# 2012-AUSNZ-conference-proceedings

**Conference Paper** · June 2021



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# Soil Science

**JOINT AUSTRALIAN AND NEW ZEALAND SOIL SCIENCE CONFERENCE**

*Soil solutions for diverse landscapes*

WREST POINT HOTEL AND CONVENTION CENTRE, HOBART, TASMANIA 2 - 7 DECEMBER 2012

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# Proceedings

Proceedings of the 5th Joint Australian and New Zealand Soil Science Conference
Published in 2012 by:
Australian Society of Soil Science Incorporated
PO Box 1349, Warragul, Victoria, Australia 3820
Phone: 03 5635 2370
office@soilscienceaustralia.org



The editors verify that all full 4 page papers presented in these Proceedings have been peer reviewed by expert referees who were independent to the Editorial Committee.

Dr Lucy Burkitt
Dr Leigh Sparrow
Editors of the Proceedings of the 5th Joint Australian and New Zealand Soil Science Conference

The correct reference for papers presented at this conference is: [Authors] (2012) [Title of paper]. In: Proceedings of the 5th Joint Australian and New Zealand Soil Science Conference: Soil solutions for diverse landscapes. Hobart. (Eds LL Burkitt and LA Sparrow). [page No.]. (Australian Society of Soil Science Inc.)

ISBN: 978-0-646-59142-1

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## KEYNOTE INVITED SPEAKER

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# Soil – the long continuum

Henry Janzen

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Terrestrial ecosystems worldwide now face impending pressures, driven in part by the demands of a growing human presence. These stresses – related to food, water, energy, climate, wastes, biodiversity, among others – are all intertwined. And all are tied, one way or another, to soil: the thin evolving membrane woven by life and time across all lands, an interface between earth and sky, between solid and fluid, between living and inert. But soil spans not only continents; it stretches also over time, retaining memories of what once was and harbouring prospects of what is yet to be. My modest aim is to contemplate this long continuum, the soil, in several dimensions, exploring the questions: How can soils and the way we use them restore the seamless continuity among all biota, within and among ecosystems, and especially over time? And how can we, who delight in soil's mysteries, sustain and elevate this renewing, cohesive function, now so critical in the face of coming stresses? Such questions, and the collective learning they elicit, might help us and our successors live more wisely on the land. We ask the questions, not only of each other, but of the soil, the long continuum, probing its memories and its latent prospects.

# Monday 3 December

## SOIL FERTILITY AND SOIL CONTAMINANTS (NUTRIENT LOSS/MANAGEMENT)

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# A system to improve catchment water quality and a profitability of pasture-based dairy farms

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The loss of phosphorus (P) from dairy farms impairs profit and water quality. Splitting mixed ryegrass-white clover swards into monocultures can increase milk solids. The low P requirement of ryegrass also means that it can be placed in areas that contribute runoff, and little P applied to maintain yield, while higher P requiring clover can be placed in non-runoff producing areas of the catchment. This was also hypothesized to decrease P loss. A paired catchment trial was conducted of streamflow and pasture yield over 5 years. During the first 3 years, both catchments were treated the same, but afterwards, 40% of one catchment, near the stream was cultivated and ryegrass planted, while the rest of the catchment was direct drilled in clover and chicory. Pasture yield and modelling suggested profitability increased by $46/ha over a mixed sward. Stream losses of algal available P and total P decreased by 45 and 24%, respectively. This indicates considerable potential to improve profitability and water quality in an intensive dairy system.

# Risk of phosphorus runoff following wastewater application for two Tasmanian pasture soils used for dairying

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Native soil phosphorus (P) levels are regularly augmented to provide the nutrient levels required by pastures to support milk and livestock production. Food processing industries produce large quantities of wastewater that can contain P in concentrations useful for optimising pasture growth. Wastewaters that do not contain harmful contaminants can provide viable alternatives to supplement traditional fertilisers for properties located near industries.As with all fertiliser use, wastewater irrigation must be managed to prevent runoff and leaching of nutrients that could be detrimental to surface and ground water quality.

In this study the P sorption properties of two Tasmanian pasture soils commonly used for dairying were investigated - a high P sorbing Ferrosol (clay loam) and low P sorbing Hydrosol (sandy loam). A series of runoff experiments were conducted on miniswards of perennial ryegrass to determine temporal trends in P runoff concentration with the timing of a runoff event after P application from wastewater and other sources.

The sorption properties of soils, the timing of runoff events and soil P concentration are all important variables in determining the risk of P being transported in runoff from pastures. High P sorbing soils have a lower risk of environmentally detrimental P concentration occurring in runoff compared to low P sorbing soils. The risk of environmentally detrimental P concentrations occurring in runoff increases after application of P and at higher P application rates.

# Applying a 1D soil model to assess diffuse pollution at a regional scale

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Multiple model executions encompassing the range of soil, land use and management, climate and terrain in a region can provide insight into model sensitivity as well as geographic variability. The spatial and temporal leaching behaviour of a suite of pesticides and nutrients over the lower south-east of South Australia were simulated using information derived from state soil, land use and weather data, with the aim of evaluating groundwater contamination risks. A 1D soil model (LEACHM) was adapted to read raster maps of landscape variables likely to influence solute leaching. Unique combinations of soil, crop, management and climate were identified and numbered. Library files contained detailed input data for each soil type, land use, management practice (e.g. nutrients, irrigation) and climate. Sixty-year simulations for each unique combination were performed. Simulations included degradation and leaching of pesticides, carbon and nitrogen cycling, and plant uptake and leaching of N and P for a number of assumptions regarding depth to groundwater. Each automated simulation set consisted of about 1600 60-year simulations, but required only a few hours of execution time on an eight-core computer. Procedures were developed to enable efficient integration, analysis and display of output data for evaluating pollution risks associated with cropping and management systems and generation of input files for groundwater models. Any of the model input files can be rerun as stand-alone simulations, which allows for more detailed study of, for example, the consequences of changes in management.

# Using a Bayesian network to investigate environmental management of vegetable production in the Lake Taihu region of China

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## Key Words

Nitrogen, exports, vegetable, China, Bayesian network, modelling

## Abstract

Vegetable farms are one of many nitrogen sources adversely affecting Lake Taihu in eastern China. Given the lack of quantitative "cause and effect" relationships and data relating to these systems, we developed a conceptually based Bayesian network to investigate and demonstrate causal relationships and the effects of different mitigation strategies on nitrogen (N) exports from vegetable farms in the Lake Taihu region. Structurally, the network was comprised of one primary transport factor, one primary source factor, three post-mobilisation mitigation strategies and three output factors.

In general the network suggests that N exports are more sensitive to transport factors (i.e. runoff volumes) than source factors (i.e. fertiliser application rates) although the cumulative effects of excessive fertiliser were not considered. Post-mobilisation mitigations such as wetlands and ecoditches appear to be particularly effective in decreasing N exports, however their implementation on a regional scale may be limited by land availability. While optimising N inputs would be prudent, the network suggests that better irrigation practice including improved irrigation scheduling, using less imported water and optimising rainfall utilisation would be more effective than simply limiting N supply in achieving environmental goals.

## Introduction

Lake Taihu is in the lower Changjiang (Yangtze) river delta. As China's third largest freshwater lake (2,334 km$^2$ and depth of approximately 2 m), Lake Taihu is an important source of domestic, industrial and agricultural water, assists water quantity regulation and provides transport, aquaculture and tourism services to one of China's most developed areas. Intensive vegetable production is one of many agricultural industries aiming to mitigate their contribution to the non-point source pollution, particularly N, entering Lake Taihu. While deterministic models have been used to simulate N exported from agricultural systems and minimise nutrient exports, in this instance linking agricultural management to N exports and downstream effects is particularly difficult. There are few empirical relationships on which to base a deterministic model of N exports and there is a general lack of parametric information relating to the site specific characteristics.

Bayesian networks (Pearl 1988) are an alternative to conventional modelling. In summary, Bayesian networks provide a graphical representation of "cause and effect" relationships with the strength of the interdependencies (causal links) represented as conditional probabilities. The "nodes" represent variables with defined properties called "states" (i.e. values), and directed links (also called arcs which pass from the parent node to the child node) are used to represent dependencies between variables. Dependencies are quantified in a Conditional Probability Table (CPT) associated with each node which considers all combinations of parent node states. The probability distributions defined in the CPT's are referred to as ***prior*** probabilities and relate to the general properties of the environment (i.e. region) and system (i.e. type of farm) to which the Bayesian network applies. As evidence of state values is received for specific nodes (i.e. the attributes of specific farms are identified), they are added into the network by selecting the appropriate state value (i.e. giving that state a probability of 100%). The resulting ***posterior*** probability distributions for the remaining nodes in the network, in particular, for a set of query nodes (i.e. N exports), are computed based on basic laws of probability. Consequently, as evidence is added to a network in the form of node values (states), the possible outcomes that the network represents do not change, only the relative probabilities of those outcomes. The changes in the relative probabilities of states (and related mean estimates for nodes with numerical values) before and after evidence of state values is added to the network, reflect projected differences between specific systems and the expected "average" for the system under consideration. These projected differences from "average" can be used to compare and contrast a range of different scenarios for both the prognostic and diagnostic analyses.

The aim of this study was to develop a Bayesian network that could be used to investigate and demonstrate causal relationships and the effects of different mitigation strategies on N exports from vegetable farms in the Lake Taihu catchment of eastern China using Xingeng Village farm as a case study. Given the lack of quantitative "cause and effect" relationships and data relating to these systems a relatively simple conceptual model and conceptually based mathematical equations, rather than experiential data, were used to populate the CPTs.

*Network development process*

Xingeng Village is located in Wangting town in the Xiangcheng district, in the northwest corner of Suzhou city (N 31°26', E 120°28'). The village uses 6.7 ha for organic vegetable production, of which *c.* 4 ha is covered by poly-film greenhouses, *c.* 0.7 ha by multi-span greenhouses, *c.* 1 ha by insect nets, and *c.* 1 ha is exposed land. The average annual temperature in Xingeng Village is 16°C (range 38 to -5°C) and annual rainfall *c.* 1,200 mm with the period between March and August accounting for about 65% of that total. The vegetable growing area receives *c.* 650 kg N/ha and *c.* 220 kg P/ha annually.

The network development drew heavily on the processes used to develop similar networks for other industries (McDowell *et al.* 2009; Nash *et al.* 2010). To constrain intra-annual variation the network was conceptualised using an annual time-step. At Xingeng Village farm vegetable production areas are surrounded by drainage channels approximately 1 m deep. Consequently, the network was conceptualised as applying at the "plot" scale, where a plot is defined as a hydrologically isolated production area. Using this definition plots on Xingeng Village farm varied between 0.024 and 0.288 ha in size.

The network was developed using NETICA, version 4.08 (Norsys Software Corp., Vancouver, Canada) software. Where possible, quantitative data (i.e. rainfall records) and deterministic equations (i.e. generally derived from conservation of mass), were used. The *Wetland Efficiency* equation was adapted from Kadlec and Knight (1996) and modified for local conditions. In the absence of sufficient data to develop statistical relationships, both *Nitrogen Concentration* and *Gaseous Emissions* estimations were conceptualised based on the "expected" rather than measured interactions between factors (Equation 1).

$$y = K_1 + K_2/(1 + \exp(-\eta)) \qquad \text{Equation 1}$$

Where:

| | |
|---|---|
| y = *Gaseous Emissions, Nitrogen Concentration* (dependant variable) | $K_1$ = Minimum value |
| $K_2$ = Maximum value | $\eta = K_3*(\text{Runoff}-K_4)+K_5*(x - K_6)$ |
| $K_3$ = Estimated parameter | Runoff = *Runoff* (independent variable) |
| $K_4$ = Estimated parameter | $K_5$ = Estimated parameter |
| x = *Potential Nitrogen Load* or nitrogen additions (independent variable) | $K_6$ = Estimated parameter |

As there was no comprehensive data set that could be used for formal validation, the network was assessed by examining a limited number of case studies, and comparing the network output with the expectations of experts familiar with these systems and the available literature.

*Network analyses and application*

The primary network (i.e. with no specified *posterior* probabilities, Figure 1) represents the "average" expectations for vegetable farms in the Lake Taihu region. The network suggests that *Nitrogen Exports*, *Nitrogen Concentrations* and *Export Efficiency* are respectively (mean ± standard deviation), Medium-Low

(110±55), Low (15±9) and Very-Low (20±15) for those farms. The standard deviation being ≥50% of the estimated mean values for output, and many other key nodes, suggests that care needs to be taken when using the network. Importantly, the lack of quantitative information that could be used to develop the network, in addition to the error estimates, suggest that the network should be used primarily for analyses of general trends rather than absolute predictions of nodal values, especially at the margins (Nash and Hannah 2011).

*Nitrogen Concentration* was most sensitive to *Total Nitrogen Exports* and visa versa, reflecting the computational relationship between the two. *Runoff* and *Imported Water* were the next most important factors, presumably reflecting the direct relationship between transport factors and *Nitrogen Concentration*. *Potential Nitrogen Load*, which was a parent node, had less effect on the *Nitrogen Concentration* than *Imported Water.* This implies that management of imported water is probably the best way to optimise N concentrations. The low sensitivity of *Nitrogen Concentration* to *Total N Additions* compared to *Gaseous Emissions* highlights the potential importance of denitrification in these systems.

