due to root density distribution and adsorption processes in mineral horizons. However, where degradation has previously occurred, SOC can show a smaller decrease with depth in profiles, and is less likely to be retained at depth compared to non-degraded profiles due to lower initial SOC levels at the surface (Kalbitz 2001). This was also reflected in this study, with lower SOC concentrations evident in the topsoil of the *Gunyah Scalded* and *Gunyah Eroded* profiles. This trend continued with depth, compared to the *Vegetated* and *Pasture* profiles. Similarly, the SOC content in the *Tarcoola Scald* profile did not display a decrease with depth, which is may be indicative

of previously high losses of SOC at the surface. Where extensive redistribution of soil occurs through processes such as erosion, SOC profiles can be expected to differ significantly.

Nitrogen concentrations followed a similar pattern to that of SOC, exhibiting lower levels in the *Scald* and *Eroded* profiles and higher levels in the *Vegetated* and *Pasture* profiles. N concentrations were also were highly correlated with SOC concentrations. Soil C and N cycles are intimately linked, and hence, generally follow similar patterns (Breuer *et al.* 2006), being strongly tied to SOM input. N is frequently limiting for growth in disturbed or degraded soils (Ross *et al.* 1982), with N fertilisation shown to increase SOC concentrations and decrease SAR in a Solonetzic soil in Canada, despite a decrease in pH (McAndrew and Malhi 1992). Hence, while the lower N concentrations in the *Scald* and *Eroded* profiles are the result of a limited SOM input, the re-establishment of vegetation in such areas is most likely limited by low N concentrations. Therefore, any future rehabilitation efforts will also need to consider inputs of N.

6.4.3 Historical Salinity Issues in the Region

A previous study by Wagner (2001) indicated that saline areas in the Lachlan and Murrumbidgee region catchments have increased dramatically since settlement. Prior to the onset of salinity, scalded areas and non-scalded areas were equally vegetated, ascertained with the use of aerial photography. From the historical aerial photographs, it was seen that scalded areas in the Bevendale region have been expanding since the 1940s, and have become subject to extensive sheet and gully erosion. In the Rugby region, saline areas developed in the 1960s and have been expanding since that period. On some farming properties in the area, structural works have been constructed in conjunction with tree planting and sowing of salt-tolerant grasses in an effort to rehabilitate the area. This was evident in a separate paddock at the *Tarcoola Vegetated* site, with revegetation with *Pinus radiata* having been undertaken approximately 15 years ago with limited success (M. Rankin pers comm.).

As a result of increasing salinity, losses of SOC have been occurring over a period of over 60 years due to a number of processes. In this time frame, losses due to erosion are likely to become more apparent, which is reflected in the lower SOC stocks of the *Scald*

and *Eroded* profiles. As erosion continues, the exposure of subsoil layers of the profile further enhances SOC loss, as described above. In vegetated areas of similar soil types, there can be up to 2.5 times more SOC stocks compared to those profiles which had been scalded, as shown in Figure 6.13. However, it has been noted that management improvements and land use conversions which increase forage production, and hence vegetation, will generally increase SOC (Conant *et al.* 2001). This is evident in the SOC stocks found in the *Gunyah Pasture* profile following revegetation, on a formerly scalded area, which were comparable to the SOC stocks found in the *Gunyah Vegetated* profile, with the apparent ease of revegetation largely attributed to the neutral pH values of the site and exclusion of stock.

6.4.4 Area Affected by Salinity

It has been estimated that 7330 ha of land was affected by salinity in 2004 in the Upper Lachlan catchment (ACT Government 2004). Because both the Gunyah and Tarcoola sites lie in the Upper Lachlan catchment, the difference in the average of the SOC stocks from the *Vegetated* and *Scald* profiles was taken to determine the loss of SOC associated with salinisation. Therefore, at a very coarse scale, the total loss of SOC stocks in the Upper Lachlan Catchment is estimated to be in the vicinity of 190 000 t of SOC to a depth of 30 cm. It should be noted, however, that extrapolation of results from paddock scale to one at a catchment scale should always be done with caution due to differences across catchments in geomorphology, geology and soil types. A number of sites in the Upper Lachlan catchment will need to be assessed to further refine this figure.

Morevoer, it is also notoriously difficult to have well paired sites in saline and sodic landscapes, as the expression of salinity and sodicity are dependent on geomorphology. Salinity and sodicity effects occur at a catchment scale, and exhibit high temporal and spatial variability. These effects are usually evident in low areas of the catchment and where there is a break of slope (McFarlane and George 1992). One option in a paired sites study in salt-affected areas is to select a scalded site, with its opposite pair located in an adjacent catchment at the same position in the landscape which is not salt-affected. However, this option also has drawbacks as differences in hydrology and geology, in particular, will render the pair incomparable. Therefore, in this study, the pairs that were chosen were located within the same catchment at the same landscape position. It is

likely that soil types between saline and/or sodic sites and unaffected sites will be different due to differences in chemistry as a result of salinity and sodicity. Therefore, in selecting paired sites in such studies, while soil types cannot be exactly matched, the underlying geology, land use, hydrology, vegetation and geomorphology should be the same where possible.

Whilst it was unfortunate that the *Tarcoola Depression* site had what was most likely an in-fill layer overlying a buried soil surface, such incidences are not unique in the Southern Tablelands region of NSW (B. Murphy, pers. comm.). However, as the microsites had already been established, it was not viable to establish another *Vegetated* microsite. Therefore, the *Tarcoola Vegetated* profiles were sampled to compare the *Tarcoola Scald* site with another vegetated site.

6.5 Summary and Conclusion

Scalded and vegetated profiles display very different soil properties at the surface and at depth. These differences are most likely due to the presence of vegetation, which mediates soil properties largely through organic matter deposition and processes related to root respiration and growth. Thus, soil pH, EC, SAR and ESP were generally lower in the surfaces layers of those profiles with vegetation compared to those without. Total N and SOC are largely related to SOM accumulation, and hence, followed similar patterns. SOC concentration was higher in the profiles that were vegetated with both native and sown pasture, and lower in those profiles that were scalded or eroded. Similarly, SOC stocks followed a similar pattern, with the profiles that had been formerly scalded and subsequently revegetated displaying similar SOC stocks to those under native pasture in the top 30 cm. However, SOC stocks in eroded profiles that had lost the top 5 cm of soil had also lost a substantial amount of SOC compared to a similar scalded profile where the top layer was still intact. Therefore, in salt-scalded areas, SOC is substantially lower than that found in non-degraded vegetated and revegetated profiles, highlighting the losses in SOC stocks as a result of increasing salinity and sodicity. Further losses in SOC will occur if the scalded profiles are subsequently eroded. Chapter 7 will discuss the links between SOC stocks described in this chapter, and SOC flux described in previous chapters.

CHAPTER 7: GENERAL DISCUSSION

7.1 Carbon Processes in Landscapes Affected by Salinity and Sodicity

The accumulation of SOC stocks is essentially a balance between inputs by plants and losses by decomposition, erosion and leaching, with accumulation occurring where inputs are greater than losses. The importance of maintaining SOC levels, particularly in agricultural soils, is well established. This is evident in terms of a soil's buffering capacity, where losses of SOM, particularly in an agricultural soil, can significantly reduce a soil's CEC and hence, retention of available nutrients for plant growth, and the soil's capacity to buffer against environmental changes (Slattery *et al.* 1998). The importance of SOC lies in its close association with the SMB and its impact on plant health, as changes in the soil environment can place the microbial community and vegetation under high levels of stress, as indicated in Chapter 4. Higher levels of SOM can also aid in maintaining soil structure and soil fertility, as reviewed in Chapter 2, but was beyond the scope of this project. This chapter will integrate the results from Chapters 3, 4, 5 and 6, as shown in Table 7.1.

7.1.2 Losses of Soil Organic Matter in Saline and Sodic Environments

The SMB only makes up a small proportion of the total SOC (ie. 1-5%), yet is the driving force of soil C turnover, as all organic material has to pass through the SMB. The benefits of having high levels of SMB are well established, and include efficient soil ecosystem and nutrient cycling processes, and hence, accessibility to plant available nutrients, as reviewed in Chapter 2. Due to the faster turnover rate of the SMB compared to the total SOC pool, microbial parameters can be more sensitive and consistent indicators of management-induced changes to soil quality than other soil physical or chemical properties when comparing the impacts of management (Bending *et al.* 2004).

Table 7.1 Integration of results chapters

Chapter	Description	Experimental Conditions	Key Finding
3	Leaching of non-scalded vegetated soil with a combination of saline and sodic solutions	Controlled temperature and moisture conditions	SMB was highest in the high-salintiy treatments, attributed to more easily accessible and decomposable SOM due to high salt concentrations. Therefore, it is possible that SOC is rapidly lost as salinity and sodicity increase.
4	Gypsum addition to saline-sodic soils which are scalded and free of vegetation	Controlled temperature and moisture conditions	Low levels of SMB and cumulative soil respiration are due to a lack of substrate, confirmed in the SOC stocks in Chapter 6, as the soil surface is scalded. Therefore, in extreme cases, little microbiological activity is occurring in scalded areas.
5	Organic material and gypsum addition to saline-sodic soils which are scalded and free of vegetation	Controlled temperature and moisture conditions	Higher levels of SMB and soil respiration compared to the results from Chapters 3 and 4 are due to availability of substrate for decomposition. Therefore, the SMB is limited by substrate, rather than by high EC, ESP and adverse pH conditions.
6	SOC stocks in salt-scalded, eroded, revegetated and unaffected soil profiles.	Field conditions	Low levels of SOC stocks found in salt-scalded profiles, are compounded by erosion. Following revegetation, SOC stocks can increase to levels similar to those found in unaffected soil profiles.

Increasing salinity and sodicity ultimately results in a decrease in SOC through a number of mechanisms. Chapter 3 showed higher levels of SMB in the high-salinity treatments compared to the control treatments. Soil respiration did not follow similar patterns to the SMB, which is attributed to a shift in the community structure from one dominated by fungi to one dominated by bacteria. The survival of specialised and adapted species in saline conditions may result in a microbial community dominated by bacteria with lower respiration rates compared to a population dominated by fungi (Adu and Oades 1978), with a bacteria dominated community also less active and less diverse (Pankhurst *et al.* 2001). It is possible that the shift in community structure will also influence the qCO_2, as discussed in Chapter 3.

It is suggested that the higher levels of SMB in the high-salinity treatments found in Chapter 3 is due to the increased solubility, decomposability and accessibility of SOM. Extrapolating from these results, it is also suggested that with the onset of salinity and sodicity, native SOM can be rapidly lost. Concurrently, C inputs into the soil are decreased as salinity and sodicity cause plant health to decline through adverse soil physical and chemical conditions. Under these conditions, it is likely that concentrations of dissolved SOC increase due to increased solubility of SOM. This process provides additional substrate which is easily decomposed by the microbial population, as shown by Jandl and Sollins (1997), and can also be easily lost by leaching. Dispersion of aggregates due to sodicity, many with cores containing organic material (Tisdall and Oades 1982), also increases the availability of C. As a result, SOC accessibility and degradability is increased for the microbial population, which can also offset stresses placed on the microbial biomass, discussed below. It is also possible that additional substrate can become available for decomposition when SOC is released from clays with increases in salinity. Under such conditions, SOM adsorbed on clays is released due to exchange processes as cations flood exchange sites, as described in Chapter 3. It should be noted, however, that the SMB levels and soil respiration rates found in Chapters 3, 4 and 5 represent the maximum response of the active C pool, as the analyses were conducted on disturbed samples. Therefore, oxygen availability was increased compared to *in situ* conditions in the field, particularly at depth. In addition, the soils were placed in optimal moisture and temperature conditions during the experimental period.

Initial losses of SOC can be attributed to the response of the faster-cycling C pools that contribute most of the decomposition flux according to the processes suggested above. However, in the longer term, decadally cycling pools continue to lose C at rates that are significant in terms of ecosystem level C storage, but are frequently not detectable as they represent less than 5 % increase in soil respiration rates after the first several years (Trumbore 2006). Death of vegetation occurs with high levels of salinity and sodicity, resulting in bare, scalded patches which are increasingly susceptible to further losses of C by water and wind erosion, as seen in Plates 6.2, 6.5 and 6.6, and in the low SOC stocks in the *Scald* and *Eroded* profiles discussed in Chapter 6. The SMB is placed under increasing stress as substrate availability and decomposability decline, with little SOC input occurring due to the absence of vegetation. It is likely that vegetation death, which results in scalding of the soil surface, will generally precede the decline in SMB,

as vegetation is generally less tolerant of saline and sodic conditions. This was shown in a study by Rietz and Haynes (2003), who found that sugarcane yields were negatively correlated to sodicity rather than salinity, while the qCO_2 increased with both salinity and sodicity. However, despite the decreases in microbial activity, there was still substantial activity occurring in areas where vegetation had died. Over time, the microbial population can become adapted to a high salt environment (Polonenko *et al.* 1981; Zahran 1997), which may have been the case in the studies reported in Chapters 4 and 5. For example, following leaching with distilled water, the qCO_2 was 0.080 mg CO_2-C/d/mg SMB-C, while leaching with the high-salinity high-sodicity solution gave a qCO_2 of 0.010 CO_2-C/d/mg SMB-C in the surface layer of soil in Chapter 3. In comparison, the qCO_2 found at the same site from a scalded profile was 0.078 CO_2-C/d/mg SMB-C in Chapter 4, which suggests some adaptation to the soil environmental conditions found at the site of high ESP and high pH. However, as suggested previously, the *q*CO2 in this study only provides an indication of the stresses placed on the microbial population, while Section 2.3.1.1 describes the mixed results which have resulted with the use of the qCO_2. Further research with the use of other microbial indices such as dehydrogenase activity and arginine ammonification rate and fluorescein diacetate (FDA) hydrolysis (Chander and Brookes 1991a; Haynes 1999), which measures enzymatic activity, and ergosterol content, which measures fungal biomass (Rasul *et al.* 2006), would clarify the results found in this study.

Low levels of SMB were found in scalded profiles (Chapter 4), which were attributed to limited SOC input at these sites. However, it is suggested that the SMB in scalded soils is dormant, and becomes active where substrate, such as kangaroo grass, is available for decomposition. The adapted microbial population can rapidly multiply when substrate becomes available despite adverse soil conditions. This most likely occurred following the addition of organic material in Chapter 5, where the C_{mic}:C_{org} increased compared to those results found in Chapter 4, while the qCO_2 decreased. It is possible that increased substrate availability can offset the stresses placed on to the microbial community. This can also occur in the field either through direct incorporation of organic material, such as straw, in the rehabilitation process or increasing vegetation cover through replanting. The addition of gypsum with organic material did not adversely affect the population, despite increases in soil-solution EC. Therefore, it may be possible to re-establish microbial activity and hence, nutrient cycling in the field following the addition of organic material in conjunction with gypsum. As plant growth is established, SOC input

increases, largely due to inputs from litterfall, and rhizodepositions. The presence of plants further promotes microbial activity and the build-up of microbial biomass, as root exudates are a source of substrate for the microbial community, favouring remediation processes (Tejada *et al.* 2006).

7.1.3 Soil Properties and Geomorphic Factors

A number of opposing processes affecting SOC stocks and fluxes occur during salinisation and sodication. Dalal and Mayer (1986) have linked the loss of SOM to factors that affected its accessibility and stability to attack by the microbial population and enzymes. Macro- and microaggregates can contain, and physically protect a considerable portion of SOC (Conant *et al.* 2004). In sodic soils, dispersion of aggregates on wetting can increase substrate accessibility and availability (Oades 1984). However, on drying, the bulk density of a soil increases and waterholding capacity decreases which decreases the availability of SOM to the microbial biomass. Thus, hardsetting soils of high bulk density restrict substrate availability to the microbial population due to the breakdown of soil structure on wetting and its subsequent formation of massive structure when dry, as substrate can be located in pores that are too small for the microbial population to access.

Wetting of the soil, either through rainfall or irrigation, can also result in soil structural breakdown at the soil surface and the formation of surface crusts. These crusts result in restricted infiltration causing waterlogging on the soil surface and dry subsoils. Such conditions further decrease the decomposition of SOM. In saline soils, high soil-solution EC results in flocculation of clay particles into aggregates which may also restrict substrate availability and hence, the decomposition of SOM. Any process which slows decomposition in normal circumstances will also result in increases in SOC. For example, Tan *et al*. (2004) found that poor drainage conditions favour C sequestration, regardless of land use, as a result of reduced oxidation of SOC from the upper layers of soils. Similarly, at depth, waterlogging will also enhance SOC accumulation due to reduced oxygen availability. This process probably also played a role in the higher concentrations of SOC found in the buried surface horizon (50-70 cm layer) in the *Tarcoola Depression* profile, which was very moist, as discussed in Chapter 6 and described in Appendix D.

The presence of perched ephemeral aquifers has implications for C flux and oxygen availability for soil biota and is frequently responsible for waterlogged conditions in the landscape. However, despite the common occurrence of waterlogging, in general, saline and sodic soil conditions result in losses of SOC due to their adverse effects on plant growth. This could occur either directly through ion toxicities, or indirectly through decline in soil structure and limited access to nutrients and water for plants. Because vegetation cover is the dominant factor in determining SOC stocks and fluxes, SOC levels will most likely show a general net loss in the long term if highly saline and sodic conditions persist.