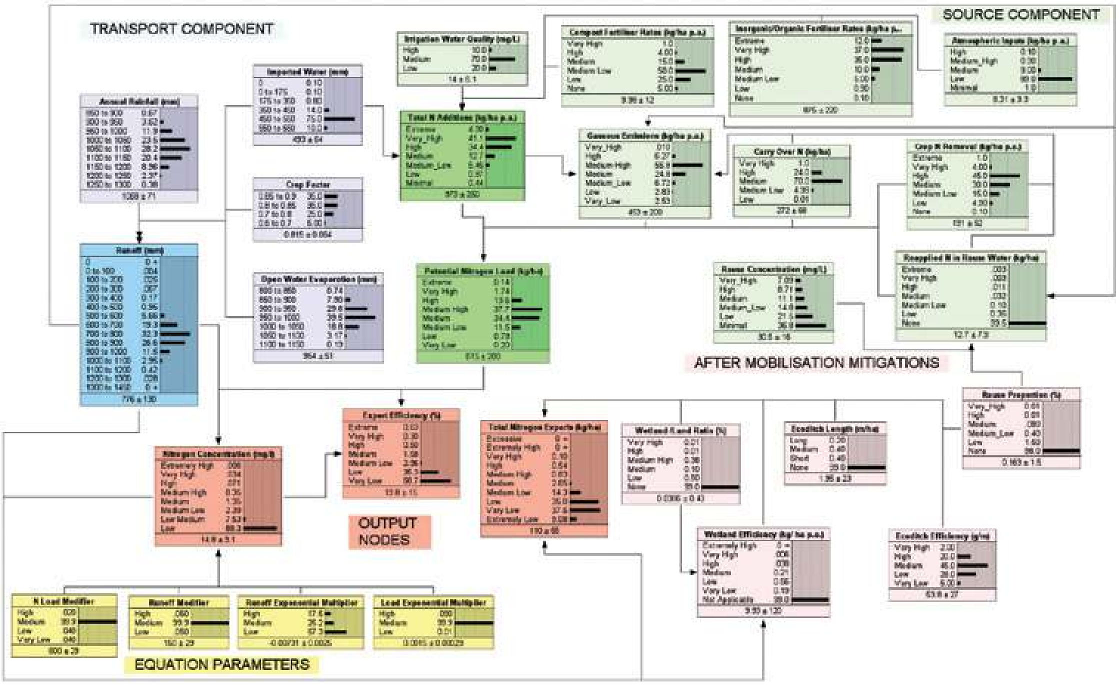


**Figure 1: A Bayesian network of nitrogen exports from vegetable production.**

Xingeng Village farm was used to investigate the properties of the network and the effects of different farming systems. The network suggests that generally, water exiting plots on Xingeng Village farm has a Low-Medium *Nitrogen Concentration* (32±9), the farm has Low *Export Efficiency* (25±11), and Very-Low *Total Nitrogen Exports* (47±33) at the whole of farm scale (i.e. where After Mobilisation Mitigations are included). These figures are generally consistent with snapshot monitoring (mean 19.7 mg N/L mean and range of 9.6 to 39.6 mg N/L) while *Total Nitrogen Exports* in particular compares favourably with other farms in the region. Interestingly, comparing current practice with the upper and lower limits for the parameters in the conceptual equation relating *Runoff* and *Potential Nitrogen Load* to *Nitrogen Concentration* had a relatively minor effect on the states of the Output nodes.

Compared to current practice, the network suggests that an absence of After Mobilisation Mitigations is likely to be associated with similar *Nitrogen Concentration* and *Export Efficiency*, but significantly increased *Total Nitrogen Exports* (i.e. Medium-Low compared with Very-Low). The network suggests that 40-50% drainage reuse does not compensate for the loss of the other mitigations as there would probably be an increase in *Total Nitrogen Exports* compared to current practice (i.e. Low rather than Very-Low). The observation that the changes to *Total Nitrogen Exports* were in proportion to the *Reuse Proportion* suggests that the additional N in reuse water has little overall impact on N exports. The network also suggests that a maximal *Wetland/Land Ratio* (10%) has a similar effect on *Total Nitrogen Exports* to 50% drainage reuse. This appears reasonable given that the reuse water is assumed on average to have a Low to Medium-Low N concentration (*c.* 30 mg N/L). Overall, the network suggests that ecoditches are the most effective of the After Mobilisation Mitigations. Ecoditches (i.e. vegetated drains) appear to be capable of achieving similar results to the current suite of reuse, wetland and ecoditch mitigations.

Source management is an alternative strategy that can be used to mitigate N exports. The network suggests that for farms similar to the Xingeng Village farm, reducing fertiliser applications from Medium (500-700 kg N/ha) to Low (0-300 kg N/ha) has a small effect on *Nitrogen Concentration* (i.e. Low as compared to Low-Medium) but is probably less effective than the After Mobilisation Mitigations in lowering *Total Nitrogen Exports* (i.e. Low as compared to Very-Low). This is consistent with the relative insensitivity in the short-term of the Output nodes to Source nodes as identified in the sensitivity analyses. However, being based on an annual time-step, it is noteworthy that this network does not consider the cumulative effects of nutrient inputs.

## Concluding discussion

This paper outlines the development of a conceptual Bayesian network examining N exports from vegetable production. The outputs from the network make sense and, with the possible exception of *Gaseous Emissions,* are generally consistent with the limited literature that is available from the target area. Data regarding *Gaseous Emissions* is equivocal. Where drainage losses of N have been estimated in these systems conservation of mass would suggest that losses of gaseous N are significantly greater than that measured as nitrogen dioxide and di-nitrogen oxide. The conceptual equation relating *Runoff* and *Potential Nitrogen Load* to *Nitrogen Concentration* is also important. However, comparing network estimations using the upper and lower parameter limits that might be expected suggests the equation parameters have relatively minor effects on the states of the Output nodes.

In terms of resource allocation, this study would suggest that After Mobilisation Mitigations are particularly effective in decreasing N exports. Given that water storages exist in many parts of the target area, but there is limited availability of land, ecoditches and reuse systems appear useful ways to mitigate N exports and may be integrated with wetlands and/or other agricultural pursuits (i.e. fish farming). However, these are farming systems where consideration needs to be given to the interactions between factors. Factors including water and soil pH, N concentration and N species (i.e. ammonium concentration) would be important in such systems and affect the processes and therein the relationships in a Bayesian network developed to describe their impact. Importantly, there appear to be limited opportunities for unexpected outcomes with post-plot mitigation strategies.

At the plot scale, limiting N inputs is often seen as the simplest and easiest way to reduce N concentrations in drainage. However, the relationships between *Runoff* and other factors, especially *Gaseous Emissions*, and the high N requirements of vegetable crops suggests that such a simple solution, while helping mitigate N exports, may not be the best option as it ignores the complex interactions occurring in these systems. Better irrigation practice including improved irrigation scheduling, using less imported water and optimising rainfall utilisation, when coupled with nitrification inhibitors, may be more effective than simply limiting N supply, in achieving environmental goals. This is not to argue that optimising N fertiliser application rates is unimportant. Rather, it is to argue that these are farming systems and, as the network amply demonstrates, the optimum combination of mitigation strategies will depend on the farm in question.

## Acknowledgements

The authors would like to acknowledge the financial support of AusAID for this project. The authors would also like to acknowledge the assistance of Tom Callender, Enquan Huo, Hua Chen, Li Jun, Feiqin Meng, Hua Pan, regional officials and the people and farmers of Xingeng village for their invaluable contribution to the project and their warm hospitality.

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## Soil nutrient budgets of Australian natural resource management regions

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The amounts of N, P, K and S applied as fertilisers and removed as agricultural commodities were calculated using ABS data for 2007-8 and 2009-10 for all Australian NRM regions.

The mass of P in commodities was 43% and 50% of mass applied in 2007-8 and 2009-10 (net input of 190 kt P and 170 kt P) nationally. Regions with (commodity P)/(fertiliser P) < 0.3 and > 10 kt P net input in both years were Murray, Murrumbidgee, Lachlan, and Glenelg Hopkins.

Nationally, the mass of K in commodities was 2.5 and 3.1 times mass applied in 2007-8 and 2009-10 (net export of 196 kt P and 256 kt P). Regions with > 10 kt K net export and (commodity K)/(fertiliser K) >10 in both years were Wimmera, and Northern and Yorke.

The mass of S removed in commodities was 49% and 62% of the mass applied in 2007-8 and 2009-10 (net export of 91 kt S and 62 kt S) nationally. Strong net addition of S in both survey years occurred in South-East (SA), South-West (WA) and Glenelg Hopkins.

Nitrogen balance depends on the method used to calculate biological nitrogen fixation. Simply using fertiliser amounts, regions with strong net input of N were Port Phillip and Westernport, West Gippsland, Hawkesbury-Nepean, Wet Tropics, and Condamine.

These balances will be contrasted with land management practices and soil test values, as well as the 1992-1997 Land and Water Audit. Implications for nutrient management research and policy will be discussed.

# Monday 3 December

## SOIL FERTILITY AND SOIL CONTAMINANTS (NUTRIENT LOSS/MANAGEMENT)

*Presented Posters*

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# Changes in soil and soil water phosphorus and nitrogen after cultivation

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**Abstract**

Un-tilled dairy pasture has the potential to release more phosphorus to the environment than a regularly ploughed pasture. In this paper we report the initial results of a study comparing the effects of cultivation, phosphorus (P) fertiliser (10, 35, and 100 kg P/ha) and two types of vegetation (ryegrass (*Lolium perenne*) or ryegrass mixed with clover (*Trifolium repens*)) in a randomised complete block design. Soil P was measured in soil samples taken from depths of 0-20 mm and 0-100 mm and soil water P measured on the the 0-20 mm samples. In all cases, the P concentrations (Olsen P, Colwell P, Total P, $CaCl_2$ extractable P, dissolved Reactive P, and total dissolved P) in the top 20 mm declined with ploughing. Dissolved reactive P measured in the soil water was 70% less overall in the ploughed plots compared with the unploughed plots; and by 35 weeks after P treatments the decrease in dissolved reactive P was 66%. The N data were more equivocal with the effects of the fertiliser and pasture treatments being inconclusive. The data suggest that ploughing can lower the risk of P exports from intensive dairy farms in the trial area at least in the short term.

**Key Words**

Mouldboard ploughing, P stratification, dairy, pasture

## Introduction

Excessive phosphorus (P) in surface waters is a major environmental issue in Australia. In many freshwater ecosystems, P limits primary production and excessive P inputs contribute to eutrophication and the development of cyanobacterial blooms. This is especially true in the Gippsland Region of south-eastern Australia, and agricultural enterprises, particularly dairy farms, contribute to excessive P concentrations.

Numerous studies have demonstrated that, surface-applied fertilisers, decaying plant material and wastes from grazing animals increase soil P and N concentrations and lower P adsorptive capacity at the soil surface, thereby increasing P export potential. Cultivation is one way of lessening soil nutrient stratification, increasing P adsorption near the soil surface and potentially lowering P exports. For example, studies into laser grading, an extreme form of cultivation, have been carried out in the Gippsland area. In the initial study, three years after laser grading, surface soil P adsorption had increased, soil test P decreased, soil water P and N concentrations decreased and total P (TP) and total N (TN) in irrigation runoff decreased by 41% and 36%, respectively (Nash, Webb *et al.* 2007b).

Pastures used for dairying in Gippsland are commonly a mix of ryegrass (*Lolium spp.*) and clover (*Trifolium repens*). Clover fixation of atmospheric nitrogen enhances soil N fertility, but clover requires a higher soil P (Olsen P of > 15 mg/kg) compared with ryegrass (Olsen P > 12 mg/kg). This suggests that it may be possible to grow adequate ryegrass pasture at lower soil P, and hence lower P export potential, if N fertiliser replaces the N fixing function of clover. The lower required soil P concentration would thereby enhance the benefits of cultivation in mitigating P exports. In this study we examined the effects of destratification of soils by mouldboard ploughing along with the effects of three P fertiliser application rates on pasture that is either a monoculture of ryegrass or mixed sward of ryegrass and clover over a period of 12 months.

## Material and Methods

The experimental site was located near Poowong, Victoria, Australia (-38.2782°, 145.7404°) in an undulating low hills landscape with a slope of 4%. The soil is a Grey Dermosol, of fine sandy clay loam texture. This site was a managed dairy pasture and, in the five years prior to this study being implemented, approximately 270 kg/ha single superphosphate, 130 kg/ha muriate of potash, and 100 kg/ha of urea were applied annually. The soil at a depth of 0-100 mm has a mean Olsen P of 41 mg/kg, 132 mg/kg Colwell P, and a total N of 4400 mg/kg which would classify it as having moderate to high soil P fertility and moderate P sorption which is typical of soils and farms in this region. The average annual rainfall in the

Poowong area is *c.* 1100 mm. A comparison of the 0-20 mm and 0-100 mm soil test results show that prior to ploughing the soil had higher concentrations of P nearer the surface, for example, 182 and 132 mg Colwell P/kg for 0-20 and 0-100 mm respectively. Generally, N concentrations were also higher near the surface. However, $NO_3^-$ concentration was lower at the 0-20 mm depth (5.4 mg N/kg) compared to the 0-100 mm depth (7.3 mg N/kg). This is possibly due to the effects of leaching (Nash, Halliwell *et al.* 2002) moving N to lower depths following high spring rainfall and denitrification.

The experiment had a randomised complete block design with 12 treatments consisting of the complete factorial combinations of two types of sod preparation (mouldboard ploughed or unploughed), two types of vegetation (ryegrass monoculture or a mixed sward of white clover and ryegrass) and three rates of phosphorus fertiliser (10, 35, 100 kg/ha). The trial site (c. 0.5 ha) was divided into three blocks of twelve plots (12 . 6 m). Blocks were arranged with a 3 m buffer between each block. The effect of cultivation, vegetation, and P fertilising levels were analysed by ANOVA.

Plots were grazed twelve times over the nine months following pasture establishment with 500 cows for about 45 minutes. After grazing, manure was immediately removed and followed by mowing the to leave pasture 80 mm high to promote even growth in keeping with local farming practice. All grass clippings were removed off-site.

Soil was sampled four times per year to collect 50 cores/plot of 0-20 mm depth and 10 cores/plot at 0-100 mm depth. Soils were not sampled in the four week period following fertiliser application. Soil samples were analysed by standard methods for moisture, Olsen P, Colwell P, Calcium chloride-extractable P ($CaCl_2$ P), total P (TP), total N (TN), ammonium N ($SNH_4^+$), nitrate N ($NO_3^-$), P buffer index with Colwell fertility correction ($PBI_{+ColP}$), Skene potassium (Skene K), available sulphur (CPC S, calcium phosphate plus charcoal extractable sulphur), total carbon (TC), and oxidisable organic carbon (OOC, Walkley-Black method). Soil water was analysed for dissolved reactive P (DRP, <0.45μm, measured within 24 hours), total dissolved P (TDP), total dissolved N (TDN), ammonium N ($WNH_4^+$) and nitrate/nitrite ($NO_x$) N using standard methods based on flow injection analysis.

**Results and Discussion**

After ploughing and sowing, but prior to the P treatments being implemented, ploughing lowered concentrations of P ($P < 0.001$) but increased concentrations of N ($P < 0.001$). For example, DRP and TDP concentrations in soil water for ploughed and unploughed plots were 0.1 and 1.3 mg/l DRP and 0.6 and 3.3 mg/l TDP respectively. The equivalent N data for ploughed and unploughed plots were 60.0 and 30.5 mg/l N for TDN, 10.9 and 2.8 mg/l N for $WNH_4^+$ and 33.2 and 8.2 mg/l N for $NO_x$. Decreased P concentrations are consistent with other studies (Mathers and Nash 2009; Nash, Webb *et al.* 2007a) and probably reflect the relocation of topsoil away from the soil surface, in addition to increased P adsorption where ploughing brought fresh clay material to the surface. Increased N concentrations following ploughing are consistent with organic matter disturbance and aeration stimulating the microbial population resulting in increased ammonification and nitrification.

Examining the grand mean concentrations for the period after treatments were applied suggests that available P and N generally decreased. For example, in soil water, TDN ($P = 0.001$) concentrations over the four sampling dates were 19.9, 8.6, 9.4 and 10.8 mg/l N, and $NO_x$ ($P < 0.001$) concentrations were 4.4, 1.8, 0.4 and 1.2 mg/l N. Over the same period, 0-20 mm soil test results for P, TN, and $NO_3^-$ decreased ($P <$ 0.001 for Olsen P, Colwell P, and $NO_3^-$; $P = 0.004$ for TN). Sampling date did not affect $SNH_4^+$.

After P fertiliser treatments were applied, soil water results (Table 1) show that ploughing lowered P ($P <$ 0.001) concentrations. For example, the mean DRP and TDP concentrations in ploughed and unploughed plots were 0.25 and 0.8 mg/l P, and 0.51 and 1.52 mg/l P respectively. N concentrations were lower in the ploughed versus the unploughed plots (TDN, $P < 0.001$; $NH_3$, $P = 0.014$; and $NO_x$, $P = 0.023$). Nitrogen concentrations for ploughed and unploughed plots were 10.0 and 13.1 mg/l N for TDN, 0.4 and 0.5 mg/l N for $WNH_4^+$, and 1.2 and 1.6 mg/l N for $NO_x$. These results are consistent with ploughing having stimulated the rapid decomposition of organic matter and this source of N being exhausted near the soil surface prior to the fertiliser treatments being initiated. The results of the 0-20 mm soil tests were consistent with the soil water tests. For example Olsen P, Colwell P, Organic P and $CaCl_2$ P concentrations decreased for ploughed versus unploughed plots ($P < 0.001$). The respective means were 39 and 66 mg/kg P for Olsen P, 99 and 205 mg/kg P for Colwell P, and 0.5 and 1.8 mg/l P for $CaCl_2$ P.

There were no effects of vegetation on soil water parameters. In the 0-20 mm soil the concentration of $SNH_4^+$ ($P$ = 0.003) was higher for the mixed sward plots (11.0 mg/kg N) compared to the monoculture plots (8.5 mg/kg N), presumably from N fixation by the clover in the mixed sward.

Higher P fertiliser application rates increased the concentration of P in the soil water ($P$ < 0.001). For example, at the P rates of 10, 35 and 100 kg/ha, concentrations were 0.27, 0.36, and 0.91 mg/l for DRP, and 0.62, 0.73, and 1.52 mg/l for TDP. The soil test P results of the 0-20 mm soil and Olsen P in the 0-100mm soil showed similar P trends to the soil water tests ($P$ < 0.001). For example at fertiliser rates of 10, 35, and 100 kg/ha P, Colwell P concentrations were, 122, 137 and 198 mg/kg, and $CaCl_2$ P were 0.66, 0.83 and 1.6 mg/kg for the 0-20 mm samples, respectively. Similar trends have been found elsewhere (Barlow, Nash *et al.* 2005; Robertson and Nash 2008).

**Table 1: Mean concentrations of phosphorus and nitrogen in soil water after fertiliser treatments were applied. Values in brackets represent significance at 5% level. Values before the comma compare ploughed versus unploughed within the fertiliser treatment, while values after the comma compare fertiliser treatments within the same sod preparation. Within a column, items with the same letter are not significantly different.**

| Treatments | | Concentration (mg/L) | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | DRP | TDP | TP | TDN | TN | NH3 | NOx |
| 10 kg/ha P | Unploughed | 0.55 (a, a) | 1.2 (a, a) | 1.6 (a, a) | 13.2 (a, a) | 14.6 (a, a) | 0.6 (a, a) | 1.5 (a, a) |
| | Ploughed | 0.13 (b, a) | 0.3 (b, a) | 0.4 (b, a) | 11.3 (a, a) | 12.0 (a, a) | 0.4 (a, a) | 1.2 (a, a) |
| 35 kg/ha P | Unploughed | 0.62 (a, a) | 1.3 (a, a) | 1.7 (a, a) | 13.2 (a, a) | 14.7 (a, a) | 0.5 (a, a) | 1.7 (a, a) |
| | Ploughed | 0.21 (b, b) | 0.4 (b, a) | 0.5 (b, a) | 9.8 (b, a) | 10.4 (b, a) | 0.4 (a, a) | 1.3 (a, a) |
| 100 kg/ha P | Unploughed | 1.5 (a, b) | 2.3 (a, b) | 2.9 (a, b) | 13.0 (a, a) | 14.6 (a, a) | 0.5 (a, a) | 1.5 (a, a) |
| | Ploughed | 0.55 (b, c) | 1.0 (b, b) | 1.2 (b, b) | 9.0 (b, a) | 9.7 (b, a) | 0.4 (a, a) | 1.2 (a, a) |

Aside from the main treatment effects, there were also treatment interactions. For all soil water P analyses, there were Sample Date by P Fertiliser Rate interactions where the treatment effect of fertiliser declined with time ($P$ = 0.02 for TN, $P$ < 0.001 for others, see Figure 1). A Vegetation by Cultivation interaction for $NO_x$ ($P$ = 0.003) was also observed. $NO_x$ in the mixed sward plots accounted for the main cultivation effect, while in the monoculture plots, unploughed plots had reduced $NO_x$ concentrations compared to ploughed plots. With the notable exceptions of $SNH_4^-$ or $NO_3^-$, the 0-20 mm soils test results had a similar Sample Date by Cultivation interaction ($P$ < 0.001) to the soil water analyses. For P and TN, the difference between ploughed and unploughed plots decreased over time. For example, Olsen P concentration in ploughed and unploughed plots was 42 and 87 mg/kg P in May 2010, and 31 and 52 mg/kg in November 2010.

**Concluding Discussion**

This study suggests that cultivation where soil P has accumulated in the upper layers, especially in high impact areas, could be a useful tool for reducing nutrient loss risk if incorporated into farm management at convenient times, for example, when pasture needs renovating or when a summer forage crop is to be planted. However, care needs to be taken to ensure that the potential for soil erosion and consequent export of particulate P does not outweigh the advantage of ploughing to decrease dissolved P exports. Further evaluation of the overall risk needs to be undertaken.

In this study, a mouldboard plough was used for tilling the soil, rather than a power harrow which is more commonly used for commercial cultivation in the Gippsland. To optimise the environmental effects of cultivation it would appear, at least conceptually, that a combination of these implements would provide optimal results. Further research is needed to identify the most appropriate way of incorporating the environmental and agronomic needs of pasture based grazing.

## Acknowledgments

The authors would like to acknowledge the support of Brett and Jodie Loughridge on whose property this work was undertaken and thank them for their enthusiastic help. The authors also wish to thank John Gallienne (John Gallienne & Co Pty. Ltd.) for his advice, Murray Hannah who provided help with the statistical analyses and Dr Hayley Castlehouse who assisted with the establishment of the project.


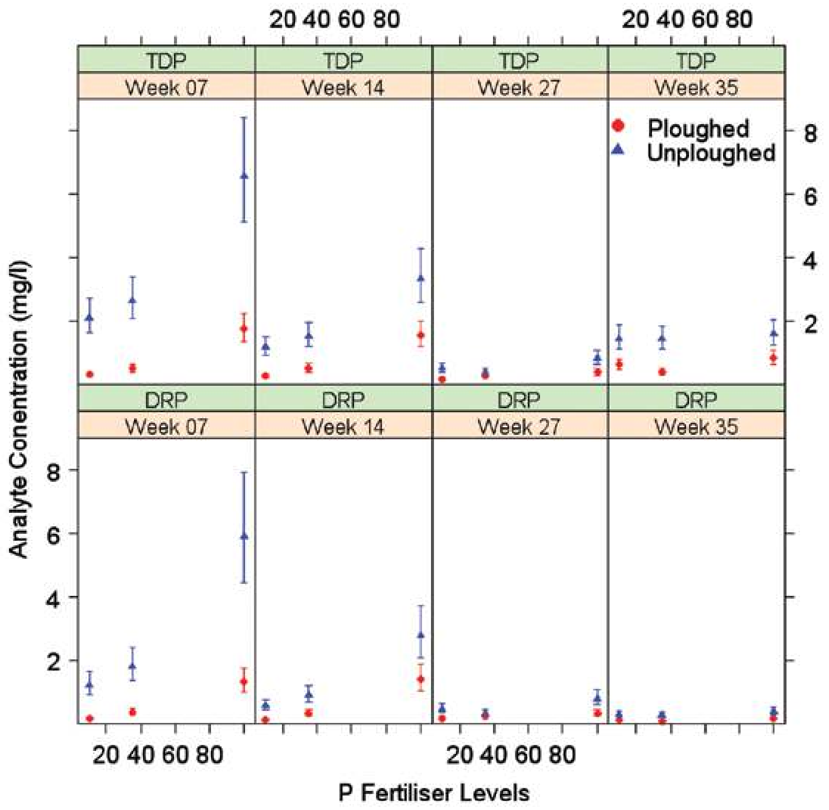


**Figure 1: Mean TDP and DRP concentrations in soil water for ploughed and unploughed treatments at different P fertiliser application rates. Week represents the number of weeks since fertiliser treatments began. Error bars represent significance at 5% level.**

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# Improving nutrient management on dairy farms in northwest Tasmania

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There has been a significant reduction in soil phosphorus and sulphur over the past six years on dairy farms in the Togari and Brittons swamp areas in the northwest of Tasmania. There are still many paddocks with extremely high values of plant available soil phosphorus (Olsen P > 80), and soil potassium has increased. The decrease in soil phosphorus will have saved farmers money in reduced fertilizer application as well as reducing the risk of nutrient loss to waterways. These economic and environmental gains have been due to a combination of effective RD & E by TIA, Dairy Tas, Dairy Australia and Cradle Coast NRM, plus we were helped by a cyclical downturn in milk prices that focused the farmers' attention on input costs. The RD & E has been locally focused with effort put into relevant, farm specific information for each individual farmer. Individual farm results for the nine farms re-sampled in 2011 show that seven farms had a significant reduction in plant available soil phosphorus but only two had agronomic optimum mean values (20 – 30 Olsen P). Comprehensive on-farm soil testing has identified nutrient 'hot spots' that need to be addressed by improved nutrient distribution around each farm. The use of comprehensive soil testing and whole farm nutrient budgets has raised the profile of nutrient management in the Montagu catchment and has allowed farmers to maintain or reduce soil nutrients to optimum levels for maximum pasture production.

# Leaching of $NH_4^+$-N and *Escherichia coli* from stony soils following application of dairy shed effluent

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Irrigation of dairy shed effluent (DSE) onto land is an integral part of New Zealand's farming practice. However, the use of inappropriate soils can result in contamination of surface and ground waters. A gap in our knowledge is the ability of stony soils to renovate animal wastes. Replicates of three stony soils were collected from the Mackenzie basin as intact soil lysimeters 500 mm diameter and 700 mm high. The soils had either gravels to the surface, 300 mm fines over gravels or 600 mm fines over gravels. To generate breakthrough curves (BTC) DSE (25 mm) spiked with Br (2000 ppm) was applied to the soil cores followed by continuous artificial rainfall, for one pore volume (PV), at 5 mm $h^{-1}$. Soil leachates were analysed for the Br, $NH_4^+$-N and *Escherichia coli*. Br concentrations in leachates peaked at 0.5 to 0.8 PV. The BTC for $NH_4^+$-N was similar to that of Br for soil cores with gravel to the surface. Concentrations of $NH_4^+$-N in leachates were <1 ppm for soils cores with overlying fines, except for one core exhibiting preferential flow characteristics. In this core, with 300 mm fines overlying gravel, peaks in $NH_4^+$-N and *E. coli* were detected at <0.1 PV. For all other soil cores *E. coli* concentrations in soil leachates were typically low.

We anticipate that the preferential flow characteristics of these soils could be vulnerable to land management practices, particularly under irrigated intensive dairying.