In scalded soils, the A horizon has frequently been eroded, as shown in the *Gunyah Eroded* profile in Chapter 6. Following erosion, the less fertile B horizon remains as the soil surface. Because SOC generally decreases with depth (Murphy *et al.* 1998), erosion and increased mineralisation of the SOM in the B horizon result in a substantial loss of soil C. Transported sediments are frequently enriched in SOC, of relatively low density due and concentrated close to the soil surface. Therefore, where erosion occurs, SOC levels are lower in eroded compared to uneroded soils (Lal 2001). This was evident in the eroded profile in Chapter 6, with the *Gunyah Eroded* profile containing half the SOC stocks found in the *Gunyah Scald* profile. This indicates that a substantial amount of SOC can be lost in scalded soils as a result of erosion, particularly where the topsoil is lost in the process in addition to SOC losses as a result of scalding.

Areas affected by salinity and sodicity are characterised by high spatial and temporal heterogeneity in the landscape. While SOC stocks are generally assessed at a regional scale, variations between vegetated and scalded soils occur at a paddock scale, resulting in difficulties in the accurate determination of C stocks and fluxes. Topographic effects further complicate assessments, as soils in lower slope positions have EC profiles which decrease with depth due to evaporation from shallow groundwaters, while those from upper slopes show increasing EC with depth, indicating that leaching has taken place (Harker and Mikalson 1990). Therefore, it is difficult to determine differences in SOC stocks in salt-affected compared to non salt-affected landscapes. Because expressions of salinity and sodicity are frequently governed by topography, those salt-affected sites are likely to be located in lower slope positions, while a non salt-affected analogue in a similar slope position may be difficult to establish, as described in Chapter 6. Also, because increasing salt concentrations can increase SOC losses, as shown in Chapter 3,

EC profiles and salt flux are likely to affect mineralisation of SOC at different depths in the profile. This has implications for C dynamics, as SOC is frequently less labile, older and more stable at depth (Wang *et al.* 1996).

7.2 Building Up Soil Organic Carbon Stocks

There is a high potential to build up SOC stocks in salt-scalded areas as there is a higher capacity to accumulate SOC where stocks are initially low. In this study, the low SOC stocks found in the *Scald* and *Eroded* profiles in Chapter 6, which were nearly three times lower than those found in *Vegetated* and *Pasture* profiles, are most likely caused by the absence of vegetation cover on those areas. This results in little or no C input in the scalded areas. In Chapter 4, soils sampled from a scalded profile also displayed low levels of SMB and soil respiration. Similarly, Pankhurst *et al.* (2001) found lower SOC levels in saline soils compared to non-saline soils. They attributed this to reduced inputs of organic matter due to sparser plant cover and the reduced presence of salt sensitive pasture. Soil C stocks are influenced by land use and land management practices, and hence, any decrease in biomass production will also decrease SOC levels. Due to very low SOC stocks in salt-scalded profiles, successful revegetation of these landscapes can result in rapid SOC accumulation. For example, revegetation with introduced pasture at the Gunyah site resulted in an increase in SOC stocks to levels similar to those found under native pasture when assessed to a depth of 30 cm after 10 years. However, these results are specific to this site alone, as the results are only indicative of the possible magnitude of the impact of planting pasture. Further research is required to confirm these findings with a replicated field study based on a time-series of change after scalds are re-vegetated.

7.2.1 Land Management and Rehabilitation of Salt-Affected Areas

In saline and sodic areas, the key issue in rehabilitation is the maintenance of biomass production in an environment that is essentially adverse and often prohibits plant growth. The influence of vegetation on a number of soil properties and processes is well established, as discussed in Chapter 2. Briefly, the maintenance of soil structure in saline and sodic profiles is aided by the presence of vegetation and its associated root systems. Roots and root hairs are continuously decomposed, while root mucilages stabilise soil structure in the area surrounding the root-zone (Oades 1984). Clays can

stabilise SOM through direct interactions with microbes, alter the rate and pathways of microbial metabolism, and promote aggregation through sorption (Sollins *et al.* 1996). The incorporation of organic material can also stabilise clays into macroaggregates and increase the CEC, with preferential retention of Ca^{2+} over Na^{+} (Muneer and Oades 1989b). As nutrient levels in saline and sodic soils are frequently low, litterfall can provide a substantial concentration of nutrients, which can sustain plant growth (Garg 1999).

The presence of plant roots has been shown to increase P_{CO2} in aerobic (Mishra and Sharma 2003) and waterlogged soils (Boivin *et al.* 2002). Because salt-affected soils commonly occur in alkaline conditions, the increase in P_{CO2} has been shown to decrease soil pH. This effect was reflected in the *Vegetated*, *Pasture* and *Gully* profiles, which were all vegetated, at both the Tarcoola and Gunyah sites, as discussed in Chapter 6. In Australia, sodic soils are commonly alkaline and contain $CaCO_3$ in the profile, usually in the subsoil but remains relatively insoluble due to high pH conditions. Decreasing soil pH by increasing P_{CO2} and organic matter in soils prevents $CaCO_3$ precipitation and enhance its solubility, facilitating the reclamation of sodic areas (Chorom and Rengasamy 1997). However, this process may release previously sequestered C; further examination of this is beyond the scope of this project.

As the demand for high quality water for urban supply increases, the use of lower quality irrigation water for agricultural areas will also increase, as described in Section 3.1. However, attempts to increase biomass production through application of poor quality irrigation water, which is often saline and/or sodic, can result in the development of moderately to highly saline and sodic soils with the concomitant decline in biomass production (Rogers 2002). According to the processes described in Chapter 3, it is possible that with the application of saline irrigation waters, losses of native SOM will most likely increase as SOM can be rapidly solubilised and lost. Similarly, where hydraulic conductivity and infiltration need to be improved in a sodic soil, a solution of high EC is required to ameliorate the soil. However, as outlined above, increasing EC can deplete SOM stores in the soil prior to remediation taking place.

While this project has focused on the amelioration of pasture systems affected by salinity and sodicity, other studies have shown that catchments planted with trees reduced levels of salinity due to lower recharge rates (eg. George *et al.* 1999; Schofield

1992). The planted trees then lower water table levels to below the salt bulge in the soil profile and hence, prevent the release of more salt into the groundwater (Salama *et al.* 1993b). It has been suggested that catchments require between 70-80 % tree cover in order for groundwater levels to stabilise or decrease (George *et al.* 1999). Revegetation in recharge areas may be more successful than the process of revegetating seepage areas or areas that are scalded. Recharge areas are frequently at higher positions in the landscape, where the expression of salinity and sodicity may not be as severe, and hence soil environmental conditions may be less hostile such that plant establishment and growth may be more successful. Revegetation of discharge areas may only be a short to medium term strategy if plant establishment is initially successful, as evapotranspiration by trees will concentrate the salts and cause an increase in the salinity of the groundwater in the longer term (Stolte *et al.* 1997). Concentration of salts not only occurs following revegetation with trees, as perennial shrubs and grasses may also accumulate salt in their root zones which can lead to vegetation health decline in the longer term (Barrett-Lennard 2002). Difficulties also exist in establishing and maintaining vegetation growth in salt-affected sites, as revegetation needs to be successful at a catchment scale to reduce watertable levels. In addition, sites severely affected by salinity which are of high value will most likely also require engineering strategies to be employed to pump groundwater in conjunction with revegetation for these areas to remain in production (Clarke *et al.* 2002).

Not only can the presence of trees reduce recharge rates, but they can also increase C inputs into the soil to a greater depth compared to pasture. A study by Young *et al*. (2005) found that SOC stocks under pasture and woodland were comparable to a depth of 20 cm. However, when stocks were assessed to a depth of 1 m, woodland soils contained significantly more SOC than the pasture soil. This was a result of C allocation by deeper roots in sites with trees. The allocation of SOC at depth can decrease decomposition rates and hence, enhance SOC accumulation. This is mainly attributed to limited N availability, limited oxygen availability and increased bulk density, while SOM in the form of plant roots is of a lower quality (Newey 2005).

Estimation of SOC stocks to a depth of 30 cm can result in an underestimation of actual stocks, as described in Chapter 6. The lower depth limit of 30 cm has been established to focus on the effects of land use and management on the labile C pool, as labile C dominates the upper layers of a soil profile and is easily oxidised and lost. For example,

a study by Jinbo *et al*. (2006) found that the effects of land use on the total SOC and labile fraction organic C were mainly observed in the upper 20 cm of the soil profile, with leachate from the topsoil providing a substantial portion of SOC at depth. However, a meta-analysis by Conant *et al.* (2001) found that SOC can be gained or lost at depths greater than one metre following land use or land management conversion resulting in an underestimation of SOC stocks. In this study, one SOC profile assessed to a depth of one metre displayed significant accumulations of SOC below 30 cm, attributed to the burial of a former surface horizon, a process which is not uncommon in saline and sodic landscapes (Chapter 6).

7.2.2 Gypsum and Organic Amendments

This research demonstrates that where sodic or saline-sodic soils are remediated, the presence of organic material can aid in re-establishing soil ecosystem functions. The use of Ca compounds as soil ameliorants is essential, particularly in sodic or saline-sodic soils, as described in Chapter 2, while the presence of high levels of Ca can aid in plant growth and establishment as shown in Chapter 6. The addition of Ca compounds can accelerate changes in soil-solution composition conducive to reclamation of sodic soils. Addition of Ca as gypsum or lime is critical for plant growth in saline and sodic sites, which are frequently Ca-limited, resulting in Ca deficiencies in plants (Reid and Smith 2000). Initial addition of gypsum to sodic soils in the field aids in improving soil physical properties for vegetation growth. It is likely that further reclamation of sodic soils by organic matter can also facilitate remediation, as demonstrated under controlled conditions in Chapter 5. In this study, gypsum caused a reduction in ESP and, in alkaline conditions, resulted in reduced soil pH, as shown in Chapters 4 and 5.

It has been suggested that increasing the SMB will generally improve soil condition. Microbial cells generally possess a net negative charge which assists in flocculation of clay particles (Oades 1984) which improves soil structure, while the decomposition of organic material by the SMB is essential for nutrient cycling. In sodic soils with high levels of insoluble Ca, commonly found in arid and semi-arid regions of Australia due to the presence of $CaCO_3$, an increase in the SMB may aid in the greater solubilisation of Ca^{2+}. The ESP can be decreased in these areas as a result of greater CO_2 evolution and humic acid formation from decomposition of SOM (Malik and Haider 1977). Similarly, linkages can be formed between products of microbial decomposition

processes and Ca^{2+} to further aid in improving soil condition (Baldock and Skjemstad 2000). Humic acids are stronger acids than H_2CO_3, which is commonly formed in the presence of respiration and water, and can have a greater potential to dissolve inorganic carbonates while releasing Ca^{2+} (Nelson and Oades 1998), as described in Chapter 5. Mineralisation of organic matter also has the potential to release Ca, as complexes with Ca are formed more readily than those with Na (Nelson *et al.* 1998).

Despite increases in EC following the addition of gypsum, microbial activity remained unaffected (Chapters 4 and 5). The maintenance of microbial activity was most likely due to the presence of a microbial population that was adapted to high EC and pH conditions, particularly where scalded soils have been present for many decades. Results of previous studies on the effect of salinity and/or sodicity on soil microbiological processes have been contradictory, particularly where salinity has been induced, as described in Chapter 2. The contradictory effects may have been due to a range of adaptation mechanisms, or lack thereof, by the microbial community to saline soil environmental conditions in the different studies. Thus, in those soils where salinity and/or sodicity have occurred for a number of years, the microbial population has most likely developed adaptations to cope with hostile environments. In addition, higher activity of Cl^- ions compared to SO_4^{2-} ions can produce a greater increase in EC, with Cl^- more toxic to the microbial population at the same EC (Garcia and Hernandez 1996). The sites of Avoca and Tarcoola, discussed in Chapters 4 and 5, have been scalded for periods of approximately 10 and 60 years, respectively; this has most likely allowed the microbial population time for adaptation to such conditions at both sites. As a result, the addition of gypsum had little or no effect on the SMB.

The presence of plant cover in establishing an active SMB for nutrient cycling is important and has been well established by studies relating to ecological succession (eg. Rutigliano *et al.* 2004). Amelioration of hostile soil environmental conditions through the addition of gypsum alone, or in combination with lime, has been linked to higher levels of plant growth and accumulations of SOC (Valzano *et al.* 2001b). However, in terms of management of saline and sodic landscapes, while reclamation of soils with the use of gypsum may remediate soil conditions in the root zone of plants, leaching of Na^+ ions may lead to further problems due to increasing sodicity at depth and in the groundwater (Surapaneni and Olsson 2002). Australian soils commonly have different layers in the soil profile which suffer from different constraints in the different layers,

resulting in difficulties when attempting to establish or maintain plant growth. This is often due to the common occurrence of duplex soils, where the topsoil of a profile may be sodic while the subsoil is saline or vice versa (Rengasamy 2006). Revegetation was successful in restoring SOC stocks in this study at the Gunyah site by revegetation alone, which is attributed to neutral pH values and adequate Ca concentrations, as discussed in Chapter 6. However, it is likely that at more severely affected sites, such as Tarcoola, a number of strategies will need to be employed for revegetation to be successful. This includes deep-ripping to break up the hard pan that has formed as a result of sodicity and the addition of ameliorants, which may include gypsum, lime, organic material, or a combination of all three prior to sowing, as discussed in Chapter 6.

Lime is a commonly-used ameliorant where soils are acidic, and it would have been of interest to assess the differences in the effects between lime and gypsum in the SMB and microbial respiration, particularly in the acidic Avoca soils discussed in Chapters 4 and 5. However, time and budgetary constraints prevented further investigation. A previous study by Haynes and Naidu (1998) showed that short term effects following additions of lime to ameliorate soil pH conditions have resulted in a flush of microbial activity, and hence, increased mineralisation rates and loss of SOC content while soil aggregation was improved due to production of microbial products. This was attributed to an improvement in soil environment, with pH conditions more amenable for microbial growth. Where soils are moderately saline and/or sodic, a similar process may occur following the addition of gypsum, with a flush of microbial activity when pH conditions improve, followed by a longer term build up in SOC levels as plant growth improves due to improved soil conditions.

The effect of soil pH change is likely to play a major role in C dynamics, particularly in the degradation and rehabilitation processes of salt-affected landscapes, due to its effects on both the microbial population and vegetation health. Results reported in Chapter 6 indicated that soil pH was consistently lower where plant growth had become established. The presence of vegetation is likely to mediate pH conditions affected through processes such as the production of root exudates and litterfall for decomposition (Kemmit *et al.* 2006; Xu *et al.* 2006). Soil pH change, as reported in Chapter 5, appeared to affect respiration, but not SMB, while pH did not appear affect microbial activity (Chapter 4). High pH decreases the solubility and availability of a number of plant nutrients including phosphates, Fe, Zn and Mn, while low pH can

induce iron and aluminium toxicities (Russell 1973) for plant growth. Therefore, to increase SOC stocks, soil pH conditions will need to be mediated in order for plant growth to become re-established in salt-affected areas, and hence, allow for efficient functioning of the microbial population.

The addition of organic material in the form of manures, sewage sludge and plant material for rehabilitation of degraded landscapes has been commonly undertaken in the past (eg. Kumar and Singh 2003; Liang *et al.* 2005; Suriadi *et al.* 2002; Tejada *et al.* 2006). In a previous field-based study, the incorporation of organic materials into mine spoils, which were originally free of vegetation and susceptible to erosion and crusting, was shown to decrease crust strength and increase soil moisture, allowing for salt-tolerant plants to establish in a saline-sodic environment (Grigg *et al.* 2006). Chapter 5 showed that the addition of plant material to salt-affected soils can aid in the rehabilitation process by increasing the SMB and microbial activity, and decreasing soil pH in highly alkaline conditions. Organic material can provide a buffer and reduce microbial sensitivity to adverse soil conditions (McCormick and Wolf 1980), while the solubilisation of organic matter at high pH into colloidal forms results in increased availability of substrates, thus relieving the pH stress on microbes (Pathak and Rao 1998).

The addition of plant material, as reported in Chapter 5, also showed that the amendment incorporated does not need to have a narrow C/N ratio, despite low N contents present in the soil. However, while the addition of organic material and gypsum may aid in the recovery of scalded areas, the overall aim of the rehabilitation process is to establish vegetation on these vegetation-free areas so that the incorporation of organic material is part of a self-sustaining system. One study showed that the incorporation of organic amendments may lead to spontaneous vegetation growth on a saline soil due to the amelioration of ESP and soil structure, and the efficient functioning of microbial and enzymatic activities (Tejada *et al.* 2006). It should be noted, however, that while SOC stocks can accumulate following successful revegetation quite rapidly, Buyanovsky and Wagner (1995) found that in a short period of time, it is unlikely that the main reserve of SOM can be significantly altered, with the build up in SOC stocks likely to result in an initial build up of the more labile fractions. Therefore, if vegetation fails to establish, the accumulated SOM can also be rapidly lost.

7.3 Summary

The potential to accumulate significant amounts of SOC in salt-affected landscapes is high, as SOC stocks are initially low in salt-scalded areas. It is suggested, from Chapter 3, that during the degradation process, SOC can be rapidly lost as the SMB increases with increasing salt concentration which is attributed to the increased solubility and decomposability of native SOM. As salinity and sodicity continue to increase, SOC loss continues as decomposition continues, while SOC inputs decline as vegetation productivity decreases. In extreme cases, scalding of the soil surface occurs resulting in very low SOC stocks. Scalded soils are susceptible to further losses caused by erosion, resulting in low levels SMB in both scalded and eroded profiles, probably due to low levels of SOC. However, it is suggested that SOC can also be rapidly accumulated during reclamation by the addition of organic material, replanting, or a combination of both, which can increase the standing biomass and hence, increase in SOC stocks. It is also likely that soil ecosystem processes are also restored in the process, which results in efficient nutrient cycling, and hence, C cycling.