# Variability of phosphorus content in dryland pasture topsoils of the Moe River catchment (Gippsland, Victoria)

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Topsoil phosphorus is an important driver of pasture productivity, but can also lead to nutrient losses from runoff events. Phosphorus content of topsoils varies with land management and soil properties. Whilst the impact of land management practices (such as fertiliser management, tillage operations, and grazing intensity) on soil phosphorus content is well known, the interactions of land management with soil characteristics are less well characterised and reported. The aim of this research was to explore soil and land management interactions in phosphorus content of dryland pasture topsoils of the Moe River catchment (Gippsland, Victoria). Soils in the region were grouped into three major groups according to soil hydrology (profile permeability and structure) and topsoil organic matter content. Four land management systems (beef, extensive dairy, medium-input dairy, and intensive dairy) were defined to reflect different fertiliser regimes and levels of grazing intensity. Topsoil samples were taken on at least 12 representative paddocks of each soil-land management combination, for a total of 138 paddocks sampled. Samples were taken at two depths (0-2 and 0-10 cm) to verify stratification in phosphorus topsoil content. Samples were analysed to measure Colwell dissolved phosphorus ($P_{Col}$, mg $kg^{-1}$), Phosphorus Buffer Index (PBI), total phosphorus (TP, mg $kg^{-1}$), and phosphorus soluble in $CaCl_2$ ($P_{CaCl2}$). Statistical variability of each phosphorus measurement in relation to the three factors (soil group, land management, and soil depth) will be presented and discussed in the light of soil productivity and risk of nutrient runoff losses.

# The effect of urban biochar on phosphorus fractions in an acid dairy soil

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A laboratory experiment was conducted to determine the effect of Urban Biochar produced by pyrolysis of a 15% biosolid / 85% greenwaste feedstock on the phosphorus (P) fractions of a dairy pasture soil from Poowong East, Victoria. The soil was an acidic Brown Dermosol and samples were obtained from the top 0-10 cm from two areas of the same paddock with contrasting Olsen P status (P low (PL), 9 mgP $kg^{-1}$ and P high (PH), 200 mgP $kg^{-1}$). Soils were incubated in small containers (250 cm3) at 25 oC for 45 days at constant moisture content. Biochar was mixed with the soil at levels equivalent to 0, 10, 30 and 50 tonne ha-1. At the end of the incubation the treatments were fractionated according to the Hedley method, which included $H_2O$-P, $CHCl_3/NaHCO_3^-$P, NaOH-P, HCl-P and finally residual P. The initial distribution of P differed between the soils with around 40 % (PL) and 80 % (PH) of the P in inorganic forms. The Biochar treatments were found to have a liming effect on both soils. Addition of 50 t $ha^{-1}$ Biochar increased soil pH by 0.2 (PL) and 0.26 (PH) relative to the control. Biochar addition significantly increased the labile forms of P in PL soil. However Biochar addition significantly decreased water soluble P in PH soil. The results suggest that Urban Biochar may have a role in increasing P availability in P limited environments and decreasing P loss in high P environments.

# Use of vegetable oil to enhance bioaccessibility of coal tar constituents for soil remediation

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A lack of bioaccessibility of hydrophobic organic contaminants (HOC) in soils is one of the most decisive factors for bioremediation efforts. Natural lipids such as spent vegetable oil can be applied economically as biocompatible extractant to increase the bioaccessible fraction of coal tar constituents. The present study investigates whether this approach will mobilize heterocyclic (hPC) and alkylated (aPC) aromatic coal tar constituents from soils, in addition to polycyclic aromatic hydrocarbons (PAH).

The bioaccessible contaminant share was determined using a silicone based passive sampling device. Fresh and spent vegetable oils were used to release a variety of aromatic contaminants from historically contaminated soils. Extraction and degradation yields were compared with bioaccessibility data from passive sampling. Bioslurry degradation tests revealed an increase in the bioaccessible and thus biodegradable PAH (16 EPA PAH) fraction up to four rings upon addition of 1 to 5% (w/w) vegetable oil. Moreover, PAH alkylation and substitution induces different sorptive behaviour in heterocyclic and alkylated PAH as opposed to homocyclic PAH. Accordingly, a pronounced difference in extraction efficacy of aPC and hPC vs. EPA PAH by spent and fresh oils were observed in lab-scale experiments.

The results underline the application of vegetable oil being a promising approach to assist in the bioremediation of soils contaminated with coal tar products. Consequently, this approach can be efficiently applied also to treat alkylated and substituted tar oil constituents aside the traditional homocyclic 16 EPA PAH. It offers support to microbial degradation of contaminants as well as novel prospects for the recycling and eventual disposal of spent vegetable oils.

# Monday 3 December

## SOILS AND INFRASTRUCTURE DEVELOPMENTS

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# Natural capital and New Zealand's *Resource Management Act* (1991)

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## Abstract

Quantifying natural resources as natural capital and the valuation of the ecosystem services that flow from natural capital stocks are emerging areas of science. Are these developing concepts compatible with current resource management legislation? Can these ideas be used in judicial proceedings to protect natural capital and maintain the portfolio value of nature's ecosystem services? We describe two recent cases in New Zealand where natural capital concepts were used in the Environment Court to protect land from peri-urban creep and to protect receiving water quality through the allocation of a nutrient discharge allowance to land. Results have been mixed, with prospects appearing good.

**Key Words:** environmental legislation, ecosystem services, judicial process, life-supporting capacity, natural resources

## The *Resource Management Act* (1991)

In 1991, New Zealand passed an innovative and omnibus legislation to deal with environmental and developmental issues: the *Resource Management Act* (RMA). Section 5 details that the '... purpose of this Act is to promote the sustainable management of natural and physical resources'. The Act would enable "... managing the use, development and protection of natural and physical resources to enable people and communities ... to provide for their social economic and cultural well being and for their health and safety while ...

a) sustaining the potential and natural physical resources ...
b) safeguarding *the life-supporting capacity* of air, water, soil, and ecosystems; and
c) avoiding, remedying, or mitigating any adverse effects of activities on the environment."

How then is the RMA applied? Regional Councils are required to develop Regional Plans, which Section 63 states are for '... the purpose of the preparation, implementations and administration of regional plans to assist a regional council to achieve the purpose of [the] act".

Likewise, District Councils are required to prepare district plans (Section 72) "... to assist territorial local authorities to achieve the purpose of [the] act". The Environment Court hears appeals against regional and district plans. We discuss two appeal cases in which natural capital arguments were used by the respondents in these appeals: one against a District Council; the other a Regional Council

### *Natural Capital*

Natural capital can be defined as the earth's stocks of natural material and energy, which is akin to the RMA's prescient words of 'natural and physical resources'. The sum of these natural capital stocks can be referred to as our ecological infrastructures. From natural capital stocks there are flows of ecosystem services, which can be defined as the beneficial flows of goods and services between natural capital stocks, or between these stocks and humans.

The RMA's goals even presaged those of the Millennium Ecosystem Assessment released by the United Nations in 2001 (http://www.maweb.org/documents/document.429.aspx.pdf ), wherein the four types of ecosystem services were linked to the constituents of human well-being (Figure 1).

It would seem then, given the apparent link between 'natural capital' and 'natural resources', plus that between 'life supporting capacity' and 'ecosystem services', that arguments in Court hearings based on natural capital and ecosystem services (Figure 1) are compatible with the goals and applications of the RMA.

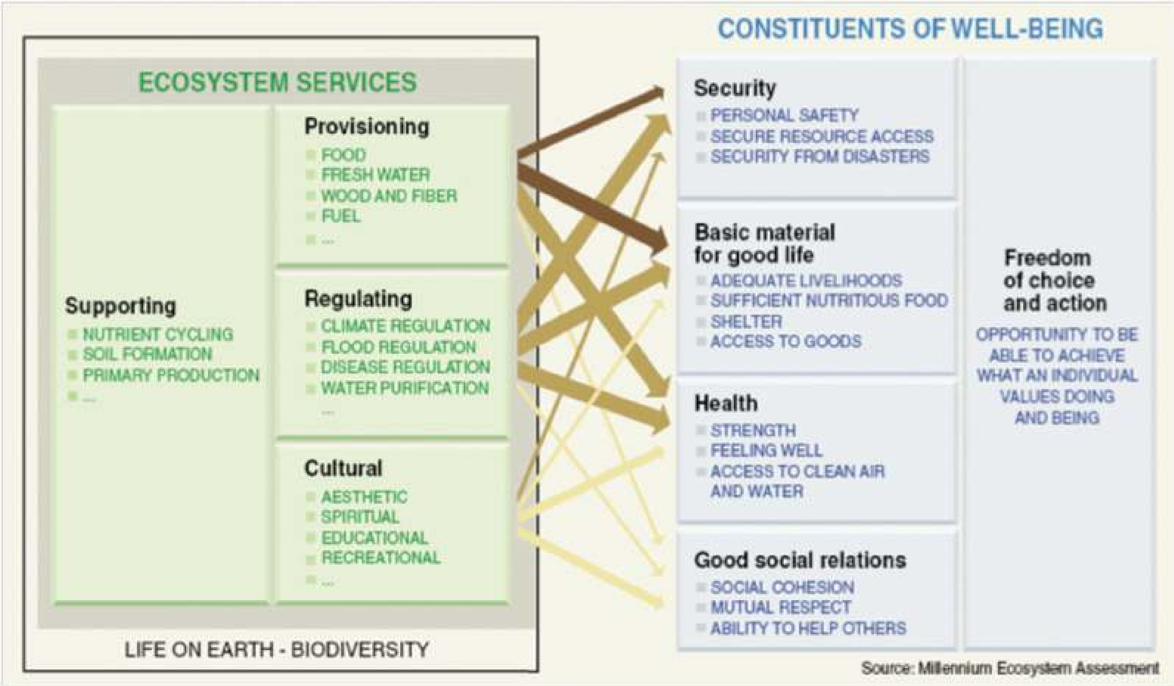


**Figure 1: Linking ecosystem services to the constituents of human well-being (MEA, 2001).**

In other jurisdictions, particularly with Europe, ecosystem-services thinking has been guiding policy development in a patent manner, along with the burgeoning development of regulatory controls based on natural capital and ecosystem services. This can be seen in the United Kingdom with the agency DEFRA (Department of Environment Food and Rural Affairs) (http://www.defra.gov.uk/environment/natural/ecosystems-services/ ), as well as in the European Union (http://ec.europa.eu/environment/integration/research/newsalert/pdf/20si.pdf ).

However, here in New Zealand, it would seem that there have only been a few attempts to use natural-capital thinking to sustain natural resources and safeguard life-supporting capacities. We describe two attempts. The first is in relation to peri-urban expansion of a city on to horticultural land and involved a district plan, whereas the second is in relation to farming as a controlled activity to protect receiving-water quality in a regional plan.

*Peri-Urban Expansion*

The hardware retailer Bunnings' purchased 4 ha of orchard land on the outskirts of Hastings and sought a consent to build a large-format store. The Hastings District Council (HDC) appointed independent commissioners to hear the Bunnings' application. In July 2009 the Commissioners declined Bunnings' consent and stated that "... if these soils are as valuable as described, their loss should be avoided". Bunnings' appealed that decision and the appeal was heard during March 2011 (NZEnvC ENV-2009-WLG-000182). One of us (BEC – hereafter referred to as 'I') acted as an expert witness for the respondent, the HDC. I argued that "... we cannot afford to lose such valuable natural capital assets, whose presence is needed for their ecosystem services, and whose use will be needed to enable the horticultural industries to realise their strategic goals, and whose functioning will continue to enhance the life-supporting capacities of the Heretaunga Plains (Paragraph 99)", as required by the Hastings' District Council's District Plan for the Heretaunga Plains.

An expert witness for Bunnings argued that "... the concept of natural capital value was still an emerging discipline" and that the concept of natural capital was in his view "... unhelpful in terms of the issue confronting this Court. That issue is, as expressed in the RMA, 'safeguarding the life supporting capacity of the air, water, soil and ecosystems' ". Bunnings' lawyer in his closing address considered that "... there is no quantitative or qualitative analysis of the ecosystem services at the site other than in relation to food production".

The judgement was cautious and noted that "... we do not propose to enter that [natural capital] debate ... but it seemed to us that Dr Clothier took a somewhat more holistic approach to assessment of the value of the soils of the site". The judgement noted that although the "... loss of 4 ha of Plains land is insignificant in itself the wider policy implications are significant." The appeal was declined, and costs awarded to the HDC. So although the Judge and his two Commissioners did not directly buy into a natural capital argument, they did note a holistic view was needed. Holism is, it seems to us, an ecosystem services approach in principle, at least in a quasi-judicial sense. It would appear then that some headway has been made for the use of natural capital reasoning in judicial proceedings in relation to 'safeguarding the life-supporting capacity of … soil, and ecosystems" (RMA, Sect., 5). Yet, precedence in a legal sense would not seem to have been registered. But the indications are somewhat positive.

*Nutrient discharges & natural capital*

The Manawatu-Wanganui Regional Council (known as Horizons) embarked on a second-generation Regional Plan, called the Proposed One Plan (POP) several years ago. They decided that intensive agriculture would become a controlled activity requiring of resource consent. Part of that consent would be a nutrient discharge limit for intensive farming.

A team from SLURI (the Sustainable Land Use Research Initiative of 3 CRIs) was contracted to determine the nutrient discharge limit. We (ADM and BEC were part of the SLURI team) first looked into the loading of nutrients in the river so as to work back to a loss rate from the various farm types in the catchment. Then we used the Land Use Capability (LUC) class as a proxy for natural capital. "Attainable potential livestock carrying capacity" is akin to an 'ecosystem service' as noted in the extended legend of Land Use Capability Classification. By using the OVERSEER™ model to determine the nutrient leachate levels from this attainable potential livestock carrying capacity, it was possible to link nitrate leaching to LUC class, as shown in Figure 2 below.