CHAPTER 8: CONCLUSIONS

8.1 Research objectives revisited

Salinity, sodicity and SOC dynamics are three critically important yet seemingly separate issues in natural resource management. By investigating all three issues, this study found that salinity and sodicity adversely impacted upon SOC stocks, largely due to the importance of vegetation production on SOC inputs. Increasing salinity and sodicity results in declines in vegetation health, and hence, decreases SOC inputs. Similarly, during the rehabilitation process, SOC can be accumulated as a result of revegetation to levels similar to those found in unaffected soils. Moreover, the lower SOC stocks found in saline and sodic landscapes were not due to impacts on vegetation alone. The overall aim of this thesis was to determine SOC dynamics as affected by salinity and sodicity. This section revisits the objectives of this study, as set out in Section 1.1. The processes involved in decreasing SOC during degradation, and conversely increasing SOC during rehabilitation are described below.

8.1.1 Quantification of the effects of different levels of salinity and/or sodicity on carbon stocks and fluxes

Under controlled temperature and moisture conditions in the laboratory, the SMB was highest in the high salinity treatments. It was suggested that the high levels of SMB were due to the increased solubility of the SOM which renders it more easily decomposable. Soil respiration did not follow the same patterns as the SMB, which may have been due to a shift in community structure, from one dominated by fungi to one dominated by less active bacteria. Therefore, as salinity and sodicity increase, it is suggested that SOC input decreases due to declining vegetation health, while the SOM present continues to be decomposed. As a result, SOC can be rapidly lost where salinity and sodicity levels increase in a vegetated soil profile.

8.1.2 Determination of the behaviour of the labile carbon pool in a saline-sodic soil, with and without gypsum amendment

The SMB and cumulative soil respiration rates were low over the 12-week incubation period in soils sampled from scalded areas when compared to soil from a vegetated profile. The addition of gypsum did not affect the SMB or soil respiration. It is suggested that the low levels of SMB and soil respiration are the result of limited C input due to a lack of vegetation associated with the scalded areas. Vegetation-free areas are most likely caused by adverse soil pH, EC and ESP conditions.

8.1.3 Determination of how decomposition is affected in saline-sodic soils following addition of organic material, with and without gypsum amendment

Following the addition of organic material in the form of kangaroo grass, the SMB and soil respiration rates from two scalded profiles increased to decompose the available substrate. This occurred despite the adverse soil environmental conditions of high salinity and sodicity, high alkalinity in the Tarcoola soil and high acidity in the Avoca soil. It is therefore suggested that the increase in the SMB and respiration rates indicates that a dormant salt-tolerant microbial population is present in salt-scalded soils which multiplies rapidly when substrate is available. The addition of gypsum did not affect the population despite increasing the EC of the soil solution. This suggests that microbial activity is limited by substrate in scalded areas and not by adverse soil conditions such as high EC, ESP and pH.

8.1.4 Quantification of soil organic carbon stocks in vegetated, salt-scalded and revegetated profiles

SOC stocks were up to three times less in scalded profiles compared with those profiles that were under native vegetation. In a scalded profile where the topsoil had been eroded, further losses of SOC had occurred, with SOC stocks half of that found in the scalded profiles. Where one of the scalded areas had been revegetated with introduced pasture, it is possible that SOC stocks to a depth of 30 cm are comparable to those found under native vegetation which had not been degraded. It is tentatively suggested that rehabilitation of these salt-scalded landscapes by revegetation has the potential to

restore SOC stocks to original levels to a depth of 30 cm where plant growth can be established.

8.2 Limitations of the Research

In all projects, time and budgetary constraints limit the scope of what is to be achieved. This project concentrated on soils characteristic of the Southern Tablelands region of NSW, which were duplex in character and formed on Devonian and Ordovician metasediments. However, areas in the southwest of Western Australia (WA), in Victoria and South Australia are extensively salt-affected, and salinity is also described as an emerging problem in Queensland. Because soil texture can play a large role in C stocks and dynamics, it is important to determine the differences in C dynamics and C stocks in these other areas of Australia that are salt-affected, particularly in the sandier textured soils of WA. While it was determined that SOC stocks in scalded areas were up to three times less than those found in vegetated areas, it would be unwise to scale this figure up to a regional level from two sites. A more extensive determination of SOC stocks is required in order to accurately gauge SOC stocks from salt-affected landscapes in Australia.

8.3 Future Research

This research has shown that increasing salinity and sodicity results in increased C mineralisation, and hence, increased soil C losses. In those areas that are extensively scalded and eroded as a result of scalding, C stocks can be up to five times less than those found in areas that are vegetated and not eroded. Where salt-affected areas have been revegetated, SOC stocks can be increased to levels comparable to non salt-affected areas. While this research has established baseline data in terms of C stocks and fluxes in salt-affected soils, further research is required if the effects and implications of salinity and sodicity on C stocks and fluxes are to be fully understood. As previously discussed, a replicated field study based on a times series of change following revegetation of scalds would confirm the extent to which SOC stocks can be accumulated. Because the C and N cycles are intricately linked, and Australian soils are frequently N-limited, research is also required into how salinity and sodicity affects the N cycle and whether N dynamics follow patterns similar to that of C dynamics in these degraded areas. The C sequestration potential in these severely degraded salt-affected

areas is high due to the low C stocks that are currently found. Potential, therefore, exists for studies to determine how to maximise C stocks while re-vegetating salt-affected areas for crop and pasture production. Similarly, as agroforestry increases in popularity as a means of controlling groundwater levels, potential also exists for studies to determine how to maximise C stocks in such land use practices.

The data presented in this project does not represent a complete C budget. While an in-depth study on the effects of salinity and sodicity on SOC was undertaken, the effects and influences of carbonates, and hence SIC, were not examined. It is likely that SIC plays a large role in the C cycle in these degraded landscapes, particularly in soils where alkalinity is an important issue, such as the site at Tarcoola. For a complete understanding of C dynamics in salt-affected soils, the role SIC plays will need to be determined, particularly in areas where groundwater high in CO_3^{2-} and HCO_3^- interacts with respiration from soils and vegetation to form SIC. Due to the common occurrence of alkaline conditions in saline and sodic landscapes, the effect of inorganic C in the form of $CaCO_3$ will most likely play a large role in C dynamics in these landscapes. The role of carbonates will be particularly important where processes affect changes in soil pH. Where Ca^{2+} is mobilised from $CaCO_3$ as a result of decreasing pH from decomposition and respiration processes, soil physical properties can be improved. However, CO_2 will be released from respiration processes in addition to the dissolution of $CaCO_3$. While the issues related to inorganic C are beyond the scope of this project, loss of C related to $CaCO_3$ solubilisation is likely to play a substantial role in C flux in these alkaline landscapes.

The effects of pH have been notable throughout the trials conducted within the project. Those soils sampled from areas affected by salts in general showed alkali pH values, while leaching of a non-degraded soil caused the pH to decrease to values that affected the microbial community. Where highly alkaline conditions occur (ie. pH > 8), the potential exists for respired CO_2 to be sequestered as inorganic C. Conversely, where the pH decreases, which occurred following leaching, any inorganic C becomes soluble and available for mineralisation. The effects of changing soil pH on the active C pool cannot be discounted, and further research is required to determine how soil pH affects C stocks and fluxes. While salinity and sodicity are major soil degradation issues in Australia, soil acidification also plays a major role in soil degradation, and its effects on C would also be of interest in terms of C accounting.

To assess the effects of salinity and sodicity on soil carbon stocks and fluxes, this project focused on the labile C pool and measured SMB and respiration over time. The methods used in this study assessed the labile C pool by measuring the SMB and soil respiration rates. However, the assessment of functionality of the microbial population, the population structure, and the determination of the SMB by different methods would be useful. It would be interesting to determine how salinity and sodicity affect SOC and decomposition by focussing on the chemical, rather than biological aspects by assessing the chemical composition of the SOM and the extent of decomposition. It is likely that the increase in SOC in scalded profiles following revegetation with introduced pasture, as described in Chapter 6, was predominantly due to an increase in the more labile POC fraction, rather than the more stable humus fraction. This requires further investigation.

By addressing these issues of uncertainty, our understanding of C cycling in an environment degraded by salinity and sodicity and during the rehabilitation process, will be enhanced. This will allow for more accurate assessments of C stocks and fluxes, and the promotion of management practices to maximise accumulations of SOC stocks where rehabilitation efforts are undertaken.

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APPENDIX A

A1.1 Bulk Density

Bulk density was determined by extracting soil cores of volume 91.952 cm^3 in the field. The cores were hammered into appropriate depth interval of a soil pit so that the mid-point of each depth interval could be sampled. In the 0-5 and 5-10 cm depths, the whole interval was samped as each core had a height of 5 cm. The cores were oven dried at 105^oC for 24 hours before weighing to determine the mass of dry soil per unit of volumetric space occupied.

A1.2 Preparation of 1:5 extracts

Electrical conductivity (EC), pH and soluble cations were measured in 1:5 soil:water extracts. A 1:5 soil:water extract was shaken for one hour on a rotary shaker, centrifuged for 10 minutes at a rate of 2000 rpm and filtered through Whatman's No. 41 filter paper prior to analysis.

Exchangeable cations were extracted with 1 M CH_3COONH_4 buffered to a pH of 7 with acetic acid. Where the EC $\geq$ 0.3 dS/m of the 1:5 soil:water extract, soluble salts were removed with an ethanediol/ethanol wash, described below. The 1:5 soil: CH_3COONH_4 extracts were shaken for one hour on a rotary shaker, centrifuged for 10 minutes at a rate of 2000 rpm and filtered through Whatmans's No. 42 filter paper. Each sample was extracted three times, and the extract made up to 100 mL.

The removal of soluble salts is based on a method described in Rayment and Higginson (1992). 100 mL of ethanediol and 36 mL of deionised water was bulked to 1 L with ethanol. 20 mL of the ethanediol/ethanol mixture was added to 2.5 g of air-dried soil, and shaken for 30 minutes on a rotary shaker. The samples were centrifuged at 3000 rpm for five minutes, and the supernatant decanted. The process was undertaken twice per sample prior to extraction with CH_3COONH_4 described above.

A1.3 pH, Electrical Conductivity, Soluble and Exchangeable Cations Measurements

pH was measured using a standard pH meter with a Denver Instrument Ultra Basic UB-10 pH/ mV meter after calibrating with standard buffer solutions of pH 4.0 and 7.0. EC was measured using a Radiometer CDM3 conductivity meter. Soluble and exchangeable cations were determined by Inductively Coupled Plasma-Atomic Emission Spectroscopy (ICP-AES).

APPENDIX B

Table B1 An example of CO_2 calculations

Dish No.	Depth (cm)	EC	SAR	Replicate	Weight Soda Lime (Pre incubation) (g)	Weight Soda lime (Post Incubation) (g)	Difference (g)	Average Blank (g)	CO_2 (mg/g)	Oven dry equivalent soil (g)	CO_2 (mg/g/2 wks)	CO_2-C (mg/kg/2 weeks)	CO_2-C (mg/kg/ wk)
61	2.5	0.5	1	1	101.9142	102.4319	0.5177	0.3075	0.3552	69.8202	5.0879	1387.6083	693.8042
62	2.5	0.5	1	2	102.4875	102.9381	0.4506	0.3075	0.2418	69.5048	3.4795	948.9430	474.4715
63	2.5	0.5	1	3	104.5749	105.0286	0.4537	0.3075	0.2471	69.5836	3.5508	968.4022	484.2011
76	2.5	0.5	30	1	100.3511	100.8074	0.4563	0.3075	0.2515	68.1559	3.6897	1006.2705	503.1353
77	2.5	0.5	30	2	103.8100	104.2187	0.4087	0.3075	0.1710	68.2529	2.5058	683.3995	341.6998
78	2.5	0.5	30	3	107.7663	108.1383	0.3720	0.3075	0.1090	67.8865	1.6057	437.9168	218.9584
Blank				1	105.7435	105.4407	0.3028						
Blank				2	101.0943	101.4011	0.3068						
Blank				3	105.858	106.1708	0.3128						

APPENDIX C

Table C1. Soluble cation concentrations following addition of organic material

Depth (cm)	Site	Gypsum (t/ha)	Al ($cmol_c/kg$)	Ca ($cmol_c/kg$)	Mg ($cmol_c/kg$)	Na ($cmol_c/kg$)	Fe ($cmol_c/kg$)	K ($cmol_c/kg$)
0-5	Tarcoola	Bulk soil	0.025	0.06	0.04	0.73	0.006	0.0031
5-10	Tarcoola	Bulk soil	0.036	0.05	0.04	0.87	0.007	0.0024
10-20	Tarcoola	Bulk soil	0.063	0.01	0.09	0.74	0.012	0.0033
20-30	Tarcoola	Bulk soil	0.031	0.01	0.16	0.35	0.003	0.0016
30-50	Tarcoola	Bulk soil	0.045	0.01	0.22	0.30	0.005	0.0021
0-5	Tarcoola	0	0.001	1.06	0.03	2.57	nd	0.0026
5-10	Tarcoola	0	nd	1.93	0.08	2.08	nd	0.0023
10-20	Tarcoola	0	nd	0.68	0.10	1.74	nd	0.0017
20-30	Tarcoola	0	nd	0.53	0.23	0.79	nd	0.0015
30-50	Tarcoola	0	0.001	0.13	0.11	0.53	nd	0.0014
0-5	Tarcoola	10	0.021	0.20	0.64	0.71	0.005	0.0155
5-10	Tarcoola	10	0.019	0.07	0.19	0.31	0.004	0.0063
10-20	Tarcoola	10	0.002	0.11	0.29	0.42	nd	0.0044
20-30	Tarcoola	10	0.001	0.09	0.23	0.31	nd	0.0031
30-50	Tarcoola	10	nd	0.10	0.17	0.88	nd	0.0092
0-5	Avoca	Bulk soil	0.001	2.82	1.28	1.12	0.001	0.0220
5-10	Avoca	Bulk soil	nd	3.70	0.26	0.31	nd	0.0096
10-20	Avoca	Bulk soil	nd	1.24	0.26	0.36	nd	0.0048
20-30	Avoca	Bulk soil	nd	0.58	0.16	0.21	nd	0.0025
30-50	Avoca	Bulk soil	0.001	0.37	0.32	0.64	nd	0.0105
0-5	Avoca	0	0.025	0.06	0.04	0.73	0.006	0.0031
5-10	Avoca	0	0.036	0.05	0.04	0.87	0.007	0.0024
10-20	Avoca	0	0.063	0.01	0.09	0.74	0.012	0.0033
20-30	Avoca	0	0.031	0.01	0.16	0.35	0.003	0.0016
30-50	Avoca	0	0.045	0.01	0.22	0.30	0.005	0.0021
0-5	Avoca	10	0.001	1.06	0.03	2.57	nd	0.0026
5-10	Avoca	10	nd	1.93	0.08	2.08	nd	0.0023
10-20	Avoca	10	nd	0.68	0.10	1.74	nd	0.0017
20-30	Avoca	10	nd	0.53	0.23	0.79	nd	0.0015
30-50	Avoca	10	0.001	0.13	0.11	0.53	nd	0.0014

Table C2. Exchangeable cation concentrations following addition of organic material

Depth	Site	Gypsum (t/ha)	Al ($cmol_c$/kg)	Ca ($cmol_c$/kg)	Mg ($cmol_c$/kg)	Na ($cmol_c$/kg)	Fe ($cmol_c$/kg)	K ($cmol_c$/kg)
0-5	Tarcoola	Bulk soil	nd	0.1003	0.1507	0.0343	nd	0.0094
5-10	Tarcoola	Bulk soil	nd	0.0240	0.0207	0.1254	nd	0.0022
10-20	Tarcoola	Bulk soil	nd	0.0058	0.0381	0.0757	nd	0.0023
20-30	Tarcoola	Bulk soil	nd	0.0045	0.0885	0.0678	nd	0.0024
30-50	Tarcoola	Bulk soil	nd	0.0017	0.0922	0.0582	nd	0.0024
0-5	Tarcoola	0	nd	0.0411	0.0488	0.0070	nd	0.0040
5-10	Tarcoola	0	nd	0.0176	0.0210	0.0039	nd	0.0028
10-20	Tarcoola	0	nd	0.0111	0.0110	0.0029	nd	0.0023
20-30	Tarcoola	0	nd	0.0084	0.0077	0.0023	nd	0.0021
30-50	Tarcoola	0	nd	0.1591	0.2209	0.0458	nd	0.0104
0-5	Tarcoola	10	nd	0.0005	0.0499	0.3413	nd	0.0035
5-10	Tarcoola	10	nd	0.0005	0.1081	0.2308	nd	0.0039
10-20	Tarcoola	10	nd	0.0005	0.1247	0.1403	nd	0.0041
20-30	Tarcoola	10	nd	0.0005	0.1174	0.1306	nd	0.0039
30-50	Tarcoola	10	nd	0.0005	0.0982	0.0552	nd	0.0033
0-5	Avoca	Bulk soil	nd	0.2278	0.0245	0.2692	nd	0.0034
5-10	Avoca	Bulk soil	nd	0.3175	0.1096	0.1868	nd	0.0049
10-20	Avoca	Bulk soil	nd	0.2053	0.1245	0.1281	nd	0.0045
20-30	Avoca	Bulk soil	nd	0.1741	0.1575	0.1715	nd	0.0049
30-50	Avoca	Bulk soil	nd	0.0785	0.1325	0.1026	nd	0.0045
0-5	Avoca	0	nd	0.1003	0.1507	0.0343	nd	0.0094
5-10	Avoca	0	nd	0.0240	0.0207	0.1254	nd	0.0022
10-20	Avoca	0	nd	0.0058	0.0381	0.0757	nd	0.0023
20-30	Avoca	0	nd	0.0045	0.0885	0.0678	nd	0.0024
30-50	Avoca	0	nd	0.0017	0.0922	0.0582	nd	0.0024
0-5	Avoca	10	nd	0.0411	0.0488	0.0070	nd	0.0040
5-10	Avoca	10	nd	0.0176	0.0210	0.0039	nd	0.0028
10-20	Avoca	10	nd	0.0111	0.0110	0.0029	nd	0.0023
20-30	Avoca	10	nd	0.0084	0.0077	0.0023	nd	0.0021
30-50	Avoca	10	nd	0.1591	0.2209	0.0458	nd	0.0104

APPENDIX D

Locality: **Bevendale: Property "Tarcoola"**
Scalded Profile 1
Elevation: 505 m
UTM: 0691216
6178847
Date: 19/12/05
Site Morphology: Footslopes
Vegetation: Unvegetated
Land use: Grazing

Layer	Horizon	Depth (cm)	Field pH	Colour	Texture	Mottle %	Structure	Notes
1	A1	0-1	9					Strongly versicular crust
1	A1	1-6	8	10YR 5/4 wet 10YR 7/2 dry	Loamy fine sand		Extensively layered	Variability in surface layer
1	A2	6-12	9	10YR 5/4	Loamy coarse sand		Structureless	
2	B1	12-40	9	10YR 6/3 mottled with 10YR 6/4	Fine sandy loam	< 10		Bleached, 10% coarse fraction of rounded gravel 2-5 mm
2	B1	40-80	9	10YR 6/2 mottled with 7.5YR 5/6	Light medium clay	40	Sub-angular blocky	10% coarse fraction of rounded gravel 2-5 mm
2	B1	80-100	9	7.5YR 5/6 mottled with 10YR 6/2	Light medium clay	40	Sub-angular blocky	Approximately 30 % coarse fraction of rounded gravel 2-5 mm; free water at the bottom of the pit (120 cm) with vertically bedded fractured bedrock

Scalded Profile 2

Elevation: 505 m
UTM: 0691216
6178847
Date: 19/12/05
Site Morphology: Footslopes
Vegetation: Unvegetated
Land use: Grazing

Layer	Horizon	Depth (cm)	Field pH	Colour	Texture	Mottle %	Structure	Notes
1	A1	0-12	9	10YR 4/4	Loamy fine sand		Platy	5 % coarse fraction of rounded gravel 2-5 mm
1	A2	12-23	9	10YR 4/6	Clayey sand		Structureless, massive	5 % coarse fraction of rounded gravel 2-5 mm
2	B1	23-66	9	10YR 6/8 mottled with 7.5YR 4/6	Light medium clay	30	Sub-angular blocky	10 % coarse fraction of rounded gravel 2-5 mm
2	B1	66-100	9	7.5YR 4/6 mottled with 10YR 6/8	Silty clay loam	30	Sub-angular blocky	10 % coarse fracion of 10 % coarse fraction of rounded gravel 2-5 mm; free water at the bottom of the pit (120 cm) with vertically bedded fractured rock

Depression Profile 1
Elevation: 498 m
UTM: 0691122
6178862
Date: 19/12/05
Site Morphology: Footslopes
Vegetation: Red grass (*Botriochloa* spp), minor occurrences of tall wheat grass
Land use: Grazing

Layer	**Horizon**	**Depth (cm)**	**Field pH**	**Colour**	**Texture**	**Mottle %**	**Structure**	**Notes**
1	In fill layer	0-50	8	10YR 5/3 mottled with 7.5YR 5/6	Loamy fine sand	< 10%	Weak	Many roots; layered
2	A1	50-70	9	10YR 4/1 with rust flecks	Silty loam		Very moist at time of sampling: difficult to describe structure	Clear change from Layer 1 to Layer 2; true soil profile' charcoal at boundary between layer 1 and layer 2
2	A2	70-90	9	10YR 5/4 mottled with 7.5 YR 5/6	Sandy clay loam	40	Very moist at time of sampling: difficult to describe structure	Gradual change from horizon above; 10-20 % coarse fraction of rounded bedrock
2	B2	90-100	10	10YR 5/4 mottled with 7.5YR 5/6	Light medium clay	40	Very moist at time of sampling: difficult to describe structure	Distinct change in horizon; free water at the bottom of the pit (120 cm) with vertically bedded fractured rock

Depression Profile 2

Elevation: 498 m
UTM: 0691122
6178862
Date: 19/12/05
Site Morphology: Footslopes
Vegetation: Red grass (*Botriochloa* spp), minor occurrences of tall wheat grass
Land use: Grazing

Layer	**Horizon**	**Depth (cm)**	**Field pH**	**Colour**	**Texture**	**Mottle %**	**Structure**	**Notes**
1	In fill layer	0-60	8	10YR 5/3 mottled with 7.5YR 5/6	Loamy fine sand	< 10%	Weak	Many roots; layered
2	A1	60-90	9	10YR 5/4 mottled with 7.5YR 5/6	Sandy clay loam	40	Very moist at time of sampling: difficult to describe structure	Gradual change from horizon above; 10-20 % coarse fraction of rounded bedrock ; true soil profile
2	B2	90-100	9	10YR 5/4 mottled with 7.5YR 5/6	Light medium clay	40	Very moist at time of sampling: difficult to describe structure	Distinct change in horizon; free water at the bottom of the pit (120 cm) with vertically bedded fractured rock

Locality: **Bevendale: Property "Riverview"**

Vegetated Profile 1

Elevation: 500 m

Date: 08/02/06

Site Morphology: Gully wall

Vegetation: Wallaby grass

Land use: Fenced from stock

Layer	Horizon	Depth (cm)	Field pH	Colour	Texture	Mottle %	Structure	Notes
1	A1	0-3	4	10YR 4/3	Sandy loam		Well structured: sub-angular blocky	Extensive roots
1	A2	3-12	5.5	10YR 5/4	Silty loam		Sub-angular blocky	Bleached; fine roots present; minor occurrences of charcoal and gravel (<5mm)
2	B1	12-22	6	10YR 4/4	Silty loam	Minor	Sub-angular blocky	Fine roots present
2	B1	22-34	6.5	10 YR 4/4 mottled with 10 YR 5/6	Silty loam	30	Sub-angular blocky	Fine roots present; minor occurrences of charcoal and gravel (< 5mm)
2	B2	34-50	7	10YR 6/4	Silty loam		Versicular	

Vegetated Profile 2
Elevation: 500 m
Date: 08/02/06
Site Morphology: Gully wall
Vegetation: Wallaby grass
Land use: Fenced from stock

Layer	Horizon	Depth (cm)	Field pH	Colour	Texture	Mottle %	Structure	Notes
1	A1	0-5	4	10YR 4/3	Sandy loam		Well structured: sub-angular blocky	Extensive roots
1	A2	5-14	5.5	10YR 5/4	Silty loam		Sub-angular blocky	Bleached; fine roots present; minor occurrences of charcoal and gravel (<5mm)
2	B1	14-23	6	10YR 4/4	Silty loam	Minor	Sub-angular blocky	Fine roots present
2	B1	23-26	7	10 YR 4/4 mottled with 10 YR 5/6	Gravel layer	30		Fine roots present; gravel layer
2	B2	26-50	8	10YR 6/4	Silty loam		Versicular	30-40 % large gravel

Locality: **Rugby: Property "Gunyah"**

Pasture Profile 1

Elevation: 547 m

UTM: 0685901
6182381

Date: 24/01/06

Site Morphology: Plain

Vegetation: Revegetated with tall wheat grass, minor occurrences of Wallaby Grass and Couch

Land use: Sheep grazing

Layer	Horizon	Depth (cm)	Field pH	Colour	Texture	Mottle %	Structure	Notes
1	A1	0-8	7	5YR 4/2	Silty loam		Sub-angular blocky	Extensive roots
1	A2	8-35	6.5	7.5YR 6/4	Silty loam		Weak	Roots present, bleached, gradual change from horizon above
2	B1	35-60	8	5YR 7/3 mottled with 7.5 YR 6/8	Silty clay loam	40	Sub-angular blocky	Fine roots present
2	B1	60-100	8	5YR 5/8	Silty clay loam		Massive	Fine roots present

Pasture Profile 2

Elevation: 547 m
UTM: 0685901
6182381
Date: 24/01/06
Site Morphology: Plain
Vegetation: Revegetated with tall wheat grass, minor occurrences of Wallaby Grass and Couch
Land use: Sheep grazing

Layer	**Horizon**	**Depth (cm)**	**Field pH**	**Colour**	**Texture**	**Mottle %**	**Structure**	**Notes**
1	A1	0-8	7	5YR 4/2	Silty loam		Sub-angular blocky	Extensive roots
1	A2	8-35	6.5	7.5YR 6/4	Silty loam		Weak	Roots present, bleached, gradual change from horizon above
2	B1	35-60	8	5YR 7/3 mottled with 7.5 YR 6/8	Silty clay loam	40	Sub-angular blocky	Fine roots present
2	B1	60-100	8	5YR 5/8	Silty clay loam		Massive	Fine roots present

Vegetated Profile 1

Elevation: 543 m
UTM: 0685908
6182497
Date: 24/01/06
Site Morphology: Plain
Vegetation: Wallaby grass with minor occurrences of Couch
Land use: Fenced from stock

Layer	Horizon	Depth (cm)	Field pH	Colour	Texture	Mottle %	Structure	Notes
1	A1	0-5	5.5	7.5YR 5/2	Loam		Weak	Extensive roots
1	A2	5-20	5	7.5 YR 6/3	Loam		Sub-angular blocky	Bleached, gradual change from horizon above
1	A2	20-45	6	7.5YR 7/1	Silty loam		Sub-angular blocky	Bleached, fine roots present, charcoal present
2	B1	45-55	6.5	10YR 8/1 mottled with 10YR 5/6	Silty loam	20	Sub-angular blocky	Bleached, charcoal present
2	B2	55-100	7	5YR 5/8	Light clay		Massive	

Vegetated Profile 2

Elevation: 543 m
UTM: 0685908
6182497
Date: 25/01/06
Site Morphology: Plain
Vegetation: Wallaby Grass with minor occurrences of Couch
Land use: Fenced from stock

Layer	Horizon	Depth (cm)	Field pH	Colour	Texture	Mottle %	Structure	Notes
1	A1	0-10	5.5	7.5YR 3/3	Loam		Weak	Extensive roots
1	A2	10-23	6	7.5 YR 6/4	Loam		Sub-angular blocky	Bleached, gradual change from horizon above
1	A2	23-59	6	7.5YR 7/2	Silty loam		Sub-angular blocky	Bleached, fine roots present, charcoal present
2	B1	59-70	8	7.5YR 7/2 mottled with 7.5YR 6/8	Silty loam	20	Sub-angular blocky	Bleached, charcoal present
2	B2	70-100	8.5	5YR 5/8	Light clay		Massive	

Eroded Profile 1

Elevation: 544 m
UTM: 0685890
6182507
Date: 25/01/06
Site Morphology: Plain
Vegetation: Unvegetated
Land use: Fenced from stock

Layer	Horizon	Depth (cm)	Field pH	Colour	Texture	Mottle %	Structure	Notes
1	A2	0-10	8.5	5YR 5/2	Sandy clay loam		Platy; versicular crust	Black surface crust
1	A2	10-30	8.5	5YR 6/2	Sandy clay loam		Massive	Bleached, charcoal present, fine roots present
2	B1	30-48	9	7.5YR 6/1 mottled with 7.5YR 6/8	Light clay	40	Massive	Coarse fraction of rounded gravel and quartz <5mm
2	B2	48-100	9	7.5YR 5/8 mottled with 7.5YR 6/1	Light medium clay	30	Moist	Free water at the bottom of the pit (110cm)

Eroded Profile 2

Elevation: 544 m
UTM: 0685890
6182507
Date: 25/01/06
Site Morphology: Plain
Vegetation: Unvegetated
Land use: Fenced from stock

Layer	Horizon	Depth (cm)	Field pH	Colour	Texture	Mottle %	Structure	Notes
1	A2	0-13	8.5	7.5YR 7/1	Sandy clay loam		Platy; versicular crust	Black surface crust
1	A2	13-28	9	7.5YR 5/4	Sandy clay loam		Massive	Bleached, charcoal present, fine roots present, coarse fraction of rounded gravel and quartz <5mm
2	B1	28-57	9	10YR 7/4 mottled with 5YR 6/8	Light clay	40	Massive	Coarse fraction of rounded gravel and quartz <5mm
2	B2	57-100	9	7.5YR 6/8 mottled with 5YR 5/8	Light medium clay	30	Moist	Free water at the bottom of the pit (110cm)

Scalded Profile 1

Elevation: 542 m
UTM: 0685879
6182510
Date: 25/01/06
Site Morphology: Plain
Vegetation: Unvegetated
Land use: Fenced from stock

Layer	Horizon	Depth (cm)	Field pH	Colour	Texture	Mottle %	Structure	Notes
1	A1	0-3	8.5	10YR 5/2	Loam		Weak	Depositional material, fine roots present
1	A2	3-20	9	10YR 6/2	Loam		Weak	Charcoal present, fine roots present
2	B1	20-41	9	10YR 6/1 mottled with 10YR 6/4	Clay loam	30	Weak	30% coarse fraction of rounded gravel < 5mm
2	B2	41-100	9	7.5YR 5/8 mottled with 5YR 7/1	Light medium clay	30	Moist	10% coarse fraction of rounded gravel < 5mm, free water at the bottom of the pit (110cm)

Scalded Profile 2

Elevation: 542 m
UTM: 0685879
6182510
Date: 25/01/06
Site Morphology: Plain
Vegetation: Unvegetated
Land use: Fenced from stock

Layer	Horizon	Depth (cm)	Field pH	Colour	Texture	Mottle %	Structure	Notes
1	A1	0-5	9	10YR 4/2	Loam		Weak	Depositional material, fine roots present
1	A2	5-28	9	7.5YR 5/2	Loam		Weak	Charcoal present, fine roots present
2	B1	28-41	9	7.5YR 6/1 mottled with 7.5YR 5/6	Clay loam	30	Weak	30% coarse fraction of rounded gravel < 5mm
2	B2	41-100	9	7.5YR 5/6 mottled with 2.5YR 4/8	Light medium clay	30	Moist	10% coarse fraction of rounded gravel < 5mm, free water at the bottom of the pit (110cm)

Table D1 Particle size distribution from each depth at each microsite and site

Site	Microsite	Profile Number	Depth (cm)	Sand (%)	Silt (%)	Clay (%)	Total
Gunyah	Eroded	1	0-5	65.29	11.79	17.89	94.96
Gunyah	Eroded	2	0-5	78.37	22.02	14.32	114.71
Gunyah	Eroded	1	5-10	66.48	17.77	19.86	104.11
Gunyah	Eroded	2	5-10	66.49	26.31	22.45	115.25
Gunyah	Eroded	1	10-20	65.18	21.21	12.51	98.90
Gunyah	Eroded	2	10-20	64.87	23.86	23.96	112.69
Gunyah	Eroded	1	20-30	66.69	23.18	24.48	114.35
Gunyah	Eroded	2	20-30	62.90	24.39	26.47	113.76
Gunyah	Eroded	1	30-50	58.12	22.81	33.62	114.56
Gunyah	Eroded	2	30-50	53.77	20.55	36.61	110.93
Gunyah	Eroded	1	50-70	44.58	22.38	45.86	112.83
Gunyah	Eroded	2	50-70	29.68	20.57	58.62	108.87
Gunyah	Eroded	1	70-100	27.99	28.93	53.77	110.69
Gunyah	Eroded	2	70-100	29.84	30.41	52.38	112.62
Gunyah	Pasture	1	0-5	68.66	15.60	18.46	102.73
Gunyah	Pasture	2	0-5	73.60	13.79	13.89	101.28
Gunyah	Pasture	1	5-10	69.32	17.80	18.70	105.82
Gunyah	Pasture	2	5-10	72.91	13.71	17.79	104.41
Gunyah	Pasture	1	10-20	69.31	19.53	20.41	109.25
Gunyah	Pasture	2	10-20	70.24	15.85	17.96	104.05
Gunyah	Pasture	1	20-30	68.17	25.14	18.22	111.54
Gunyah	Pasture	2	20-30	72.57	21.79	17.89	112.26
Gunyah	Pasture	1	30-50	62.76	17.69	26.54	107.00
Gunyah	Pasture	2	30-50	72.84	20.49	19.39	112.72
Gunyah	Pasture	1	50-70	67.40	15.68	28.49	111.57
Gunyah	Pasture	2	50-70	63.29	14.40	31.29	108.98
Gunyah	Scald	1	0-5	68.73	25.91	13.05	107.70
Gunyah	Scald	2	0-5	61.79	29.75	15.27	106.81
Gunyah	Scald	1	5-10	68.24	24.37	19.27	111.88
Gunyah	Scald	2	5-10	59.96	17.63	17.19	94.78
Gunyah	Scald	1	10-20	66.32	24.40	23.30	114.02
Gunyah	Scald	2	10-20	74.36	25.72	23.23	123.31
Gunyah	Scald	1	20-30	67.86	22.36	23.26	113.48
Gunyah	Scald	2	20-30	64.79	21.11	25.19	111.09
Gunyah	Scald	1	30-50	56.07	18.36	35.23	109.66
Gunyah	Scald	2	30-50	61.18	21.80	31.30	114.28
Gunyah	Scald	1	50-70	29.95	20.36	45.22	95.53
Gunyah	Scald	2	50-70	59.64	10.36	47.11	117.11
Gunyah	Scald	1	70-100	35.80	26.38	51.27	113.45
Gunyah	Scald	2	70-100	34.85	19.82	55.34	110.01
Gunyah	Vegetated	1	0-5	78.10	22.60	16.70	117.40
Gunyah	Vegetated	2	0-5	81.67	11.19	18.69	111.55
Gunyah	Vegetated	1	5-10	77.65	21.19	14.69	113.53
Gunyah	Vegetated	2	5-10	81.08	18.66	16.75	116.49
Gunyah	Vegetated	1	10-20	79.72	22.55	16.67	118.94
Gunyah	Vegetated	2	10-20	82.71	18.53	16.63	117.87
Gunyah	Vegetated	1	20-30	73.19	21.73	17.98	112.90
Gunyah	Vegetated	2	20-30	82.55	18.65	18.75	119.96
Gunyah	Vegetated	1	30-50	75.14	23.94	18.20	117.29
Gunyah	Vegetated	2	30-50	82.90	23.20	18.70	124.80
Gunyah	Vegetated	1	50-70	67.71	20.61	22.71	111.02
Gunyah	Vegetated	2	50-70	28.84	30.68	47.34	106.86