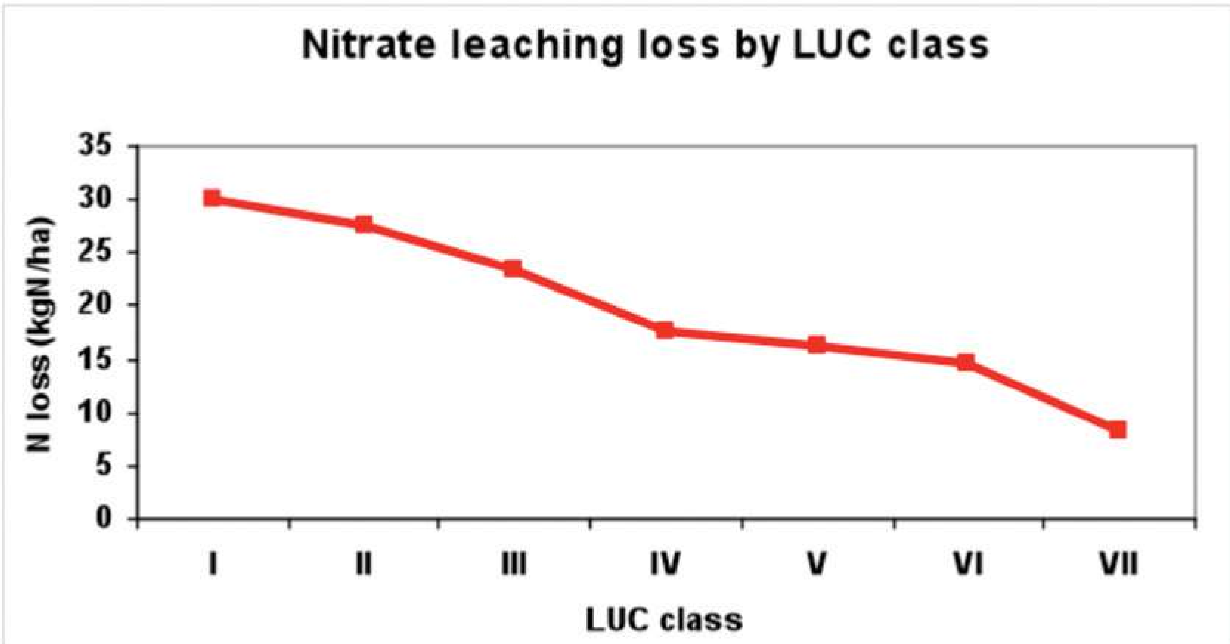


**Figure 2. The nitrate leaching loss from the attainable potential livestock carrying capacity of the various Land Use Capability (LUC) classes. LUC is a proxy for natural capital of this self-regulating grass-legume pasture system.**

The numbers in Figure 2 were entered into the notified version of the POP (NVPOP) as Table 13-2, which formed the basis for the resource consent Rule 13-1. Given the large number of submissions on the NVPOP, Horizons set up a panel of Commissioners to assess the One Plan. The Hearings took place in early 2010. The natural capital approach using LUC did not fare well in the Commissioners' decisions. They decided ...

".... Table 13.2 is not appropriate for existing dairy farms for the following reasons:

(a) Dr Mackay's "natural capital" approach is not based on technological changes that have enabled farmers to lift productivity levels since the 1980s;

(b) For existing farms, the "natural capital" approach therefore ignores existing land use and existing levels of farm production. That is inequitable and impracticable;"

The Commissioners, it seems to us, had missed the point about the natural capital of a self-regulating system of pasture growth, and the role of the built capital of technologies such as irrigation, synthetic fertilisers, and imported feed stocks.

In dispensing with Rule 13-2, they sought instead a "... nutrient management plan [that] would require the implementation of practicable and affordable "best management practices" (BMPs) that are designed to reduce nitrogen leaching." Ironically then, no designated discharge allowance was even provided, just a vague BMP. However, the Commissioners did consider that the natural capital LUC approach should apply to new dairy conversions.

The Commissioners' decisions' version of the POP (DVPOP; Manawatu-Wanganui Regional Council, 2012) was appealed by five appellants and one Section 274 party. Some appellants wanted restoration of the NVPOP, and others sought the complete removal of the LUC approach from the POP. The appeal was heard in the Environment Court in early May 2012, and judgement has been reserved. That decision is awaited with interest.

## Conclusions

Although it would seem that the purpose of New Zealand's *Resource Management Act (1991)* is very much aligned with natural-capital and ecosystem-services thinking, there is as yet only meagre evidence that this approach has directly influenced decision making in hearings. Early days?

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# Evapotranspiration on a mine cover system in central New South Wales, Australia

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Mining results in the production of waste rock and tailings material which, depending on geology and production techniques, can be sulfidic or contain elevated levels of metals. When such wastes are exposed to oxygen and water, hazardous drainage may occur, contaminating adjacent ecosystems. Australian legislation requires mining companies to prevent these impacts. In arid and semi-arid areas evapotranspiration (ET) covers are widely proposed to minimize water ingress into hazardous waste. It is assumed that the cover can store occurring precipitation in the soil and, eventually, release the soil water through plant transpiration (T) and soil evaporation ($E_{soil}$). However, these parameters have seldom been measured on ET cover systems constructed from benign waste rock.

The open top chamber technique was used to measure plant water loss from two native shrub species and soil evaporation on two cover systems in central New South Wales, Australia. Plant water use was determined by the availability of soil moisture, as opposed to climatic demands. Plot ET was dominated by $E_{soil}$ due to low plant coverage (20-25 % of ground area) and the contribution of transpiration was estimated to be negligible until plant coverage exceeded 50 %. This research emphasizes that the contribution of vegetation to ET losses from constructed cover systems is considerably less than assumed by many practitioners. Therefore, for the application of vegetation as a relevant parameter for the control of the water balance under semi-arid conditions, the maximum sustainable plant coverage under climatic and soil moisture conditions has to be carefully considered.

## Introduction

Waste material from gold mining processes can result in the formation of acid mine drainage, metal leaching or cyanide-bearing tailings solutions. Negative impacts on the environment from these hazardous processes will prevent the relinquishment of mine leases. In semi-arid areas evapotranspiration (ET) cover systems are frequently suggested for minimising water ingress into hazardous mining waste. These covers can be constructed from benign (non-acid-forming) overburden or waste rock and are designed to maximize soil water storage, as well as evaporation from soil and transpiration from vegetation established for this purpose (Williams 2011).

To assess the suitability of ET covers to prevent deep drainage into potentially hazardous tailings on a gold mine in central New South Wales two cover designs were tested. The effectiveness of ET cover systems can be evaluated by application of the water balance equation, where deep drainage depends upon the quantities of precipitation, surface runoff, the storage capacity of the cover system and water losses through evaporation from soil ($E_{soil}$) and plant transpiration (together constituting ET).

Knowledge about the contribution of vegetation on water losses of ET cover systems is limited, especially for covers constructed from benign waste rock material. However, a better understanding of plant water losses is crucial to enhance cover design and cover performance modeling. This study investigated the components of ET on recently established native vegetation covers, as well as the critical drivers for ET for cover systems in dry climates.

## Material and Methods

Evapotranspiration was measured on two ET cover systems in semi-arid central New South Wales, with a uniformly distributed annual rainfall of approximately 400 mm. The cover systems were constructed from benign waste rock material in 2002 with nominal thicknesses of 1.5 m and 2.0 m (TP1.5 and TP2.0 hereafter). Topsoil was applied in 2008 and vegetation on the trials established late that year, predominantly from seedbank germination and seeds carried in to the area.

In April 2011 vegetation density was determined as foliage projective cover (FPC) (Specht 1981), assessed every meter on two 40m transects per cover. Evapotranspiration measurements were conducted on two species which were relatively abundant on both plots, namely *Senna artemisioides* (DC. Randell) and *Sclerolaena birchii* (F. Muell). In addition, evaporation from soil was determined on one bare spot per cover. ET and $E_{soil}$ readings were taken with a calibrated open top chamber, similar to the system described by Hutley et al. (2000).

On both cover trials, two plants of each species were selected, as well as a bare spot, resulting in five locations per cover plot. Measurements started at 6:30 hrs (sunrise), using 10 min every two hours for each location until 18:30 hrs (sunset). Measurements alternated daily between TP1.5 and TP2.0 from 04/04/2011 until 08/04/2011.

**Results and Discussions**

FPC measurements showed relatively low vegetation coverage, with on average 19 % and 26 % vegetation cover on TP1.5 and TP2.0, respectively. As vegetation density is generally in equilibrium with soil moisture (Specht and Specht 1999) lower vegetation coverage has to be expected in water-limited climates, such as with annual total rainfall of about 400 mm.

Daily ET rates of *Senna artemisioides* (3.0 - 4.8 mm day$^{-1}$) generally exceeded water use by *Sclerolaena birchii* (1.4 – 2.6 mm day$^{-1}$). Evaporation from soil on TP1.5 (0.9 mm day$^{-1}$) slightly exceeded $E_{soil}$ on TP2.0 (0.7 mm day$^{-1}$) and remained stable on the last two measurement days on each cover trial.

The atmospheric demand for evapotranspiration can be expressed through the vapour pressure deficit (VPD). If ET would be solely driven through the atmospheric demand a linear relationship between VPD and ET would be observed (Takagi, Tsuboya *et al.* 1998). As a result the level of water shortage can be demonstrated through the daily time courses of relationships between ET and VPD, where the gradient of the ET/VPD curve indicates the proximate availability of water to transpiring surfaces. The extent of hysteresis of the daily course indicates availability of water for ET, with wider courses of ET/VPD curves indicating increasing water shortage.

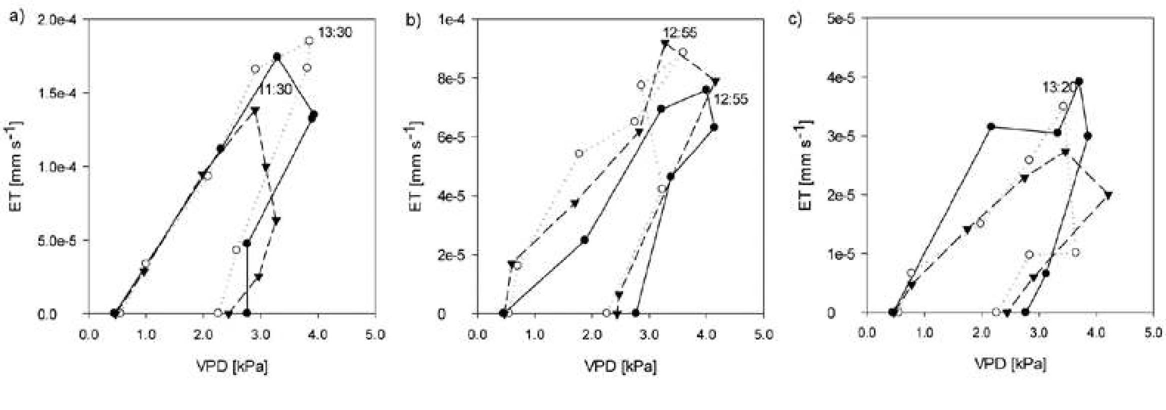


**Figure 1. ET/VPD curves for (a) *Senna artemisioides*, (b) *Sclerolaena birchii* and (c) bare soil on three days (solid circles), five days (open circles) and seven days (solid triangles) after a simulated 10 mm precipitation event on TP2.0.**

Figure 1 shows that three days after simulated 10 mm precipitation (solid circles) water loss was determined through climatic demand until midday, while water availability determined ET rates thereafter. Five days (open circles) and seven days (solid triangle) after the simulated precipitation event, moisture availability drove ET rates after late morning. The similar slopes of the curves in the early morning indicated a rehydration of the plants over night, enabling constant ET rates in the mornings (Fig. 1 a, b). $E_{soil}$ showed similar ET/VPD curves (Fig. 1c) compared to the plant species (Fig. 1a, b), reflecting the importance of night-time fluxes (liquid, vapour) in the soil.

For predicting the efficacy of an ET cover system, whole plot ET is an important parameter. To extrapolate measured chamber ET to whole plot ET both vegetation coverage and plant composition has to be taken into account. Therefore, plot ET was calculated as the weighted average of bare soil evaporation and plant evapotranspiration.

To assess the influence of vegetation composition, plot ET was calculated according to three scenarios: a) equal percentages of each of the two investigated plant species in vegetation cover; b) 70 % of vegetation coverage of the more dominant species, *Sclerolaena birchii*; c) 80 % coverage of the more dominant species (Table 1).

**Table 1. Plot ET for three scenarios of vegetation coverage of the dominant species[1]**

| Site | ET [mm day$^{-1}$] for species ratio of vegetation cover of | | |
|---|---|---|---|
| | 50% | 70% | 80% |
| TP1.5 | 1.2 | 1.1 | 1.1 |
| TP2.0 | 1.4 | 1.3 | 1.2 |

[1] *Sclerolaena birchii*

Table 1 illustrates that plant composition was a minor factor in plot ET. This was mainly due to the low vegetation coverage, making the lower $E_{soil}$ rates the main component of plot ET. This importance of $E_{soil}$ in semi-arid areas has also been observed by Stannard and Weltz (2006).

To determine the threshold when vegetation composition becomes an important factor for plot ET, plot ET was calculated for increasing plant densities in three scenarios, a) no weighing of vegetation for plant component, b) 70% of plant component taken up by species with higher water use, and c) 70% of plant component taken up by species with lower water use. These scenarios suggest that for vegetation composition to substantially influence daily plot ET, the minimum vegetation density must exceed 50 % coverage (Fig. 2).