Gunyah	Vegetated	1	70-100	23.23	46.19	23.47	92.89
Gunyah	Vegetated	2	70-100	31.24	14.61	52.74	98.60
Tarcoola	Vegetated	1	0-5	66.22	25.95	21.42	113.60
Tarcoola	Vegetated	2	0-5	63.15	20.40	23.90	107.46
Tarcoola	Vegetated	1	5-10	69.77	23.80	23.30	116.86
Tarcoola	Vegetated	2	5-10	66.94	29.96	19.61	116.52
Tarcoola	Vegetated	1	10-20	59.82	28.36	26.66	114.84
Tarcoola	Vegetated	2	10-20	66.46	30.38	27.88	124.73
Tarcoola	Vegetated	1	20-30	62.91	19.66	29.09	111.66
Tarcoola	Vegetated	2	20-30	39.05	25.03	27.93	92.01
Tarcoola	Vegetated	1	30-50	76.28	13.78	21.26	111.32
Tarcoola	Vegetated	2	30-50	69.33	20.43	23.94	113.70
Tarcoola	Scald	1	0-5	89.30	11.39	10.72	111.40
Tarcoola	Scald	2	0-5	93.95	5.19	10.68	109.82
Tarcoola	Scald	1	5-10	88.95	6.57	15.24	110.75
Tarcoola	Scald	2	5-10	87.61	7.17	15.23	110.02
Tarcoola	Scald	1	10-20	83.46	15.82	14.58	113.86
Tarcoola	Scald	2	10-20	79.73	9.09	21.63	110.45
Tarcoola	Scald	1	20-30	76.00	16.38	19.04	111.43
Tarcoola	Scald	2	20-30	68.57	18.78	25.89	113.24
Tarcoola	Scald	1	30-50	63.50	16.54	31.19	111.24
Tarcoola	Scald	2	30-50	56.21	15.77	37.84	109.82
Tarcoola	Scald	1	50-70	37.23	18.94	52.57	108.74
Tarcoola	Scald	2	50-70	40.87	19.99	49.78	110.64
Tarcoola	Scald	1	70-100	47.12	20.55	43.20	110.88
Tarcoola	Scald	2	70-100	51.09	2.18	51.47	104.74
Tarcoola	Depression	1	0-5	83.59	15.19	23.89	122.67
Tarcoola	Depression	2	0-5	72.05	14.61	21.31	107.96
Tarcoola	Depression	1	5-10	62.94	25.59	27.08	115.61
Tarcoola	Depression	2	5-10	69.08	15.94	27.19	112.21
Tarcoola	Depression	1	10-20	61.44	21.81	29.31	112.57
Tarcoola	Depression	2	10-20	57.63	28.10	30.75	116.49
Tarcoola	Depression	1	20-30	56.56	25.80	33.30	115.66
Tarcoola	Depression	2	20-30	56.56	26.48	32.55	115.59
Tarcoola	Depression	1	30-50	71.01	13.79	27.29	112.10
Tarcoola	Depression	2	30-50	49.39	30.93	36.38	116.69
Tarcoola	Depression	1	50-70	61.95	17.90	31.47	111.32
Tarcoola	Depression	2	50-70	40.10	28.39	40.40	108.89
Tarcoola	Depression	1	70-100	59.51	17.55	36.79	113.85
Tarcoola	Depression	2	70-100	53.31	26.42	35.05	114.78

Table D2 Soluble cation concentrations for each sample. Nd indicates concentration was below the detection limit

Site	Description	Profile	Depth	Rep	Al ($cmol_c/kg$)	B ($cmol_c/kg$)	Fe ($cmol_c/kg$)	Mn ($cmol_c/kg$)	Ca ($cmol_c/kg$)	K ($cmol_c/kg$)	Mg ($cmol_c/kg$)	Na ($cmol_c/kg$)
Tarcoola	Scald	1	2.5	1	0.083	nd	0.014	nd	0.000	0.004	0.051	0.160
Tarcoola	Scald	1	2.5	2	0.094	nd	0.015	nd	0.001	0.004	0.055	0.293
Tarcoola	Scald	1	2.5	3	0.151	0.004	0.034	nd	0.001	0.005	0.072	0.246
Tarcoola	Scald	1	7.5	1	0.212	nd	0.053	nd	0.001	0.006	0.122	0.204
Tarcoola	Scald	1	7.5	2	0.216	0.004	0.036	nd	0.001	0.005	0.175	0.217
Tarcoola	Scald	1	7.5	3	0.565	0.017	0.100	0.000	0.018	0.011	0.459	0.297
Tarcoola	Scald	1	15	1	0.146	nd	0.027	nd	0.000	0.005	0.114	0.344
Tarcoola	Scald	1	15	2	0.280	nd	0.049	nd	0.002	0.007	0.190	0.358
Tarcoola	Scald	1	15	3	0.328	0.004	0.039	nd	0.002	0.006	0.106	0.332
Tarcoola	Scald	1	25	1	0.101	nd	0.016	nd	0.000	0.003	0.011	0.270
Tarcoola	Scald	1	25	2	0.209	nd	0.034	nd	0.000	0.004	0.047	0.310
Tarcoola	Scald	1	25	3	0.103	nd	0.020	nd	0.000	0.003	0.011	0.267
Tarcoola	Scald	1	40	1	0.390	nd	0.090	nd	0.000	0.006	0.142	0.375
Tarcoola	Scald	1	40	2	0.336	nd	0.058	nd	0.000	0.006	0.173	0.406
Tarcoola	Scald	1	40	3	0.294	nd	0.048	nd	0.000	0.005	0.083	0.299
Tarcoola	Scald	1	60	1	0.325	nd	0.033	nd	0.009	0.008	1.378	1.052
Tarcoola	Scald	1	60	2	0.696	nd	0.025	nd	0.001	0.009	1.616	1.438
Tarcoola	Scald	1	60	3	0.108	nd	0.000	nd	0.001	0.005	1.114	1.466
Tarcoola	Scald	1	85	1	0.495	nd	0.105	nd	0.000	0.007	0.325	0.464
Tarcoola	Scald	1	85	2	0.222	0.010	0.026	nd	0.004	0.004	0.145	0.283
Tarcoola	Scald	1	85	3	0.257	nd	0.033	nd	0.000	0.005	0.213	0.331
Tarcoola	Scald	2	2.5	1	0.081	nd	0.012	nd	0.001	0.003	0.062	0.119
Tarcoola	Scald	2	2.5	2	0.123	nd	0.021	nd	0.000	0.004	0.069	0.117
Tarcoola	Scald	2	2.5	3	0.045	nd	0.008	nd	0.000	0.003	0.019	0.074
Tarcoola	Scald	2	7.5	1	0.184	nd	0.026	nd	0.002	0.006	0.180	0.252
Tarcoola	Scald	2	7.5	2	0.201	nd	0.032	nd	0.000	0.005	0.145	0.273
Tarcoola	Scald	2	7.5	3	0.306	nd	0.040	nd	0.003	0.007	0.362	0.281
Tarcoola	Scald	2	15	1	0.248	0.014	0.030	nd	0.007	0.004	0.046	0.254
Tarcoola	Scald	2	15	2	0.313	nd	0.039	nd	0.002	0.008	0.169	0.430
Tarcoola	Scald	2	15	3	0.345	nd	0.052	nd	0.001	0.008	0.175	0.402
Tarcoola	Scald	2	25	1	0.218	nd	0.044	nd	0.000	0.006	0.047	0.274
Tarcoola	Scald	2	25	2	0.243	nd	0.043	nd	0.000	0.005	0.045	0.317
Tarcoola	Scald	2	25	3	0.248	nd	0.064	nd	0.000	0.005	0.047	0.273
Tarcoola	Scald	2	40	1	0.424	nd	0.030	nd	0.000	0.009	0.412	0.598
Tarcoola	Scald	2	40	2	0.158	nd	0.022	nd	0.000	0.004	0.139	0.383
Tarcoola	Scald	2	40	3	0.026	nd	0.000	nd	0.031	0.007	1.011	2.101
Tarcoola	Scald	2	60	1	0.648	nd	0.014	nd	0.002	0.009	0.813	0.796
Tarcoola	Scald	2	60	2	0.113	nd	0.003	nd	0.003	0.005	1.201	1.349
Tarcoola	Scald	2	60	3	0.148	nd	0.002	nd	0.001	0.007	1.349	1.463
Tarcoola	Scald	2	85	1	0.280	nd	0.033	nd	0.000	0.005	0.173	0.323
Tarcoola	Scald	2	85	2	0.198	nd	0.025	nd	0.000	0.003	0.120	0.260
Tarcoola	Scald	2	85	3	0.290	nd	0.033	nd	0.000	0.005	0.218	0.365
Tarcoola	Depression	1	2.5	1	0.006	0.005	0.015	0.003	0.030	0.017	0.218	0.055
Tarcoola	Depression	1	2.5	2	0.006	0.004	0.015	0.005	0.031	0.012	0.217	0.057
Tarcoola	Depression	1	2.5	3	0.021	nd	0.009	0.000	0.009	0.005	0.119	0.054
Tarcoola	Depression	1	7.5	1	0.005	nd	0.019	0.004	0.027	0.013	0.210	0.046
Tarcoola	Depression	1	7.5	2	0.029	nd	0.011	0.001	0.009	0.004	0.106	0.052

Tarcoola	Depression	1	7.5	3	0.032	nd	0.013	0.001	0.010	0.004	0.113	0.065
Tarcoola	Depression	1	15	1	0.035	nd	0.013	0.001	0.008	0.004	0.105	0.094
Tarcoola	Depression	1	15	2	0.026	nd	0.009	0.002	0.008	0.003	0.107	0.082
Tarcoola	Depression	1	15	3	0.018	nd	0.006	0.002	0.013	0.004	0.142	0.063
Tarcoola	Depression	1	25	1	0.054	nd	0.015	0.001	0.003	0.003	0.056	0.104
Tarcoola	Depression	1	25	2	0.034	nd	0.011	0.000	0.004	0.003	0.066	0.101
Tarcoola	Depression	1	25	3	0.035	nd	0.012	0.001	0.006	0.003	0.087	0.119
Tarcoola	Depression	1	40	1	0.118	nd	0.028	0.001	0.006	0.004	0.095	0.201
Tarcoola	Depression	1	40	2	0.217	nd	0.079	0.001	0.006	0.005	0.098	0.179
Tarcoola	Depression	1	40	3	0.134	nd	0.028	0.000	0.006	0.004	0.083	0.213
Tarcoola	Depression	1	60	1	0.080	nd	0.017	0.000	0.009	0.003	0.099	0.430
Tarcoola	Depression	1	60	2	0.110	nd	0.027	0.000	0.009	0.003	0.106	0.365
Tarcoola	Depression	1	60	3	0.099	nd	0.023	0.000	0.006	0.003	0.095	0.321
Tarcoola	Depression	1	85	1	0.087	nd	0.011	0.001	0.004	0.003	0.074	0.385
Tarcoola	Depression	1	85	2	0.084	nd	0.013	0.002	0.006	0.004	0.079	0.338
Tarcoola	Depression	1	85	3	0.108	0.011	0.013	0.002	0.016	0.004	0.097	0.367
Tarcoola	Depression	2	2.5	1	0.004	0.006	0.024	0.005	0.035	0.013	0.249	0.111
Tarcoola	Depression	2	2.5	2	0.008	nd	0.020	0.004	0.030	0.012	0.218	0.099
Tarcoola	Depression	2	2.5	3	0.003	nd	0.023	0.005	0.038	0.017	0.258	0.106
Tarcoola	Depression	2	7.5	1	0.013	0.005	0.005	0.003	0.013	0.005	0.133	0.091
Tarcoola	Depression	2	7.5	2	0.153	0.005	0.025	0.001	0.000	0.004	0.048	0.117
Tarcoola	Depression	2	7.5	3	0.025	nd	0.011	0.002	0.011	0.005	0.131	0.094
Tarcoola	Depression	2	15	1	0.017	nd	0.006	0.001	0.007	0.004	0.103	0.080
Tarcoola	Depression	2	15	2	0.051	nd	0.021	0.001	0.005	0.004	0.081	0.070
Tarcoola	Depression	2	15	3	0.048	nd	0.019	0.001	0.002	0.003	0.073	0.070
Tarcoola	Depression	2	25	1	0.074	nd	0.041	0.001	0.004	0.004	0.063	0.067
Tarcoola	Depression	2	25	2	0.048	nd	0.015	0.001	0.003	0.004	0.067	0.089
Tarcoola	Depression	2	25	3	0.055	nd	0.021	0.001	0.003	0.005	0.070	0.088
Tarcoola	Depression	2	40	1	0.054	nd	0.015	0.001	0.002	0.004	0.072	0.080
Tarcoola	Depression	2	40	2	0.079	nd	0.034	0.002	0.001	0.004	0.048	0.064
Tarcoola	Depression	2	40	3	0.107	nd	0.033	0.002	0.002	0.004	0.068	0.090
Tarcoola	Depression	2	60	1	0.051	nd	0.006	0.000	0.002	0.003	0.060	0.142
Tarcoola	Depression	2	60	2	0.067	nd	0.010	0.000	0.002	0.003	0.079	0.148
Tarcoola	Depression	2	60	3	0.049	nd	0.007	0.000	0.004	0.003	0.092	0.136
Tarcoola	Depression	2	85	1	0.188	0.018	0.027	0.001	0.007	0.004	0.051	0.141
Tarcoola	Depression	2	85	2	0.168	nd	0.021	0.001	0.001	0.005	0.062	0.132
Tarcoola	Depression	2	85	3	0.104	nd	0.029	0.001	0.000	0.004	0.061	0.092
Gunyah	Pasture	1	2.5	1	0.004	nd	0.006	0.001	0.186	0.016	0.026	0.028
Gunyah	Pasture	1	2.5	2	0.005	nd	0.006	0.001	0.145	0.012	0.018	0.016
Gunyah	Pasture	1	2.5	3	0.009	nd	0.013	0.001	0.083	0.010	0.014	0.015
Gunyah	Pasture	1	7.5	1	0.016	0.005	0.007	0.000	0.012	0.003	0.007	0.009
Gunyah	Pasture	1	7.5	2	0.013	nd	0.001	nd	0.015	0.003	0.000	0.007
Gunyah	Pasture	1	7.5	3	0.014	nd	0.012	0.000	0.017	0.004	0.001	0.016
Gunyah	Pasture	1	15	1	0.016	nd	0.002	nd	0.007	0.003	0.000	0.012
Gunyah	Pasture	1	15	2	0.017	nd	0.002	nd	0.006	0.002	0.000	0.005
Gunyah	Pasture	1	15	3	0.017	nd	0.001	nd	0.008	0.002	0.000	0.007
Gunyah	Pasture	1	25	1	0.010	nd	0.000	nd	0.000	0.002	0.000	0.017
Gunyah	Pasture	1	25	2	0.011	nd	0.000	nd	0.000	0.002	0.000	0.017
Gunyah	Pasture	1	25	3	0.010	nd	0.000	nd	0.000	0.002	0.000	0.013
Gunyah	Pasture	1	40	1	0.124	nd	0.115	nd	0.007	0.003	0.053	0.151
Gunyah	Pasture	1	40	2	0.187	nd	0.144	nd	0.016	0.004	0.107	0.131
Gunyah	Pasture	1	40	3	0.094	0.005	0.094	nd	0.004	0.003	0.038	0.139

Gunyah	Pasture	1	60	1	0.099	nd	0.028	nd	0.021	0.003	0.070	0.443
Gunyah	Pasture	1	60	2	0.054	nd	0.014	nd	0.000	0.003	0.027	0.166
Gunyah	Pasture	1	60	3	0.125	nd	0.039	nd	0.001	0.002	0.094	0.524
Gunyah	Pasture	2	2.5	1	0.004	nd	0.002	0.000	0.219	0.006	0.013	0.011
Gunyah	Pasture	2	2.5	2	0.007	nd	0.004	0.001	0.147	0.011	0.013	0.007
Gunyah	Pasture	2	2.5	3	0.006	nd	0.007	0.000	0.195	0.005	0.015	0.014
Gunyah	Pasture	2	7.5	1	0.017	nd	0.018	nd	0.026	0.003	0.005	0.005
Gunyah	Pasture	2	7.5	2	0.015	nd	0.010	nd	0.027	0.003	0.003	0.010
Gunyah	Pasture	2	7.5	3	0.012	nd	0.005	nd	0.012	0.003	0.000	0.008
Gunyah	Pasture	2	15	1	0.019	nd	0.030	nd	0.021	0.004	0.007	0.006
Gunyah	Pasture	2	15	2	0.013	nd	0.031	nd	0.014	0.003	0.000	0.006
Gunyah	Pasture	2	15	3	0.010	nd	0.010	nd	0.008	0.004	0.003	0.013
Gunyah	Pasture	2	25	1	0.010	nd	0.009	nd	0.003	0.002	0.004	0.007
Gunyah	Pasture	2	25	2	0.016	nd	0.032	nd	0.005	0.003	0.001	0.006
Gunyah	Pasture	2	25	3	0.017	nd	0.032	nd	0.010	0.003	0.000	0.007
Gunyah	Pasture	2	40	1	0.006	nd	0.001	nd	0.003	0.002	0.000	0.008
Gunyah	Pasture	2	40	2	0.007	nd	0.000	nd	0.000	0.002	0.001	0.009
Gunyah	Pasture	2	40	3	0.009	0.004	0.001	nd	0.000	0.002	0.000	0.010
Gunyah	Pasture	2	60	1	0.078	nd	0.053	nd	0.000	0.002	0.041	0.173
Gunyah	Pasture	2	60	2	0.129	nd	0.087	nd	0.000	0.004	0.050	0.170
Gunyah	Pasture	2	60	3	0.199	nd	0.175	nd	0.000	0.004	0.102	0.213
Gunyah	Vegetated	1	2.5	1	0.018	nd	0.080	0.006	0.041	0.007	0.043	0.023
Gunyah	Vegetated	1	2.5	2	0.011	nd	0.059	0.005	0.033	0.010	0.036	0.023
Gunyah	Vegetated	1	2.5	3	0.013	0.005	0.059	0.007	0.038	0.012	0.060	0.020
Gunyah	Vegetated	1	7.5	1	0.019	nd	0.063	0.007	0.027	0.004	0.018	0.021
Gunyah	Vegetated	1	7.5	2	0.010	nd	0.014	0.002	0.009	0.002	0.006	0.009
Gunyah	Vegetated	1	7.5	3	0.016	nd	0.040	0.005	0.021	0.003	0.013	0.021
Gunyah	Vegetated	1	15	1	0.010	nd	0.003	0.002	0.004	0.002	0.000	0.020
Gunyah	Vegetated	1	15	2	0.009	nd	0.000	0.001	0.005	0.002	0.002	0.016
Gunyah	Vegetated	1	15	3	0.010	0.005	0.000	0.001	0.006	0.002	0.004	0.013
Gunyah	Vegetated	1	25	1	0.018	nd	0.002	nd	0.000	0.002	0.000	0.015
Gunyah	Vegetated	1	25	2	0.020	nd	0.003	nd	0.001	0.002	0.003	0.013
Gunyah	Vegetated	1	25	3	0.021	nd	0.004	nd	0.000	0.002	0.002	0.015
Gunyah	Vegetated	1	40	1	0.024	nd	0.008	nd	0.000	0.002	0.000	0.024
Gunyah	Vegetated	1	40	2	0.033	nd	0.008	nd	0.000	0.002	0.000	0.022
Gunyah	Vegetated	1	40	3	0.022	nd	0.009	nd	0.000	0.002	0.002	0.018
Gunyah	Vegetated	1	60	1	0.059	nd	0.046	nd	0.000	0.003	0.035	0.159
Gunyah	Vegetated	1	60	2	0.077	nd	0.064	nd	0.000	0.003	0.053	0.176
Gunyah	Vegetated	1	60	3	0.045	nd	0.041	nd	0.000	0.002	0.022	0.115
Gunyah	Vegetated	1	85	1	0.423	nd	0.109	nd	0.003	0.006	0.543	0.703
Gunyah	Vegetated	1	85	2	0.026	nd	0.002	nd	0.001	0.003	0.040	0.931
Gunyah	Vegetated	1	85	3	0.134	nd	0.030	nd	0.001	0.005	0.292	0.495
Gunyah	Vegetated	2	2.5	1	0.016	nd	0.036	0.002	0.025	0.004	0.032	0.039
Gunyah	Vegetated	2	2.5	2	0.017	nd	0.046	0.002	0.041	0.006	0.058	0.083
Gunyah	Vegetated	2	2.5	3	0.023	nd	0.061	0.002	0.088	0.003	0.162	0.273
Gunyah	Vegetated	2	7.5	1	0.026	nd	0.006	0.001	0.051	0.002	0.115	0.310
Gunyah	Vegetated	2	7.5	2	0.014	nd	0.036	0.003	0.030	0.003	0.083	0.162
Gunyah	Vegetated	2	7.5	3	0.011	0.005	0.005	0.001	0.006	0.003	0.021	0.024
Gunyah	Vegetated	2	15	1	0.014	nd	0.003	nd	0.000	0.002	0.002	0.109
Gunyah	Vegetated	2	15	2	0.015	nd	0.002	0.001	0.011	0.002	0.030	0.194
Gunyah	Vegetated	2	15	3	0.016	nd	0.002	0.001	0.007	0.002	0.016	0.102
Gunyah	Vegetated	2	25	1	0.016	nd	0.003	nd	0.000	0.002	0.000	0.058

Gunyah	Vegetated	2	25	2	0.017	nd	0.005	nd	0.000	0.002	0.000	0.057
Gunyah	Vegetated	2	25	3	0.016	nd	0.002	nd	0.003	0.002	0.019	0.190
Gunyah	Vegetated	2	40	1	0.013	nd	0.000	nd	0.000	0.002	0.000	0.058
Gunyah	Vegetated	2	40	2	0.070	0.012	0.000	nd	0.018	0.002	0.088	0.351
Gunyah	Vegetated	2	40	3	0.013	nd	0.000	nd	0.000	0.002	0.006	0.129
Gunyah	Vegetated	2	60	1	0.124	nd	0.085	nd	0.001	0.003	0.137	0.428
Gunyah	Vegetated	2	60	2	0.144	0.009	0.076	nd	0.011	0.003	0.156	0.420
Gunyah	Vegetated	2	60	3	0.106	nd	0.060	nd	0.001	0.003	0.142	0.421
Gunyah	Vegetated	2	85	1	0.093	nd	0.014	0.000	0.001	0.002	0.213	0.653
Gunyah	Vegetated	2	85	2	0.203	nd	0.063	nd	0.001	0.003	0.219	0.702
Gunyah	Vegetated	2	85	3	0.165	nd	0.040	nd	0.001	0.002	0.235	0.693
Gunyah	Eroded	1	2.5	1	0.066	nd	0.014	nd	0.001	0.001	0.063	0.329
Gunyah	Eroded	1	2.5	2	0.054	nd	0.012	nd	0.000	0.002	0.039	0.204
Gunyah	Eroded	1	2.5	3	0.044	nd	0.009	nd	0.000	0.001	0.047	0.215
Gunyah	Eroded	1	7.5	1	0.053	nd	0.011	nd	0.000	0.002	0.035	0.160
Gunyah	Eroded	1	7.5	2	0.030	nd	0.008	nd	0.000	0.003	0.030	0.207
Gunyah	Eroded	1	7.5	3	0.064	nd	0.012	nd	0.001	0.002	0.048	0.159
Gunyah	Eroded	1	15	1	0.072	nd	0.024	nd	0.000	0.002	0.030	0.135
Gunyah	Eroded	1	15	2	0.058	0.020	0.009	nd	0.007	0.001	0.026	0.126
Gunyah	Eroded	1	15	3	0.078	0.012	0.022	nd	0.003	0.002	0.032	0.119
Gunyah	Eroded	1	25	1	0.093	nd	0.034	nd	0.001	0.001	0.045	0.178
Gunyah	Eroded	1	25	2	0.068	nd	0.034	nd	0.000	0.002	0.037	0.096
Gunyah	Eroded	1	25	3	0.088	nd	0.042	nd	0.000	0.002	0.054	0.094
Gunyah	Eroded	1	40	1	0.200	nd	0.086	nd	0.001	0.005	0.138	0.153
Gunyah	Eroded	1	40	2	0.254	nd	0.109	nd	0.003	0.006	0.203	0.134
Gunyah	Eroded	1	40	3	0.113	nd	0.031	nd	0.000	0.003	0.130	0.132
Gunyah	Eroded	1	60	1	0.270	nd	0.066	nd	0.000	0.004	0.171	0.173
Gunyah	Eroded	1	60	2	0.255	nd	0.073	nd	0.000	0.004	0.134	0.151
Gunyah	Eroded	1	60	3	0.234	nd	0.066	nd	0.000	0.004	0.135	0.154
Gunyah	Eroded	1	85	1	0.274	nd	0.071	nd	0.000	0.004	0.161	0.170
Gunyah	Eroded	1	85	2	0.206	nd	0.055	nd	0.000	0.003	0.097	0.177
Gunyah	Eroded	1	85	3	0.255	0.013	0.062	nd	0.006	0.003	0.113	0.164
Gunyah	Eroded	2	2.5	1	0.013	nd	0.005	nd	0.000	0.002	0.030	0.195
Gunyah	Eroded	2	2.5	2	0.015	nd	0.006	nd	0.001	0.002	0.040	0.783
Gunyah	Eroded	2	2.5	3	0.005	nd	0.002	nd	0.000	0.002	0.029	0.335
Gunyah	Eroded	2	7.5	1	0.010	nd	0.005	nd	0.001	0.001	0.077	0.747
Gunyah	Eroded	2	7.5	2	0.002	nd	0.001	nd	0.003	0.002	0.187	0.671
Gunyah	Eroded	2	7.5	3	0.002	nd	0.001	nd	0.001	0.002	0.106	0.759
Gunyah	Eroded	2	15	1	0.012	nd	0.005	nd	0.001	0.001	0.178	0.395
Gunyah	Eroded	2	15	2	0.010	nd	0.004	nd	0.002	0.001	0.212	0.420
Gunyah	Eroded	2	15	3	0.008	nd	0.003	nd	0.001	0.001	0.100	0.294
Gunyah	Eroded	2	25	1	0.036	nd	0.024	nd	0.000	0.002	0.041	0.152
Gunyah	Eroded	2	25	2	0.024	nd	0.013	nd	0.000	0.002	0.026	0.116
Gunyah	Eroded	2	25	3	0.029	nd	0.018	nd	0.000	0.001	0.029	0.131
Gunyah	Eroded	2	40	1	0.133	nd	0.071	nd	0.002	0.003	0.084	0.152
Gunyah	Eroded	2	40	2	0.143	nd	0.090	nd	0.001	0.003	0.093	0.145
Gunyah	Eroded	2	40	3	0.093	nd	0.041	nd	0.002	0.003	0.091	0.164
Gunyah	Eroded	2	60	1	0.198	nd	0.061	nd	0.000	0.004	0.088	0.197
Gunyah	Eroded	2	60	2	0.249	nd	0.075	nd	0.002	0.005	0.125	0.206
Gunyah	Eroded	2	60	3	0.185	nd	0.052	nd	0.002	0.003	0.104	0.205
Gunyah	Eroded	2	85	1	0.108	nd	0.027	nd	0.000	0.002	0.052	0.166
Gunyah	Eroded	2	85	2	0.129	nd	0.037	nd	0.000	0.003	0.059	0.168

Gunyah	Eroded	2	85	3	0.098	nd	0.020	nd	0.000	0.002	0.058	0.165
Gunyah	Scalded	1	2.5	1	0.003	nd	0.001	nd	0.062	0.009	0.177	0.127
Gunyah	Scalded	1	2.5	2	0.019	0.013	0.001	nd	0.033	0.011	0.097	0.071
Gunyah	Scalded	1	2.5	3	0.002	nd	0.001	nd	0.056	0.016	0.150	0.055
Gunyah	Scalded	1	7.5	1	0.014	nd	0.005	nd	0.002	0.002	0.068	0.248
Gunyah	Scalded	1	7.5	2	0.011	nd	0.006	nd	0.003	0.001	0.075	0.222
Gunyah	Scalded	1	7.5	3	0.016	nd	0.007	nd	0.000	0.002	0.057	0.166
Gunyah	Scalded	1	15	1	0.029	0.009	0.005	nd	0.003	0.001	0.076	0.220
Gunyah	Scalded	1	15	2	0.018	nd	0.006	nd	0.000	0.001	0.067	0.204
Gunyah	Scalded	1	15	3	0.023	nd	0.006	nd	0.000	0.002	0.048	0.189
Gunyah	Scalded	1	25	1	0.024	0.011	0.003	nd	0.004	0.001	0.033	0.177
Gunyah	Scalded	1	25	2	0.031	nd	0.012	nd	0.000	0.002	0.035	0.116
Gunyah	Scalded	1	25	3	0.010	nd	0.000	nd	0.000	0.002	0.023	0.153
Gunyah	Scalded	1	40	1	0.147	nd	0.114	nd	0.000	0.005	0.098	0.203
Gunyah	Scalded	1	40	2	0.198	nd	0.098	nd	0.001	0.005	0.122	0.187
Gunyah	Scalded	1	40	3	0.176	nd	0.096	nd	0.000	0.004	0.113	0.174
Gunyah	Scalded	1	60	1	0.160	0.019	0.041	nd	0.008	0.004	0.083	0.259
Gunyah	Scalded	1	60	2	0.148	nd	0.052	nd	0.000	0.004	0.081	0.195
Gunyah	Scalded	1	60	3	0.148	nd	0.049	nd	0.000	0.003	0.076	0.204
Gunyah	Scalded	1	85	1	0.139	nd	0.035	nd	0.000	0.003	0.067	0.201
Gunyah	Scalded	1	85	2	0.180	nd	0.052	nd	0.000	0.004	0.075	0.167
Gunyah	Scalded	1	85	3	0.150	0.005	0.037	nd	0.000	0.003	0.075	0.196
Gunyah	Scalded	2	2.5	1	0.004	nd	0.000	nd	0.007	0.007	0.064	0.354
Gunyah	Scalded	2	2.5	2	0.022	nd	0.000	0.002	0.019	0.012	0.083	0.256
Gunyah	Scalded	2	2.5	3	0.003	nd	0.001	nd	0.038	0.014	0.134	0.164
Gunyah	Scalded	2	7.5	1	0.024	nd	0.002	nd	0.000	0.002	0.051	0.150
Gunyah	Scalded	2	7.5	2	0.022	nd	0.002	nd	0.001	0.002	0.057	0.159
Gunyah	Scalded	2	7.5	3	0.002	nd	0.000	nd	0.000	0.001	0.000	0.000
Gunyah	Scalded	2	15	1	0.030	nd	0.006	nd	0.001	0.003	0.046	0.143
Gunyah	Scalded	2	15	2	0.034	nd	0.006	nd	0.000	0.002	0.043	0.139
Gunyah	Scalded	2	15	3	0.021	nd	0.005	nd	0.000	0.003	0.037	0.182
Tarcoola	Vegetated	1	2.5	1	0.030	nd	0.011	nd	0.000	0.004	0.035	0.153
Tarcoola	Vegetated	1	2.5	2	0.053	nd	0.023	nd	0.000	0.004	0.037	0.131
Tarcoola	Vegetated	1	2.5	3	0.023	nd	0.004	nd	0.000	0.004	0.035	0.174
Tarcoola	Vegetated	1	7.5	1	0.081	nd	0.024	nd	0.000	0.003	0.052	0.141
Tarcoola	Vegetated	1	7.5	2	0.120	nd	0.053	nd	0.000	0.005	0.054	0.126
Tarcoola	Vegetated	1	7.5	3	0.106	nd	0.068	nd	0.000	0.003	0.059	0.144
Tarcoola	Vegetated	1	15	1	0.171	nd	0.060	nd	0.002	0.004	0.096	0.219
Tarcoola	Vegetated	1	15	2	0.190	nd	0.071	nd	0.002	0.004	0.099	0.188
Tarcoola	Vegetated	1	15	3	0.097	nd	0.027	nd	0.002	0.003	0.082	0.280
Tarcoola	Vegetated	1	25	1	0.165	nd	0.047	nd	0.000	0.005	0.064	0.283
Tarcoola	Vegetated	1	25	2	0.117	nd	0.019	nd	0.000	0.003	0.065	0.247
Tarcoola	Vegetated	1	25	3	0.112	nd	0.023	nd	0.000	0.003	0.054	0.266
Tarcoola	Vegetated	1	40	1	0.007	nd	0.031	0.005	0.017	0.008	0.033	0.007
Tarcoola	Vegetated	1	40	2	nd	nd	0.000	nd	0.000	0.001	0.000	0.000
Tarcoola	Vegetated	1	40	3	0.008	nd	0.032	0.005	0.035	0.026	0.061	0.007
Tarcoola	Vegetated	2	2.5	1	0.021	0.007	0.012	0.001	0.001	0.004	0.008	0.009
Tarcoola	Vegetated	2	2.5	2	0.016	nd	0.003	0.000	0.002	0.003	0.015	0.016
Tarcoola	Vegetated	2	2.5	3	0.014	nd	0.002	0.001	0.005	0.003	0.026	0.017
Tarcoola	Vegetated	2	7.5	1	0.013	nd	0.002	0.001	0.002	0.002	0.019	0.025
Tarcoola	Vegetated	2	7.5	2	0.019	nd	0.003	0.001	0.002	0.004	0.022	0.030
Tarcoola	Vegetated	2	7.5	3	0.016	nd	0.002	0.001	0.002	0.003	0.021	0.026

Tarcoola	Vegetated	2	15	1	0.014	nd	0.003	0.001	0.000	0.002	0.021	0.035
Tarcoola	Vegetated	2	15	2	0.016	nd	0.004	0.001	0.002	0.003	0.022	0.032
Tarcoola	Vegetated	2	15	3	0.012	nd	0.002	0.000	0.000	0.002	0.015	0.025
Tarcoola	Vegetated	2	25	1	0.026	nd	0.007	nd	0.000	0.002	0.000	0.017
Tarcoola	Vegetated	2	25	2	0.022	nd	0.005	nd	0.000	0.002	0.000	0.027
Tarcoola	Vegetated	2	25	3	0.028	nd	0.009	nd	0.000	0.002	0.000	0.024
Tarcoola	Vegetated	2	40	1	0.006	nd	0.019	0.002	0.010	0.022	0.020	0.031
Tarcoola	Vegetated	2	40	2	0.007	nd	0.017	0.002	0.008	0.006	0.023	0.014
Tarcoola	Vegetated	2	40	3	0.008	nd	0.010	0.001	0.003	0.006	0.011	0.026

Table D3 Exchangeable cation concentrations for each sample. Nd indicates that the concentration was below the detection limit.