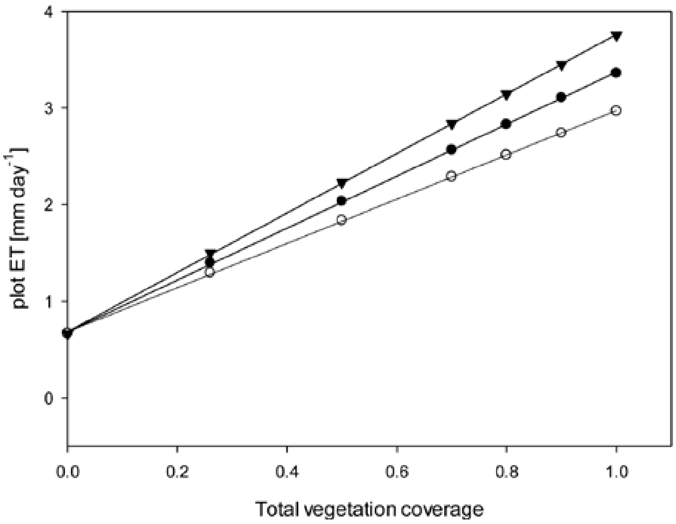


**Figure 2. Relationship of plot ET to plot vegetation density for no weighing of plant component between plant species (solid circles), 70 % of plant component formed by species with higher water use (solid triangles), 70 % of plant component formed by species with lower water use (open circles) in April 2011.**

As mentioned previously, in a sustainable plant community vegetation density is on average in equilibrium with available soil moisture. As a result vegetation density in dry ecosystems is generally low. ET cover systems are only suitable for arid and semi-arid areas, which must lead to low plant coverage on these systems. It therefore has to be accepted that the role of vegetation on ET cover systems is limited.

## Conclusions

Evapotranspiration cover systems can only be employed in arid and semi-arid areas and where rainfall intensity is low and the seasonal distribution is uniform. However, these areas are marked by water limitations, which have important consequences for the cover system.

In these regions diurnal evapotranspiration including evaporation from soil is determined by moisture availability, resulting in lower daily evapotranspiration values compared to a climatically driven system. In addition, as moisture availability determines plant coverage, low vegetation densities will occur in sustainable vegetation communities. These low vegetation densities limit the role of vegetation in the determination of plot ET from cover systems.

It is therefore crucial that practitioners understand the limited role of vegetation upon water losses from ET covers and ensure that water ingress into hazardous waste is prevented through a combination of controlled surface and subsurface runoff and water holding capacity of the cover system.

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# Coal combustion products: Assessment using acid sulfate soil methods

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The Acid Sulphate Soil (ASS) characterisation SPOCAS suite was adopted as a model for assessment of coal combustion products which include fly-ash (FA) and furnace bottom ash (FBA), representing power stations in New South Wales, South Australia, Queensland and Western Australia. The testing of coal combustion products using an ASS model was proposed because the analytical methodology interprets acidity with acidity units of mol $H^+$/t and includes interpretations for soluble and exchangeable portions in solution. Selected Australian sources of FA have been reported with low pH ranges of 4 and are also known to have low pH buffering capacity and elemental characteristics which vary with coal source. Acidity and a combination assessment of key elements Ca, Mg, S and $SO_4^{2-}$ by the SPOCAS ASS testing methodology was an opportunity to develop a greater understanding of the soluble, exchangeable and potential acid production of coal ash products, and particularly for those with pH values below 7. We suggest the ASS laboratory analysis and particularly the TAA and net acidity measures are the application of a well-defined methodological test suite to characterise coal ash materials and will be useful to identify an acidity hazard.

**Key words**
Coal combustion products (CCPs), fly ash, furnace bottom ash, low pH, SPOCAS

## Introduction

Coal combustion products[1] (CCPs) are by-products of coal-fired electricity production. The [coal] ash when produced is dry and hot, with physical and chemical characteristics controlled by the coal, the boiler and its operating conditions and post combustion parameters of dust collection (Kutchko and Kim 2006). A low pH value for a coal ash is a common property for many of black coals (Jankowski *et al.* 2006). Low pH ash of less than 7 pH units is considered unsuitable for land application in civil projects (NSW EPA 2010). In 2011 the Ash Development Association of Australia (ADAA) part of its annual Environmental Monitoring Program (EMP) expanded normal suite of assessments for pH, electrical conductivity and metal elements (As, Sb, Ba, Be, B, Cd, Cr, Co, Cu, Pb, Mn, Hg, Mo, Ni, Se, Ag, V, Zn) with an inclusion of alkali metals (Ca, Mg, K, Na); P and S, soluble elements $Cl^-$, $SO_4^{2-}$ and B ($CaCl_2$ extractable) and analysis assessment used to identify acid sulfate soil (ASS).

Potential ASS is soil typically waterlogged and rich in pyrite which has not been oxidised. Any disturbance exposing these soils to air (oxygen) can lead to the development of extremely acidic soil layers or horizons with field pH values of <4, and actual acid sulfate soils (Ahern *et al.* 2004). Evaluation of these soils is by measuring reduced inorganic S compounds in soils and sediments using a chromium reduction method ($S_{CR}$ method) (Sullivan *et al.* 2004) or with the suite of tests called Suspension Peroxide Oxidation Combined Activity and Sulfur (SPOCAS) (McElnea *et al.* 2002). SPOCAS methodology was chosen for this study because it interprets acidity with acidity units of mol $H^+$/t and includes the soluble and exchangeable portions Ca and Mg in solution. It is a step-by-step methodology interpretation of the maximum theoretical acid production from reduced inorganic sulfur, peroxide oxidisable sulfur, or total oxidisable sulfur will indicate a management and an associated risk (Ahern *et al.* 2004). Applied to CCP on the basis that sulfur and sulfate are predominant elements of coal fly ash (Ward *et al*. 2006; Sear *et al*. 2003) analysis by SPOCAS was proposed as a useful assessment of environmental risk and identification of coal ash sources that may require neutralisation of their net acidity as indicated by pH below 7.

## Method

ASS testing methodology by SPOCAS was adopted to characterise a selection of coal ash samples FA and FBA representing power stations in New South Wales, South Australia, Queensland and Western Australia. Annually the Ash Development Association of Australia[2] collects, submits for assessment and publishes the findings within an Environmental Monitor Program Report (EMP) of members' CCPs. EMP assessments

[1] Coal combustion products include fly ash (FA), furnace bottom ash (FBA), boiler slag (BS), fluidized-bed combustion (FBC) ash, or flue gas desulfurization (FGD) material produced primarily from the combustion of coal or the cleaning of the stack gases. The term coal ash is interchangeable.

[2] http://www.adaa.asn.au/environmental-monitoring-reports.php

were first conducted in 1993. The aim of the EMP is to collate and interpret the analytical knowledge on its members' CCPs through a coordinated annual sampling, analysis and reporting program. During the 2011 program seventy two (72) samples collected using methods defined in AS1141-3.1-Methods for sampling and testing aggregates (Standards Australia, 1996) were submitted for chemical analysis including total concentrations of heavy and alkali metals and salts of B and S. Standard moisture content (%), conductivity (uS/cm) and pH tests (1:5) were undertaken. Sixteen (16) samples were then submitted for SPOCAS assessment as eight (8) FA and eight (8) FBA. Results for all key, trace and soluble elements were reported as mg/kg and where required tabulated to percentage (%). The SPOCAS reporting units were mol $H^+$/t for all measures of acidity, %S for sulfur related compounds such as net acidity sulfur units, and total oxidisable sulfur (TOS), with %Ca and %Mg of acid reacted solutions. Values of kg $CaCO_3$/t were reported for liming rate value. Properties of pH were compared between source for FA and FBA and by the relationship between pH values and titratable actual acidity (TAA). Comparisons included TAA and peroxide-oxidation solutions (TSA) and $SO_4^{2-}$, total S and TAA %S. Net acidity for all ash pH values was compared with total %Ca, %Mg, %S, $\%S_{POS}$ and %TOS.

**Results and Discussion**

*pH and acidic properties*

Eight (8) FA and eight (8) FBA samples selected for the SPOCAS assessment were from power stations located in New South Wales, South Australia and Western Australia. The representative pH value of these coal ashes were between pH 3.4 to 12. All FBA had pH values of 7 or above, except one (1) at pH 5.2. Six (6) of the eight (8) FA had pH values below pH 7 with two (1) at pH 12 (Table 1). FA when acidic was observed to have lower pH than the FBA from the same source power station and when alkaline and above pH 9.5, observed to be of higher pH than the FBA. Similarly a neutral or alkaline FBA was associated with a FA that was either acidic or highly acidic. Consequently, from the same source power station FA and a FBA is not a product with the same pH characteristic. Comparative pH values are presented in Table 1.

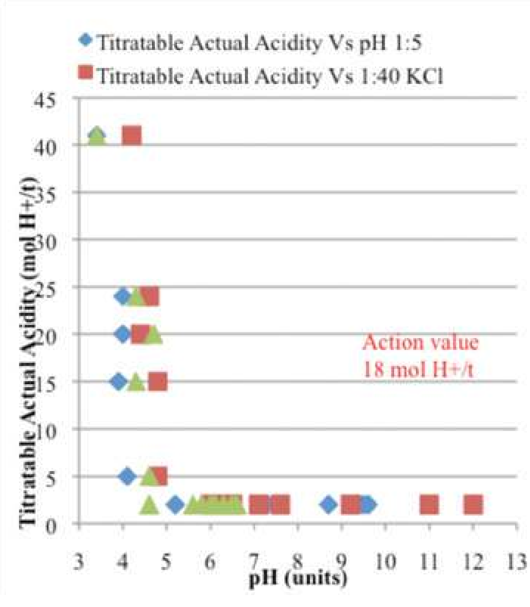


**Figure 1: Plot of TAA values furnace-bottom ash and fly-ash against pH of 1:5, TAA, TSA solutions - Diamonds are pH 1:5 standard test, Squares are pH value of the 1:40 KCl solution, Triangle as pH of the peroxide oxidised solution.**

Initial assessment of ASS properties was by comparison between the FBA and FA as a plot of the values of TAA against pH values the standard pH test result at 1:5, the pH of the 1:40 KCl and the pH values of the oxidised solution (Fig. 1). In this case the relationship between TAA and net acidity was 0.9481 $r^2$. TAA ranged from 5 to 40 mol $H^+$/t only when below pH 5.0 $_{(1:5)}$. For an ash with pH value above pH $5.0_{(1:5)}$ the corresponding TAA was below limit of detection at <2 mol $H^+$/t. Of interest is that the pH of the TAA and TSA solutions was slightly higher than the standard pH test result. This comparison identifies (i) that TAA was only quantifiable and above detection limit when pH of the analytical solutions was below pH 5.0; (ii) pH of the oxidised solution was grouped below pH 7.0 and (iii) pH of the KCl solution was indicative of the standard pH test. We suggest that acidity is acidity of the soluble/exchangeable component because post oxidation the pH value was stable, indicating a lack of residual acidity associated with an oxidised sulfur with dominant of $SO_4^{2-}$.

A primary element of coal ash is sulfur primarily as sulfate (Ward *et al*. 2006) and for this study the relationship was total S and $SO_4^{2-}$ values at $r^2 = 0.9447$. We also identify that acidic ashes with high S values had elevated S in TAA solutions and that the %S of a TAA solution for the FBA was below detection and did not mimic the trends of the $\%SO_4^{2-}$ (Fig. 2). Thus a $SO_4^{2-}$ concentration and its associated acidity was observed for an acidic FA but not an acidic FBA. Subsequent comparison of the net acidity as total oxidisable sulfur (%TOS) with total elemental %S identifies that the level of analytical detection for the SPOCAS suite at 0.02 % or 200 mg/kg was higher than the standard method reported for total S testing at

100 mg/kg. Consequently, we recommend - (i) that future testing with coal ash product uses a lower level of detection, and similar to the test for total elemental S and (ii), the consistency of analytical detection is considered also for Ca and Mg for an equitable comparison to acid-reacted Ca and Mg.

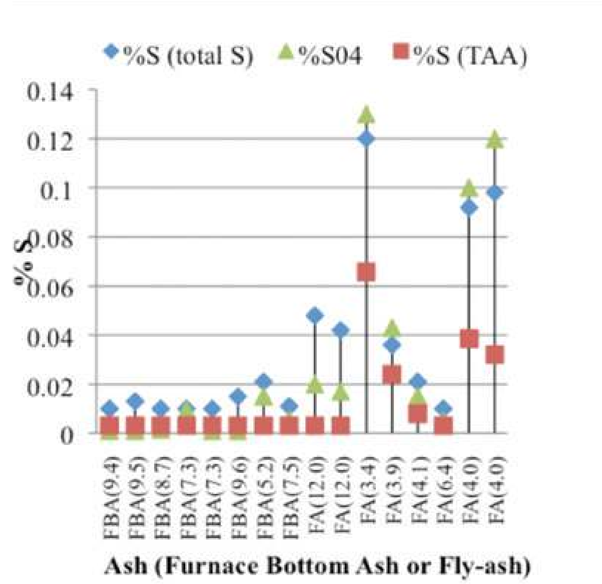


**Figure 2: Plot of % values of S (sulfate, total S and TAA) for FBA and FA across pH range 3.4–12) - Diamonds are %S (total), Squares are %S of the TAA solution, Triangles are %SO42-.**

*Net Acidity*

A SPOCAS assessment provides a calculation to determine the amount of pure $CaCO_3$ needed to neutralise the net acidity, which in this case is identified within the soluble-exchangeable component of an acidic FA. This liming capacity is determined by the addition of results for Potential Sulfidic Acidity (mol $H^+$/t) + Actual Acidity (mol $H^+$/t). For the coal ashes tested, the calculated result for the most acidic FA, with a pH value $_{(1:5\ dH2O)}$ of 3.4 pH units was 2.18 kg $CaCO_3$/t to fully neutralise its acidity. When a factor of safety of 1.5 is included the liming capacity to neutralise the net acidity was 3.2 kg Ag Lime/t, or 3.2 g Ag Lime / kg of the acidic fly ash (Table 1). Acidity hazard is identified using net acidity values at the level of 18 mol H+/t. Consequently only fly ash would be considered a potential risk. And because some fly-ash below pH 5 may not have a significant lime requirement we suggest net acidity and liming capacity is included with any CCP assessment.