Site	Description	Profile	Depth	Rep	Al ($cmol_c/kg$)	B ($cmol_c/kg$)	Fe ($cmol_c/kg$)	Mn ($cmol_c/kg$)	Ca ($cmol_c/kg$)	K ($cmol_c/kg$)	Mg ($cmol_c/kg$)	Na ($cmol_c/kg$)
Tarcoola	Scald	1	2.5	1	nd	nd	nd	nd	0.005	0.002	0.020	0.074
Tarcoola	Scald	1	2.5	2	nd	nd	nd	nd	0.004	0.001	0.017	0.055
Tarcoola	Scald	1	2.5	3	nd	nd	nd	nd	0.003	0.001	0.013	0.053
Tarcoola	Scald	1	7.5	1	nd	nd	nd	nd	0.004	0.001	0.027	0.044
Tarcoola	Scald	1	7.5	2	nd	nd	nd	nd	0.003	0.001	0.033	0.050
Tarcoola	Scald	1	7.5	3	nd	nd	nd	nd	0.004	0.001	0.037	0.048
Tarcoola	Scald	1	15	1	nd	nd	nd	nd	0.003	0.001	0.031	0.061
Tarcoola	Scald	1	15	2	nd	nd	nd	nd	0.003	0.001	0.042	0.064
Tarcoola	Scald	1	15	3	nd	nd	nd	nd	0.002	0.001	0.036	0.068
Tarcoola	Scald	1	25	1	nd	nd	nd	nd	0.001	0.001	0.013	0.072
Tarcoola	Scald	1	25	2	nd	nd	nd	nd	0.001	0.001	0.027	0.074
Tarcoola	Scald	1	25	3	nd	nd	nd	nd	0.000	0.001	0.008	0.074
Tarcoola	Scald	1	40	1	nd	nd	nd	nd	0.000	0.001	0.042	0.082
Tarcoola	Scald	1	40	2	nd	nd	nd	nd	0.000	0.001	0.045	0.074
Tarcoola	Scald	1	40	3	nd	nd	nd	nd	0.000	0.001	0.035	0.087
Tarcoola	Scald	1	60	1	nd	nd	nd	nd	0.001	0.002	0.162	0.124
Tarcoola	Scald	1	60	2	nd	nd	nd	nd	0.002	0.004	0.220	0.170
Tarcoola	Scald	1	60	3	nd	nd	nd	nd	0.001	0.003	0.165	0.141
Tarcoola	Scald	1	85	1	nd	nd	nd	nd	0.000	0.002	0.182	0.140
Tarcoola	Scald	1	85	2	nd	nd	nd	nd	0.000	0.002	0.132	0.106
Tarcoola	Scald	1	85	3	nd	nd	nd	nd	0.000	0.002	0.154	0.121
Tarcoola	Scald	2	2.5	1	nd	nd	nd	nd	0.003	0.001	0.025	0.028
Tarcoola	Scald	2	2.5	2	nd	nd	nd	nd	0.003	0.001	0.027	0.028
Tarcoola	Scald	2	2.5	3	nd	nd	nd	nd	0.003	0.001	0.023	0.027
Tarcoola	Scald	2	7.5	1	nd	nd	nd	nd	0.004	0.001	0.050	0.052
Tarcoola	Scald	2	7.5	2	nd	nd	nd	nd	0.003	0.001	0.041	0.038
Tarcoola	Scald	2	7.5	3	nd	nd	nd	nd	0.006	0.001	0.059	0.056
Tarcoola	Scald	2	15	1	nd	nd	nd	nd	0.003	0.002	0.034	0.072
Tarcoola	Scald	2	15	2	nd	nd	nd	nd	0.003	0.002	0.031	0.067
Tarcoola	Scald	2	15	3	nd	nd	nd	nd	0.003	0.001	0.034	0.064
Tarcoola	Scald	2	25	1	nd	nd	nd	nd	0.001	0.001	0.018	0.065
Tarcoola	Scald	2	25	2	nd	nd	nd	nd	0.001	0.001	0.020	0.067
Tarcoola	Scald	2	25	3	nd	nd	nd	nd	0.001	0.001	0.062	0.067
Tarcoola	Scald	2	40	1	nd	nd	nd	nd	0.001	0.001	0.027	0.062
Tarcoola	Scald	2	40	2	nd	nd	nd	nd	0.003	0.003	0.120	0.138
Tarcoola	Scald	2	40	3	nd	nd	nd	nd	0.002	0.002	0.070	0.081

Tarcoola	Scald	2	60	1	nd	nd	nd	nd	0.003	0.002	0.137	0.123
Tarcoola	Scald	2	60	2	nd	nd	nd	nd	0.002	0.003	0.154	0.129
Tarcoola	Scald	2	60	3	nd	nd	nd	nd	0.002	0.003	0.156	0.143
Tarcoola	Scald	2	85	1	nd	nd	nd	nd	0.001	0.001	0.102	0.097
Tarcoola	Scald	2	85	2	nd	nd	nd	nd	0.002	0.001	0.106	0.101
Tarcoola	Scald	2	85	3	nd	nd	nd	nd	0.000	0.001	0.107	0.098
Tarcoola	Depression	1	2.5	1	nd	nd	nd	0.001	0.040	0.004	0.137	0.010
Tarcoola	Depression	1	2.5	2	nd	nd	nd	0.001	0.050	0.005	0.169	0.010
Tarcoola	Depression	1	2.5	3	nd	nd	nd	0.001	0.050	0.004	0.180	0.011
Tarcoola	Depression	1	7.5	1	nd	nd	nd	0.000	0.036	0.003	0.180	0.012
Tarcoola	Depression	1	7.5	2	nd	nd	nd	0.000	0.037	0.003	0.187	0.010
Tarcoola	Depression	1	7.5	3	nd	nd	nd	0.000	0.036	0.003	0.181	0.011
Tarcoola	Depression	1	15	1	nd	nd	nd	nd	0.027	0.002	0.156	0.009
Tarcoola	Depression	1	15	2	nd	nd	nd	nd	0.029	0.002	0.174	0.009
Tarcoola	Depression	1	15	3	nd	nd	nd	nd	0.026	0.002	0.156	0.008
Tarcoola	Depression	1	25	1	nd	nd	nd	nd	0.025	0.002	0.160	0.009
Tarcoola	Depression	1	25	2	nd	nd	nd	nd	0.028	0.002	0.200	0.011
Tarcoola	Depression	1	25	3	nd	nd	nd	nd	0.032	0.002	0.207	0.013
Tarcoola	Depression	1	40	1	nd	nd	nd	nd	0.020	0.002	0.169	0.014
Tarcoola	Depression	1	40	2	nd	nd	nd	nd	0.019	0.002	0.159	0.012
Tarcoola	Depression	1	40	3	nd	nd	nd	nd	0.016	0.002	0.152	0.011
Tarcoola	Depression	1	60	1	nd	nd	nd	nd	0.039	0.002	0.305	0.022
Tarcoola	Depression	1	60	2	nd	nd	nd	nd	0.036	0.002	0.280	0.023
Tarcoola	Depression	1	60	3	nd	nd	nd	nd	0.045	0.002	0.351	0.020
Tarcoola	Depression	1	85	1	nd	nd	nd	nd	0.009	0.002	0.100	0.016
Tarcoola	Depression	1	85	2	nd	nd	nd	nd	0.010	0.002	0.134	0.024
Tarcoola	Depression	1	85	3	nd	nd	nd	nd	0.010	0.002	0.141	0.019
Tarcoola	Depression	2	2.5	1	nd	nd	nd	0.001	0.078	0.007	0.212	0.006
Tarcoola	Depression	2	2.5	2	nd	nd	nd	0.001	0.061	0.006	0.174	0.008
Tarcoola	Depression	2	2.5	3	nd	nd	nd	0.001	0.055	0.006	0.166	0.006
Tarcoola	Depression	2	7.5	1	nd	nd	nd	nd	0.036	0.003	0.189	0.008
Tarcoola	Depression	2	7.5	2	nd	nd	nd	nd	0.038	0.003	0.178	0.008
Tarcoola	Depression	2	7.5	3	nd	nd	nd	nd	0.036	0.004	0.191	0.010
Tarcoola	Depression	2	15	1	nd	nd	nd	nd	0.031	0.002	0.191	0.011
Tarcoola	Depression	2	15	2	nd	nd	nd	nd	0.035	0.002	0.199	0.009
Tarcoola	Depression	2	15	3	nd	nd	nd	nd	0.043	0.003	0.215	0.007
Tarcoola	Depression	2	25	1	nd	nd	nd	nd	0.032	0.002	0.235	0.017
Tarcoola	Depression	2	25	2	nd	nd	nd	nd	0.033	0.002	0.208	0.015
Tarcoola	Depression	2	25	3	nd	nd	nd	nd	0.022	0.002	0.148	0.013
Tarcoola	Depression	2	40	1	nd	nd	nd	nd	0.022	0.002	0.185	0.031
Tarcoola	Depression	2	40	2	nd	nd	nd	nd	0.024	0.002	0.196	0.036
Tarcoola	Depression	2	40	3	nd	nd	nd	nd	0.021	0.002	0.181	0.028
Tarcoola	Depression	2	60	1	nd	nd	nd	nd	0.060	0.003	0.410	0.076
Tarcoola	Depression	2	60	2	nd	nd	nd	nd	0.057	0.002	0.388	0.067
Tarcoola	Depression	2	60	3	nd	nd	nd	nd	0.033	0.002	0.272	0.053
Tarcoola	Depression	2	85	1	nd	nd	nd	nd	0.036	0.002	0.289	0.055
Tarcoola	Depression	2	85	2	nd	nd	nd	nd	0.035	0.002	0.313	0.056
Tarcoola	Depression	2	85	3	nd	nd	nd	nd	0.038	0.002	0.324	0.067
Gunyah	Pasture	1	2.5	1	nd	nd	0.001	nd	0.154	0.002	0.006	0.003
Gunyah	Pasture	1	2.5	2	nd	nd	0.000	0.000	0.148	0.004	0.010	0.003
Gunyah	Pasture	1	2.5	3	nd	nd	0.001	0.000	0.113	0.003	0.006	0.002
Gunyah	Pasture	1	7.5	1	nd	nd	0.001	nd	0.026	0.001	0.000	0.002

Gunyah	Pasture	1	7.5	2	nd	nd	0.001	nd	0.040	0.001	0.000	0.002
Gunyah	Pasture	1	7.5	3	nd	nd	nd	nd	0.021	0.001	0.005	0.003
Gunyah	Pasture	1	15	1	nd	nd	nd	nd	0.013	0.001	0.000	0.002
Gunyah	Pasture	1	15	2	nd	nd	nd	nd	0.010	0.001	0.000	0.001
Gunyah	Pasture	1	15	3	nd	nd	nd	nd	0.017	0.001	0.000	0.002
Gunyah	Pasture	1	25	1	nd	nd	nd	nd	0.005	0.001	0.002	0.003
Gunyah	Pasture	1	25	2	nd	nd	nd	nd	0.007	0.000	0.002	0.003
Gunyah	Pasture	1	25	3	nd	nd	nd	nd	0.007	0.000	0.001	0.002
Gunyah	Pasture	1	40	1	nd	nd	nd	nd	0.012	0.001	0.051	0.030
Gunyah	Pasture	1	40	2	nd	nd	nd	nd	0.012	0.001	0.051	0.027
Gunyah	Pasture	1	40	3	nd	nd	nd	nd	0.009	0.001	0.044	0.023
Gunyah	Pasture	1	60	1	nd	nd	nd	nd	0.009	0.002	0.248	0.063
Gunyah	Pasture	1	60	2	nd	nd	nd	nd	0.007	0.001	0.082	0.033
Gunyah	Pasture	1	60	3	nd	nd	nd	nd	0.014	0.001	0.078	0.039
Gunyah	Pasture	2	2.5	1	nd	nd	nd	nd	0.248	0.002	0.004	0.003
Gunyah	Pasture	2	2.5	2	nd	nd	nd	nd	0.147	0.003	0.008	0.004
Gunyah	Pasture	2	2.5	3	nd	nd	0.001	nd	0.181	0.002	0.003	0.002
Gunyah	Pasture	2	7.5	1	nd	nd	0.001	nd	0.018	0.001	0.000	0.001
Gunyah	Pasture	2	7.5	2	nd	nd	0.001	nd	0.020	0.001	0.000	0.001
Gunyah	Pasture	2	7.5	3	nd	nd	nd	0.000	0.019	0.001	0.000	0.001
Gunyah	Pasture	2	15	1	nd	nd	nd	0.000	0.029	0.001	0.000	0.001
Gunyah	Pasture	2	15	2	nd	nd	nd	0.000	0.010	0.001	0.000	0.001
Gunyah	Pasture	2	15	3	nd	nd	nd	0.000	0.008	0.001	0.000	0.001
Gunyah	Pasture	2	25	1	nd	nd	nd	0.000	0.005	0.001	0.000	0.001
Gunyah	Pasture	2	25	2	nd	nd	nd	0.000	0.004	0.001	0.000	0.001
Gunyah	Pasture	2	25	3	nd	nd	nd	0.000	0.008	0.001	0.000	0.001
Gunyah	Pasture	2	40	1	nd	nd	nd	0.000	0.007	0.001	0.000	0.001
Gunyah	Pasture	2	40	2	nd	nd	nd	0.000	0.003	0.000	0.000	0.001
Gunyah	Pasture	2	40	3	nd	nd	nd	0.000	0.003	0.001	0.000	0.002
Gunyah	Pasture	2	60	1	nd	nd	nd	nd	0.003	0.001	0.102	0.033
Gunyah	Pasture	2	60	2	nd	nd	nd	nd	0.003	0.001	0.127	0.039
Gunyah	Pasture	2	60	3	nd	nd	nd	nd	0.004	0.002	0.153	0.043
Gunyah	Vegetated	1	2.5	1	nd	nd	nd	0.002	0.047	0.004	0.023	0.003
Gunyah	Vegetated	1	2.5	2	nd	nd	nd	0.001	0.039	0.003	0.018	0.003
Gunyah	Vegetated	1	2.5	3	nd	nd	nd	0.001	0.045	0.004	0.028	0.003
Gunyah	Vegetated	1	7.5	1	nd	nd	nd	0.001	0.027	0.001	0.006	0.004
Gunyah	Vegetated	1	7.5	2	nd	nd	nd	0.000	0.022	0.001	0.005	0.002
Gunyah	Vegetated	1	7.5	3	nd	nd	nd	0.000	0.021	0.001	0.004	0.002
Gunyah	Vegetated	1	15	1	nd	nd	nd	0.000	0.017	0.001	0.008	0.002
Gunyah	Vegetated	1	15	2	nd	nd	nd	0.000	0.020	0.001	0.010	0.002
Gunyah	Vegetated	1	15	3	nd	nd	nd	0.000	0.018	0.000	0.008	0.002
Gunyah	Vegetated	1	25	1	nd	nd	nd	nd	0.004	0.000	0.013	0.004
Gunyah	Vegetated	1	25	2	nd	nd	nd	nd	0.005	0.001	0.025	0.005
Gunyah	Vegetated	1	25	3	nd	nd	nd	nd	0.006	0.001	0.013	0.003
Gunyah	Vegetated	1	40	1	nd	nd	nd	0.000	0.028	0.001	0.003	0.004
Gunyah	Vegetated	1	40	2	nd	nd	nd	0.000	0.020	0.001	0.002	0.003
Gunyah	Vegetated	1	40	3	nd	nd	nd	0.000	0.021	0.001	0.004	0.002
Gunyah	Vegetated	1	60	1	nd	nd	nd	nd	0.003	0.001	0.072	0.021
Gunyah	Vegetated	1	60	2	nd	nd	nd	nd	0.003	0.001	0.074	0.023
Gunyah	Vegetated	1	60	3	nd	nd	nd	nd	0.002	0.001	0.040	0.012
Gunyah	Vegetated	1	85	1	nd	nd	nd	nd	0.004	0.003	0.190	0.061
Gunyah	Vegetated	1	85	2	nd	nd	nd	nd	0.005	0.003	0.258	0.085