**Table 1: FA and FBA by pH (least to highest), %Ca, %Mg, Net Acidity, CaC03/t, %TOS, %S, %SPOS**

| Line | Source | Ash | pH $_{(1:5)}$ | Ca | Mg | Net Acidity | Lime | TOS | $S_{POS}$ | S |
|---|---|---|---|---|---|---|---|---|---|---|
| | | 1,2 | | (%) | (%) | (mol $H^+$/t) | kg $CaC0_3$/t | (% S ) | (%) | (%) |
| 1 | DE WW | FA | 3.4 | 0.042 | 0.014 | 42 | 3.2 | 0.07 | <0.02 | 0.12 |
| 2 | DE MP | FA | 3.9 | 0.016 | 0.014 | 15 | 1.1 | 0.02 | <0.02 | 0.036 |
| 3 | Verve CP | FA | 4 | 0.42 | 0.18 | 24 | 1.8 | 0.04 | <0.02 | 0.092 |
| 4 | Verve MP | FA | 4 | 0.5 | 0.18 | 22* | 1.6 | 0.03 | <0.02 | 0.098 |
| 5 | Tarong | FA | 4.1 | 0.0019 | 0.0052 | <10 | <1 | <0.02 | <0.02 | 0.021 |
| 6 | Callide | FA | 6.4 | 0.035 | 0.036 | <10 | <1 | <0.02 | <0.02 | 0.01 |
| 7 | Eraring | FA | 12 | 0.89 | 0.06 | <10 | <1 | <0.02 | <0.02 | 0.042 |
| 8 | DE VP | FA | 12 | 1.1 | 0.11 | <10 | <1 | <0.02 | <0.02 | 0.048 |
| 9 | Verve CP | FBA | 5.2 | 0.046 | 0.018 | <10 | <1 | <0.02 | <0.02 | 0.021 |
| 10 | Tarong | FBA | 7.3 | 0.0073 | 0.0059 | <10 | <1 | <0.02 | <0.02 | 0.01 |
| 11 | DE MP | FBA | 7.3 | 0.0078 | 0.0029 | <10 | <1 | <0.02 | <0.02 | 0.01 |
| 12 | Verve MP | FBA | 7.5 | 0.031 | 0.011 | <10 | <1 | <0.02 | <0.02 | 0.011 |
| 13 | DE WW | FBA | 8.7 | 0.036 | 0.0095 | <10 | <1 | <0.02 | <0.02 | 0.01 |
| 14 | Eraring | FBA | 9.4 | 0.2 | 0.064 | <10 | <1 | <0.02 | <0.02 | 0.01 |
| 15 | DE VP | FBA | 9.5 | 0.97 | 0.32 | 14 | 1 | <0.02 | 0.21 | 0.013 |
| 16 | Callide | FBA | 9.6 | 0.11 | 0.15 | <10 | <1 | <0.02 | <0.02 | 0.015 |

[1] FA: Fly-ash , [2] FBA: Furnace bottom ash, * Acidity Hazard action value at 18 mol H+/t equivalent to pH 4.0

Observed from Table 1 was the lowest pH of FA had the highest net acidity (line 1). Based on the work of Killingley *et al.* (2000) a relation with pH has been established between the concentration of alkaline-earth elements, expressed as CaO and MgO in the coal ashes, on the one hand and the proportion of potentially acid-generating sulphate and phosphate on the other (Ward *et al.* 2009). In this case the total %Ca or %Mg is not indicative of acid neutralisation. Also FA at pH 4.1(line 5) can have a low $H^+$ concentration and lime equivalent similar to a FBA at pH $5.2_{(1:5)}$ (line 9). High %Ca is associated with high pH for FA (line 8) but not FBA (lines 14-16). Thus net acidity and lime requirement was only associated with FA, when pH 4.00 or less and never for FBA. Of interest is that $S_{POS}$ values are below detection, except for one FBA, which indicates that residual sulphides were present in that sample even though pH 9.5.

*Risk Assessment*

In this study the analysis of CCPs by the ASS and SPOCAS methodology, we quantify net acidity and therefore risk, and submit this is a valid proposition with which to interpret risk with use of CCPs for land applications. For an ASS the potential export of acid deemed as an environment risk is the sum of the capacity to create acid and the capacity to render acid ineffective (rather than neutralise) (Mulvey 2004). Mulvey (2004) notes that $S_{POS,}$ represents the maximum amount of acid which can be generated by reduced sulphides in the soil matrix when oxidised, which is the % $H_2O_2$ oxidisable S and total oxidisable S (TOS), if no buffering capacity was available and thus, the TOS and the net acid should have the same value. Applying this same assumption to CCPs in this study, we propose that risk of acidity potential to the surrounding environment when quantified by the $S_{POS}$ values of the SPOCAS assessment confirms low risk or no risk. We suggest that this SPOCAS calculation of sulfur-based acidity identifies the low risk of sulfidic acidity for coal ash materials generally, which is a correct representation for the oxidation process of coal combustion.

## Conclusion

Our hypothesis was that the ASS methodology of analytical assessment by SPOCAS would be an informative model when applied to coal ash. By SPOCAS analysis, we identified that a low pH coal ash of less than pH 7 was not generally indicative of a net acidic risk unless at pH 4.0 and that a FA and not a FBA will have risk potential. For FA and FBA their acidity was as soluble/exchangeable acidity or soluble alkalinity, which is a consistent characteristic of the dominant alumina-silicate, iron silicate elemental structure of all coal ashes (Ward *et al.* 2006), with solubility a consistent factor of alkaline coal ash as assessed for civil applications (Sear *et al.* 2003). We acknowledge that this study represents a small sample size of the possible seventy two (72) coal ash types available for analysis across the ADAA membership. We propose that this model of acid-based accounting is a suitable suite for CCP assessments and is an acceptable method for pH characterisation to identify the need for acid neutralisation by lime application and identify acidity hazard. However, to maximise the information gained using SPOCAS in this assessment the authors will, for completeness, include the direct determination of chromium reducible sulfur rather than rely on a derived calculation because some values may be under –estimated when only using the SPOCAS methodology.

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# Information products produced by the Tasmanian Acid Sulfate Soils Information (TASSI) project

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In 2009 an NHT-funded project was completed by the Department of Primary Industries, Parks, Water and Environment (DPIPWE) in partnership with the three Tasmanian Natural Resource Management organisations. This project identified and mapped >700,000 ha of land with the potential to contain Potential Acid Sulfate Soils (PASS) using GIS modelling techniques. Of this figure 91,500ha has been classified as having "High" probability and will require careful management to avoid and/or minimise disturbance into the future.

The spatial model developed by the project effectively identified landforms that had likelihood to contain PASS in coastal, subaqueous and inland environments. Field calibration/validation and map unit classification were undertaken to National standards (Fitzpatrick *et al.* 2008), and occurred in parallel to the model development. Sampling was also undertaken according to nationally accepted sampling standards (Ahern 1998) and full soil descriptions were recorded at each site according to McDonald *et al.* (1990). ASS soil field tests, acid based accounting and metal analysis were conducted by National Association of Testing Authorities (NATA) accredited laboratories.

The project also published State management guidelines and a range of other information products designed to engage stakeholders and raise awareness of the ASS issue. These information products are located on the DPIPWE website with linkage to the mapping held on www.theList.tas.gov.au

In Tasmania land development pressure, especially in the coastal areas, continues to increase with urban, industrial and agricultural intensification. Planners and project proponents can now refer to the ASS predictive mapping and the management guidelines to help guide their developments.

# Applications of soil science within a local government context

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### Abstract

Local government has traditionally been known for its role in providing community services such as roads, rates and rubbish. This level of government is being asked to undertake more technically complex roles in land use planning, natural resource management, environmental health and waste management where a knowledge of soil science is not only vital but can mean the difference between a successful or unsuccessful development and affect the severity of environmental impacts. The Whitsunday Regional Council is a local government organisation in Central Queensland which undertakes a range of tasks and decision making which require a knowledge of soil science. This report outlines how soil science is relevant within a local government context and how the community can benefit from local governments having soil scientists on staff.

**Key Words:**
Local government, development assessment, waste management.

### Introduction

Local government is the third tier of government in Australia and is primarily involved with land use decision making at the region, town, community and neighbourhood level. Last century local government was regarded as an organisation whose primary responsibility was to provide essential community services ("roads, rates and rubbish"), however, the local government legislative and service landscape has changed and continues to change.

Local government in Queensland is now more than ever being asked to assess development applications which require greater scrutiny against an increasing range of environmental and natural resource management legislation and State planning policies. With the development of Regional Plans in Queensland and incorporation of natural resource management issues, councils are required to incorporate a range of natural resource management concepts, goals and issues into their planning schemes. In recent years there have been more pieces of legislation and policy introduced which require an understanding of soil science in order to fully appreciate how development can be better managed and reduce downstream impacts. However, it should be noted that some Councils have considered soils and natural resource management as important ingredients in their planning schemes. One notable example is the Gatton Shire planning scheme and how it used natural resource data in 1995 (Hecksher, 1996). The Gatton Shire Council used available soil information to produce agriculture suitability maps and identify land suitable for industry. Studies were also undertaken to determine and map soil and land attributes across the Shire. The soil and land attribute mapping was used to determine preferred areas for urban and rural residential land use which were incorporated into the planning scheme.

The Whitsunday Regional Council is located on the central Queensland coast adjacent to the Great Barrier Reef. Like other local governments, it has found resourcing the raised development assessment expectations a challenge. However, having staff with knowledge of soil science has been beneficial in a range of applications including:

- development assessment,
- waste management and landfill management,
- erosion and sediment control from development and building sites,
- coastal management,
- road construction,
- climate change,
- interacting with regional natural resource management organisations, community and schools.

The purpose of this report is to examine in more detail how the Whitsunday Regional Council utilises staff with skills and knowledge in soil science and how the local community can benefit from this involvement.

## Discussion

*Development assessment*

In Queensland, development is primarily regulated under the Sustainable Planning Act (2009). The Queensland government has developed regional plans, which provide guidance to local government planning schemes. There are a number of other pieces of State government legislation which need to be considered by local government during the development assessment process which have an environmental or natural resource protection focus such as the; Environmental Protection Act (1994), Waste Reduction and Recycling Act (2010), Vegetation Protection Act (1999) and Coastal Management and Protection Act (1995). In addition, local government is required to consider State planning polices on acid sulfate soils (SPP2/02), landslips, flooding and bushfire (SPP 1/03), strategic cropping land (SPP 1/11), healthy waterways (SPP 4/10) and protection of high ecological wetlands (4/11).

Development involves the utilisation of land for a specific human use. Most developments involve the excavation and manipulation of the soil to place buildings and infrastructure. While engineering as a discipline is a requirement in almost all types of development, a knowledge of soil science can assist in interpreting geotechnical investigations for landslip hazard, bore logs for the placement of sewage disposal areas and assessment of erosion hazard on sloping development sites. Having an understanding of a soil's nutrient holding capacity can assist in assessing a site's suitability for sewage disposal and determining the risk of groundwater contamination. Soil scientists and geomorphologists have an advantage in interpreting erosion hazard and soil loss potential from development sites (Renschler and Harbor, 2002). The importance of having soil science knowledge when reviewing developments was noted by Semple (1996) who stated that a knowledge of soil science is valuable to those carrying out environmental assessment. Local government staff with a soil science background are in a good position to assess possible soil erosion hazards, and assess the effectiveness of proposed mitigation measures. Knowing the difference between a stable soil and an unstable dispersive soil can make a difference to the design of a development and selection of appropriate impact mitigation measures and selection of site rehabilitation methods (Semple, 1996).

Marina developments in coastal areas, canal estates and urban subdivisions in low lying coastal areas will require the assessment of soils for their acid producing potential. With more and more development occurring in coastal areas, the likelihood of intercepting and excavating acid sulfate soils is increasing. In some coastal marina projects, marine mud has been transported and stored on land. In these situations stringent soil sampling and analysis is required to ensure these materials are treated appropriately or have effective self neutralising capacities. The ability to interpret coastal landforms and bore logs can be useful in determining whether acid sulfate soils need to be considered at the development application stage. The Whitsunday Regional Council has been involved in marina development assessment and assessing infrastructure involving acid sulfate soils.

Knowledge of soil science and geomorphology are useful when considering issues such as climate change, sea level rise and storm surge impacts in coastal areas. Many soil science courses teach students about subjects such as soil formation, geological time, landscape formation and landform formation. Knowledge of past climates, coastal landscape and landform evolution can assist in understanding processes associated with climate change and sea level rise and what may occur along the coast in the future.

Many areas in central Queensland are experiencing an unprecedented increase in development applications, including environmental impact statements and environmental authorities for mining operations and associated transport infrastructure. These land uses also interact with soil resources. The ability to interpret the soils which these land uses intend to excavate and the analysis of how the soil spoils will be managed can be beneficial in ensuring that downstream impacts on waterways are not adversely affected. The Whitsunday Regional Council staff analyse and assess soil information associated with these mining developments and provide advice and comment on possible environmental issues associated with soil disturbance and management.

*Environmental health and waste management*

The management of landfills does not initially conjure up an immediate linkage to soil science. However, landfills often require earth material to cover the waste. The coverage of daily waste during the dry season requires the use of soil, rock and gravel which has a clay content of between 15 and 30%. The clay material in the day cover is used to seal the waste layers and reduce water infiltration. The rockier components of day cover are needed to maintain a driveable surface for waste collection vehicles, particularly in the wet season.

Landfills require capping. Landfill batters are often designed for a 15-25 degree slope. The batter capping requires the placement of non-dispersive clay and topsoil. These batters require progressive application of non-dispersive soil with about 25-45% clay to seal the waste and reduce infiltration. Once a landfill has reached its design life, the plateau is then capped. The State government requires the clay material used for the capping to achieve a permeability of $1x10^{-9}$ m/s (DERM, 2009).