Gunyah	Vegetated	1	85	3	nd	nd	nd	nd	0.005	0.003	0.235	0.086
Gunyah	Vegetated	2	2.5	1	nd	nd	nd	0.001	0.028	0.001	0.012	0.005
Gunyah	Vegetated	2	2.5	2	nd	nd	nd	0.001	0.041	0.002	0.024	0.009
Gunyah	Vegetated	2	2.5	3	nd	nd	nd	0.002	0.048	0.001	0.035	0.012
Gunyah	Vegetated	2	7.5	1	nd	nd	nd	0.001	0.028	0.001	0.025	0.011
Gunyah	Vegetated	2	7.5	2	nd	nd	nd	0.001	0.020	0.001	0.019	0.014
Gunyah	Vegetated	2	7.5	3	nd	nd	nd	0.001	0.013	0.001	0.011	0.003
Gunyah	Vegetated	2	15	1	nd	nd	nd	0.000	0.013	0.001	0.011	0.010
Gunyah	Vegetated	2	15	2	nd	nd	nd	0.000	0.018	0.001	0.014	0.018
Gunyah	Vegetated	2	15	3	nd	nd	nd	0.000	0.020	0.001	0.013	0.011
Gunyah	Vegetated	2	25	1	nd	nd	nd	0.000	0.008	0.000	0.008	0.009
Gunyah	Vegetated	2	25	2	nd	nd	nd	0.000	0.006	0.000	0.005	0.008
Gunyah	Vegetated	2	25	3	nd	nd	nd	0.000	0.013	0.000	0.015	0.016
Gunyah	Vegetated	2	40	1	nd	nd	nd	nd	0.004	0.000	0.008	0.008
Gunyah	Vegetated	2	40	2	nd	nd	nd	0.000	0.006	0.000	0.016	0.009
Gunyah	Vegetated	2	40	3	nd	nd	nd	nd	0.004	0.000	0.010	0.013
Gunyah	Vegetated	2	60	1	nd	nd	nd	nd	0.006	0.001	0.086	0.027
Gunyah	Vegetated	2	60	2	nd	nd	nd	nd	0.007	0.001	0.104	0.027
Gunyah	Vegetated	2	60	3	nd	nd	nd	nd	0.004	0.001	0.063	0.019
Gunyah	Vegetated	2	85	1	nd	nd	nd	nd	0.007	0.001	0.079	0.034
Gunyah	Vegetated	2	85	2	nd	nd	nd	nd	0.008	0.002	0.282	0.093
Gunyah	Vegetated	2	85	3	nd	nd	nd	nd	0.007	0.002	0.241	0.084
Gunyah	Eroded	1	2.5	1	nd	nd	nd	0.000	0.011	0.001	0.075	0.022
Gunyah	Eroded	1	2.5	2	nd	nd	nd	nd	0.006	0.001	0.055	0.026
Gunyah	Eroded	1	2.5	3	nd	nd	nd	nd	0.012	0.001	0.074	0.026
Gunyah	Eroded	1	7.5	1	nd	nd	nd	nd	0.005	0.001	0.060	0.024
Gunyah	Eroded	1	7.5	2	nd	nd	nd	nd	0.007	0.001	0.062	0.018
Gunyah	Eroded	1	7.5	3	nd	nd	nd	nd	0.008	0.001	0.065	0.021
Gunyah	Eroded	1	15	1	nd	nd	nd	nd	0.003	0.001	0.049	0.017
Gunyah	Eroded	1	15	2	nd	nd	nd	nd	0.005	0.001	0.057	0.015
Gunyah	Eroded	1	15	3	nd	nd	nd	nd	0.003	0.001	0.043	0.016
Gunyah	Eroded	1	25	1	nd	nd	nd	nd	0.003	0.001	0.048	0.014
Gunyah	Eroded	1	25	2	nd	nd	nd	nd	0.003	0.001	0.068	0.017
Gunyah	Eroded	1	25	3	nd	nd	nd	nd	0.003	0.001	0.064	0.015
Gunyah	Eroded	1	40	1	nd	nd	nd	nd	0.003	0.001	0.118	0.025
Gunyah	Eroded	1	40	2	nd	nd	nd	nd	0.005	0.002	0.149	0.027
Gunyah	Eroded	1	40	3	nd	nd	nd	nd	0.005	0.002	0.147	0.026
Gunyah	Eroded	1	60	1	nd	nd	nd	nd	0.008	0.003	0.234	0.041
Gunyah	Eroded	1	60	2	nd	nd	nd	nd	0.008	0.002	0.219	0.035
Gunyah	Eroded	1	60	3	nd	nd	nd	nd	0.008	0.003	0.238	0.037
Gunyah	Eroded	1	85	1	nd	nd	nd	nd	0.011	0.003	0.289	0.048
Gunyah	Eroded	1	85	2	nd	nd	nd	nd	0.013	0.003	0.303	0.046
Gunyah	Eroded	1	85	3	nd	nd	nd	nd	0.013	0.002	0.276	0.041
Gunyah	Eroded	2	2.5	1	nd	nd	nd	nd	0.035	0.002	0.137	0.020
Gunyah	Eroded	2	2.5	2	nd	nd	nd	nd	0.024	0.002	0.139	0.047
Gunyah	Eroded	2	2.5	3	nd	nd	nd	nd	0.038	0.001	0.155	0.039
Gunyah	Eroded	2	7.5	1	nd	nd	nd	nd	0.009	0.001	0.061	0.020
Gunyah	Eroded	2	7.5	2	nd	nd	nd	nd	0.008	0.001	0.067	0.025
Gunyah	Eroded	2	7.5	3	nd	nd	nd	nd	0.010	0.001	0.057	0.023
Gunyah	Eroded	2	15	1	nd	nd	nd	nd	0.007	0.001	0.060	0.024
Gunyah	Eroded	2	15	2	nd	nd	nd	nd	0.006	0.001	0.052	0.017
Gunyah	Eroded	2	15	3	nd	nd	nd	nd	0.005	0.001	0.042	0.011

Gunyah	Eroded	2	25	1	nd	nd	nd	nd	0.005	0.001	0.054	0.016
Gunyah	Eroded	2	25	2	nd	nd	nd	nd	0.004	0.001	0.051	0.018
Gunyah	Eroded	2	25	3	nd	nd	nd	nd	0.004	0.001	0.048	0.015
Gunyah	Eroded	2	40	1	nd	nd	nd	nd	0.011	0.002	0.192	0.029
Gunyah	Eroded	2	40	2	nd	nd	nd	nd	0.009	0.002	0.152	0.022
Gunyah	Eroded	2	40	3	nd	nd	nd	nd	0.009	0.002	0.170	0.024
Gunyah	Eroded	2	60	1	nd	nd	nd	nd	0.020	0.004	0.370	0.041
Gunyah	Eroded	2	60	2	nd	nd	nd	nd	0.020	0.003	0.375	0.041
Gunyah	Eroded	2	60	3	nd	nd	nd	nd	0.017	0.003	0.326	0.037
Gunyah	Eroded	2	85	1	nd	nd	nd	nd	0.015	0.002	0.262	0.031
Gunyah	Eroded	2	85	2	nd	nd	nd	nd	0.016	0.003	0.297	0.034
Gunyah	Eroded	2	85	3	nd	nd	nd	nd	0.016	0.002	0.266	0.030
Gunyah	Scalded	1	2.5	1	nd	nd	nd	0.000	0.072	0.002	0.178	0.014
Gunyah	Scalded	1	2.5	2	nd	nd	nd	0.000	0.114	0.005	0.163	0.007
Gunyah	Scalded	1	2.5	3	nd	nd	nd	nd	0.118	0.005	0.160	0.005
Gunyah	Scalded	1	7.5	1	nd	nd	nd	nd	0.020	0.001	0.123	0.024
Gunyah	Scalded	1	7.5	2	nd	nd	nd	nd	0.020	0.001	0.119	0.023
Gunyah	Scalded	1	7.5	3	nd	nd	nd	nd	0.018	0.001	0.110	0.019
Gunyah	Scalded	1	15	1	nd	nd	nd	nd	0.016	0.001	0.105	0.023
Gunyah	Scalded	1	15	2	nd	nd	nd	nd	0.011	0.001	0.070	0.018
Gunyah	Scalded	1	15	3	nd	nd	nd	nd	0.012	0.001	0.080	0.022
Gunyah	Scalded	1	25	1	nd	nd	nd	nd	0.007	0.001	0.057	0.019
Gunyah	Scalded	1	25	2	nd	nd	nd	nd	0.008	0.001	0.052	0.015
Gunyah	Scalded	1	25	3	nd	nd	nd	nd	0.006	0.000	0.044	0.017
Gunyah	Scalded	1	40	1	nd	nd	nd	nd	0.009	0.001	0.141	0.027
Gunyah	Scalded	1	40	2	nd	nd	nd	nd	0.010	0.001	0.166	0.027
Gunyah	Scalded	1	40	3	nd	nd	nd	nd	0.011	0.001	0.171	0.027
Gunyah	Scalded	1	60	1	nd	nd	nd	nd	0.017	0.002	0.265	0.041
Gunyah	Scalded	1	60	2	nd	nd	nd	nd	0.015	0.002	0.217	0.030
Gunyah	Scalded	1	60	3	nd	nd	nd	nd	0.015	0.002	0.218	0.030
Gunyah	Scalded	1	85	1	nd	nd	nd	nd	0.018	0.002	0.262	0.035
Gunyah	Scalded	1	85	2	nd	nd	nd	nd	0.019	0.003	0.278	0.035
Gunyah	Scalded	1	85	3	nd	nd	nd	nd	0.019	0.002	0.275	0.037
Gunyah	Scalded	2	2.5	1	nd	nd	nd	0.000	0.096	0.003	0.238	0.036
Gunyah	Scalded	2	2.5	2	nd	nd	nd	0.000	0.068	0.003	0.235	0.023
Gunyah	Scalded	2	2.5	3	nd	nd	nd	0.000	0.104	0.003	0.193	0.016
Gunyah	Scalded	2	7.5	1	nd	nd	nd	0.000	0.021	0.001	0.101	0.015
Gunyah	Scalded	2	7.5	2	nd	nd	nd	nd	0.021	0.001	0.124	0.016
Gunyah	Scalded	2	7.5	3	nd	nd	nd	0.000	0.019	0.001	0.104	0.022
Gunyah	Scalded	2	15	1	nd	nd	nd	nd	0.012	0.001	0.066	0.013
Gunyah	Scalded	2	15	2	nd	nd	nd	0.000	0.014	0.001	0.071	0.014
Gunyah	Scalded	2	15	3	nd	nd	nd	nd	0.011	0.000	0.064	0.015
Tarcoola	Vegetated	1	2.5	1	nd	nd	nd	nd	0.008	0.001	0.054	0.013
Tarcoola	Vegetated	1	2.5	2	nd	nd	nd	nd	0.008	0.001	0.060	0.011
Tarcoola	Vegetated	1	2.5	3	nd	nd	nd	nd	0.009	0.001	0.059	0.014
Tarcoola	Vegetated	1	7.5	1	nd	nd	nd	nd	0.008	0.001	0.101	0.017
Tarcoola	Vegetated	1	7.5	2	nd	nd	nd	nd	0.009	0.001	0.096	0.015
Tarcoola	Vegetated	1	7.5	3	nd	nd	nd	nd	0.011	0.001	0.120	0.019
Tarcoola	Vegetated	1	15	1	nd	nd	nd	nd	0.019	0.002	0.257	0.030
Tarcoola	Vegetated	1	15	2	nd	nd	nd	nd	0.019	0.002	0.267	0.032
Tarcoola	Vegetated	1	15	3	nd	nd	nd	nd	0.026	0.004	0.370	0.046
Tarcoola	Vegetated	1	25	1	nd	nd	nd	nd	0.023	0.003	0.317	0.041

Tarcoola	Vegetated	1	25	2	nd	nd	nd	nd	0.021	0.002	0.303	0.038
Tarcoola	Vegetated	1	25	3	nd	nd	nd	nd	0.023	0.003	0.330	0.046
Tarcoola	Vegetated	1	40	1	nd	nd	nd	0.003	0.038	0.005	0.032	0.002
Tarcoola	Vegetated	1	40	2	nd	nd	nd	0.003	0.036	0.006	0.034	0.002
Tarcoola	Vegetated	1	40	3	nd	nd	nd	0.002	0.054	0.008	0.040	0.004
Tarcoola	Vegetated	2	2.5	1	nd	nd	nd	0.001	0.038	0.002	0.041	0.002
Tarcoola	Vegetated	2	2.5	2	nd	nd	nd	0.000	0.037	0.002	0.046	0.002
Tarcoola	Vegetated	2	2.5	3	nd	nd	nd	0.000	0.045	0.002	0.063	0.003
Tarcoola	Vegetated	2	7.5	1	nd	nd	nd	0.000	0.049	0.002	0.081	0.004
Tarcoola	Vegetated	2	7.5	2	nd	nd	nd	0.000	0.049	0.002	0.093	0.004
Tarcoola	Vegetated	2	7.5	3	nd	nd	nd	0.000	0.044	0.002	0.085	0.004
Tarcoola	Vegetated	2	15	1	nd	nd	nd	0.000	0.041	0.002	0.091	0.006
Tarcoola	Vegetated	2	15	2	nd	nd	nd	0.000	0.039	0.002	0.076	0.005
Tarcoola	Vegetated	2	15	3	nd	nd	nd	0.000	0.033	0.002	0.070	0.005
Tarcoola	Vegetated	2	25	1	nd	nd	nd	nd	0.008	0.001	0.023	0.004
Tarcoola	Vegetated	2	25	2	nd	nd	nd	nd	0.007	0.001	0.018	0.004
Tarcoola	Vegetated	2	25	3	nd	nd	nd	0.000	0.009	0.001	0.026	0.006
Tarcoola	Vegetated	2	40	1	nd	nd	nd	0.002	0.023	0.006	0.016	0.002
Tarcoola	Vegetated	2	40	2	nd	nd	nd	0.004	0.033	0.004	0.026	0.002
Tarcoola	Vegetated	2	40	3	nd	nd	nd	0.003	0.034	0.005	0.028	0.003

Table D4 Raw means of SAR of each depth at each microsite and site (* indicates that data is not applicable)

Depth (cm)		0-5	5-10	10-20	20-30	30-50	50-70	70-100
Description	**Site**							
Eroded	**Gunyah**	5.408	4.898	2.563	2.083	1.372	1.968	1.879
	Riverview	*	*	*	*	*	*	*
	Tarcoola	*	*	*	*	*	*	*
Pasture	**Gunyah**	0.115	0.21	0.261	1.13	1.271	3.406	*
	Riverview	*	*	*	*	*	*	*
	Tarcoola	*	*	*	*	*	*	*
Scald	**Gunyah**	1.636	1.973	2.479	2.611	1.812	2.394	2.799
	Riverview	*	*	*	*	*	*	*
	Tarcoola	2.264	1.738	3.132	5.508	3.698	3.575	2.409
Vegetated	**Gunyah**	0.634	0.885	2.351	3.738	3.756	2.983	5.694
	Tarcoola	0.280	0.797	1.768	5.157	2.901	*	*
	Tarcoola (Depression)	0.515	0.765	0.745	1.118	1.487	2.602	2.697

Table D5 Raw means of SOC (%) of each depth at each microsite and site (* indicates that no data is available)

Depth (cm)		0-5	5-10	10-20	20-30	30-50	50-70	70-100
Description	**Site**							
Eroded	**Gunyah**	0.3017	0.2050	0.1317	0.0850	0.0983	0.1167	0.1150
	Riverview	*	*	*	*	*	*	*
	Tarcoola	*	*	*	*	*	*	*
Pasture	**Gunyah**	2.3500	1.2817	0.9700	0.4717	0.1683	0.0933	*
	Riverview	*	*	*	*	*	*	*
	Tarcoola	*	*	*	*	*	*	*
Scald	**Gunyah**	1.5200	0.4500	0.2533	0.1517	0.0933	0.1017	0.0850
	Riverview	*	*	*	*	*	*	*
	Tarcoola	0.1583	0.1933	0.2217	0.2650	0.1400	0.1417	0.0800
Vegetated	**Gunyah**	2.3083	1.600	1.0467	0.6983	0.6667	0.2617	0.1883
	Tarcoola	1.9600	0.8333	1.1650	1.4600	0.3083	*	*
	Tarcoola (Depression)	2.7133	0.8283	0.700	0.600	0.4317	2.000	0.8550