Having a knowledge of soil science can assist in the selection of the most appropriate soil for use in landfill management. The Whitsunday Regional Council staff with soil science knowledge were able to investigate possible soil deposits, and assess their suitability for various landfill uses. The linkage between soil science and waste management has been noted in other areas of the world including the United States (Pennsylvania Association of Professional Soil Scientists, 2011).

Landfills have limits on the type and amount of regulated waste they can receive. Often the operating permits will limit the amount of contaminated soil and other regulated wastes which a landfill can accept within a given year. Knowledge of soil chemistry is useful when determining whether waste can be accepted under landfill licences or not. Having an understanding of nutrient and heavy metal leaching can assist in interpreting groundwater monitoring results. For new landfill cells, the use of geotextile- and clay-lined waste receiving basins is now preferred to reduce nutrient and heavy metal leaching and contamination of groundwater.

*Quarry materials and road construction material*

Soil maps can be interpreted to find important road and other material deposits. Blue metal deposits for road surfaces (typically andesite) can be easily indentified from soil maps. Other useful materials which can be identified from soil maps include sand deposits, binder for rural road base mixes (sandy loams), and calcium carbonate to incorporate into road mixes. Local soil maps in the Whitsunday Regional Council area have been used to short list possible quarry sites and find the nearest soil and rock resources which could be used to construct rural roads.

*Natural Resource Management and community education*

In Queensland, natural resource management institutional arrangements have changed greatly over the last ten years. Regional natural resource management (NRM) groups and other community NRM groups now have a more defined role in working with landholders to encourage improved land management. Local governments in many locations in Queensland have developed a positive working relationship with their regional and local NRM groups. This working relationship extends to corporate arrangements and development of joint projects where staff from both organisations work together on a range of NRM projects. NRM projects which focus on regional ecosystem management require an understanding of not only the botanical attributes but also the soil and geology. NRM projects designed to reduce soil loss from agriculture, catchment land use and stream banks also benefit from a knowledge of soil science. In the Whitsunday region, local government is working with regional NRM groups on projects focusing on erosion from urban areas and soil carbon. Other NRM projects include beach nourishment and determination of strategic cropping land.

Whitsunday Regional Council staff also work with the community and schools on topics including soil science. In 2010 and 2011, Council staff worked with the Proserpine High School year 12 chemistry class on acid sulfate soils. Council staff organised for pits to be dug to display acid sulfate soils and delivered field and class room talks on this topic. Hopefully this interaction with the local high school will lead to future soil science students in university.

*Linkages between university, local government and soil science*

The work disciplines in local government which can greatly benefit from knowledge in soil science include: town planners, environmental planners, engineers, environmental health officers, and waste management officers. Some of the university course units which are relevant within a local government organisation and have a link to soil science include

- soil science and soil conservation
- coastal and fluvial geomorphology
- hill slope geomorphology and landslip
- sedimentology

- acid sulfate soils
- geochemistry, soil physics and soil chemistry
- Australian soils – soil classification
- land capability and suitability

The subjects of a number of Town Planning courses in Queensland and New South Wales have been reviewed. For some Town Planning courses some subjects include aspects of soil science or could expect soil science to be discussed as a topic. The subjects “Environmental Impact Assessment” or “Australian Landscape Processes” from James Cook University, “Catchment to Coast” from the University of New England and “Catchment Processes and Management” from the University of Queensland could all be expected to have some minor soil science content. The inclusion of some of the above courses in town planning, engineering, environmental health and waste management would be well used in these work fields.

## Conclusion

When applications of soil science are discussed, often they over look its use in a local government context. It can be surprising to list the number of applications where local governments use soil science to seek better outcomes for their communities. Knowledge of soil science can assist planners and engineers to ensure developments have reduced impacts on downstream and offshore ecosystems, while environmental health officers and waste management officers can also use soil science to achieve better outcomes for the community and environment. It is a recommendation of this report that soil science should be incorporated into University degrees for town planning, engineering and environmental health (and waste management). There are numerous examples from the Whitsunday Regional Council experience where the council not only has saved money from having a soil scientist on staff but has improved environmental and community outcomes as well.

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# Monday 3 December

## SOILS AND INFRASTRUCTURE DEVELOPMENTS

*Presented Posters*

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# All mixed up: Considering enhanced small-scale variation in mine soils

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Small-scale variation (SSV) of soils is known to be enhanced in mine soils as a result of mechanical movement, soil storage and material mixing, which occurs during mining operations. The enhanced variation at the centimetre scale makes it difficult to characterise and understand mine soil properties. Understanding soils and their use in rehabilitation is fundamental for successful rehabilitation. Thus it is vital to develop more appropriate tools for understanding mine soils for their use in planning, monitoring and implementation of mine rehabilitation.

Alternative classification frameworks, surveying and sampling techniques, sample preparation, calibration and development of new analytical procedures, and alternative statistical models are some of the tools that can be used to better understand these soils. Frameworks can also help land managers determine if SSV is a problem in their soil, and how to consider SSV as part of their rehabilitation programs and ongoing monitoring requirements. Frameworks can consider everything from the aim of rehabilitation through to sampling, analytical and statistical analysis of mine soils. The development of guidelines to consider the unique nature of enhanced SSV in Anthroposols (such as mine soils) is urgently needed for surveying and measuring soil functions and properties. This paper outlines some of the key issues that need to be considered when characterising and measuring mine soils.

# Promise of developing coal combustion products (CCPs) in soil amendments

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Coal combustion products (CCPs) supply to resource the Australian agricultural and horticultural industries is an emerging market opportunity for all participatory stakeholders, with managing for sustainability an essential outcome. Given the emerging opportunities, a targeted analytical testing and review is being developed with the aim to provide regulators, users and the community with confidence about the use of CCPs on soils and the local environment impacts. Historically generators (coal-fired power stations) have focused their assessment of CCPs characteristics upon heavy metal concentrations or compositional oxides and on cementitious properties that achieve industry standards such as AS3582.1. Our approach is a cross-disciplinary industry investment into a knowledge base for coal-ash materials for land-based applications, and particularly agriculture. Low interest prevails in applying the combusted coal products (CCPs) in Australian agriculture, as in many other places, mainly due to variability in mineral and elemental composition from different and localised coal sources, which makes a unified approach to understanding these products within the realm of soil science an important strategy. We present the development of analytical testing drawn from Australian soil-chemical testing methods, representing a significant expansion to test suites currently used for ash-development projects. We suggest that this uniform approach to analytical testing and agricultural management undertaken within agroecological principles is the appropriate strategy. This paper presents results for coal ash materials analyses coal ash products from the NSW Western coal field as tested for agronomic purposes and a protocol to ensure that data interpretation will enable an informed characterisation of land applications.

# Effect of water potential on germination of seeds in ecosystem restoration, Brigalow Belt, Queensland, Australia

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### Abstract

We investigated the effect of water potential on seed germination of native species occurring in the Brigalow Belt – a semi-arid bioregion of Queensland and New South Wales, Australia. Seeds were germinated in PEG 6000 solution at nine osmotic potentials including equivalents of soil water conditions at saturation, field capacity, and permanent wilting point. Two species co-dominating the plant communities in the Brigalow Belt were used – *Eucalyptus populnea* (Poplar box or Bimble box) and *Casuarina cristata* (belah). The germination rate of *C. cristata* was generally lower for the entire range of water potentials. The water potential that maximised germination of *C. cristata* corresponded to soil water potential at field capacity (-30 to -10kPa). On the other hand, germination of *E. populnea* decreased continuously with decreasing water potential and germination was even observed for water potential as low as -1000kPa. These results are expected to be useful for physiological parameterisation of ecohydrological models. Strategies using *E. populnea* on post-mining areas rather than *C. cristata* might be more robust in the face of erratic rainfall events occurring in the Brigalow Belt.

**Key Words:** soil water potential, Brigalow Belt, ecosystem rehabilitation, seed germination

### Introduction

Environmental factors such as temperature, oxygen, light, or water availability control the germination behaviour of seeds. Amongst those factors, water availability is the most critical one in water-limited (or water-controlled) ecosystems such as those in the Brigalow Belt – a semi-arid bioregion of Queensland and New South Wales, Australia. In the Brigalow Belt, water availability is controlled by rainfall events highly variable in spatio-temporal occurrence (Lloyd 1984). Plant communities are dominated by brigalow trees (*Acacia harpophylla*) (Johnson 1980; Johnson 2004) and co-dominated by a range of overstorey (e.g., *E. populnea* (Poplar box or Bimble box), *C. cristata* (belah)) and understorey species (e.g., *Geijera* sp., *Eremophila* sp., *Myoporum* sp). Soils are rich in clay, and have high fertility and good water-holding capacities (Gunn 1984). Much of the overall native brigalow woodland had been cleared for agricultural purposes (e.g., cropping and grazing) with little of the pre-disturbance vegetation currently remaining. Likewise, significant areas are concurrently affected by coal mine developments (e.g. in the Bowen Basin). A primary goal of rehabilitation is to achieve a stable and sustainable ecosystem, which involves the re-establishment of brigalow plant communities using natural or direct seeding. A recent study on ecohydrological interactions of brigalow ecosystems revealed that the co-dominant species might control ecohydrological functions such as evapotranspiration (Arnold *et al.* 2012b).

The objective of this study was to investigate the effect of water potential on germination of selected seeds co-dominating the brigalow plant community. We conducted an experiment that controlled for water potential, while keeping other environmental factors constant.

### Material and Methods

Seeds of two native species that are co-dominant in the Brigalow Belt were used in this study – *E. populnea* and *C. cristata*. A literature review on pre-treatment methods on seed coats of both species (e.g., acid pre-treatment, heating, burning, mechanical scarification, (Schmidt 2000)) revealed that no intervention was required. All materials were sterilized using an autoclave (e.g., Petri dishes) or with ethanol (e.g., tweezers). A laminar flow cabinet was used to avoid fungal contamination in the course of experiment preparation and monitoring. Seeds were germinated in solutions of polyethylene glycol (PEG 6000) at nine osmotic potentials, 0, -10, -30, -100, -250, -500, -750, -1000, and -1500kPa, which were assumed to represent the range of soil water potentials experienced in the field. The empirical equation derived by Michel and Kaufmann (1973) and revised by Wood (1993) was used to set up the required water potential ($\psi$). Soil can be considered saturated ($\psi_s$ = 0 kPa), at field capacity ($\psi_{fc}$ = -30 to -10kPa), or at the permanent wilting point ($\psi_{pwp}$ = -1500kPa).

For each species, three replicates of 50 seeds were used in each treatment. In order to avoid water losses, edges of Petri dishes were tightly sealed with an impermeable colourless parafilm. Seeds were allowed to germinate at about 25°C in a 12h day-night cycle. A seed was considered to have germinated when the radicle had emerged and was then removed from the treatment. Seeds were removed and considered not viable if severely covered by fungi (Baskin and Baskin 2001). Experiments ended when no more germination occurred for at least five successive days.

**Results and Discussion**

The germination behaviour of both species in relation to water potential was different both qualitatively (shape of germination curves in Fig. 1) and quantitatively. In general, the germination rate of *C. cristata* was lower for the entire range of water potentials compared to *E. populnea* (Fig. 1 and Fig. 2 - note the different scales for cumulative germination). For example, for water potential corresponding to $\psi_s$ the germination rate was above 50% for *E. populnea*, but less than 15% for *C. cristata*. Viability tests at the end of the experiments revealed that about 90% of non-germinated seeds of *C. cristata* were still viable but had not overcome physical dormancy. Although previous studies did not indicate any requirement of pre-treating seed coats of *C. cristata*, the high percentage of non-germinated viable seeds indicates that future experiments should include mechanical scarification such as seed coat nicking or chipping to overcome physical dormancy through increasing the permeability of the seed coat.

While the germination of both species converged to zero with decreasing water potential, only *E. populnea* was able to germinateat a water potential as low as -1000kPa, corresponding to very dry soil water conditions. Interestingly, the decline in germination was continuous for *E. populnea*, whereas for *C. cristata* the germination curve in Fig. 1 (black dashed line) indicates an optimal water potential corresponding to $\psi_{fc}$ (cross and diamond markers in Fig. 2b) where germination is maximised. However, this result should be interpreted with caution because the relative variability amongst replicates was much higher for *C. cristata* (grey dashed lines in Fig. 1) compared to those of *E. populnea* (grey solid lines in Fig. 1). Remarkably, for *E. populnea* this uncertainty was more pronounced under high water potential (-30 to 0kPa), whereas for *C. cristata* variability amongst replicates was higher under moderate water potential (-500 to -100kPa, grey dashed lines in Fig. 1).

**Figure 1. Mean germination of *E. populnea* (black solid) and *C. cristata* (black dashed). Each replicate is illustrated by a grey line (solid for *E. populnea* and dashed for *C. cristata*).**

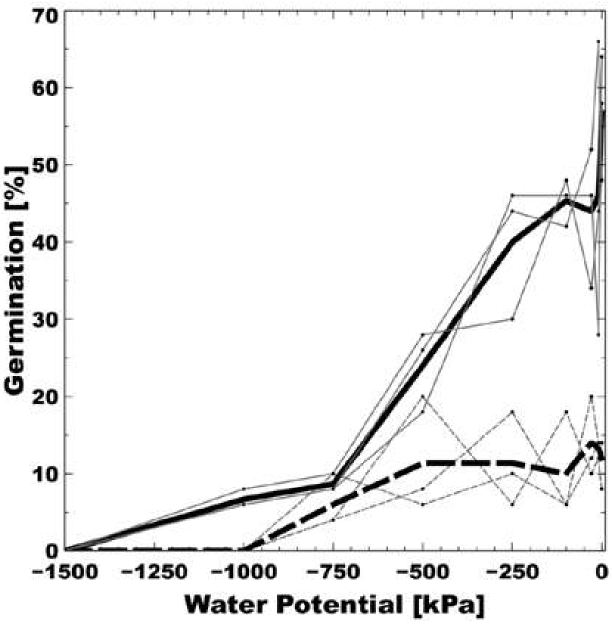
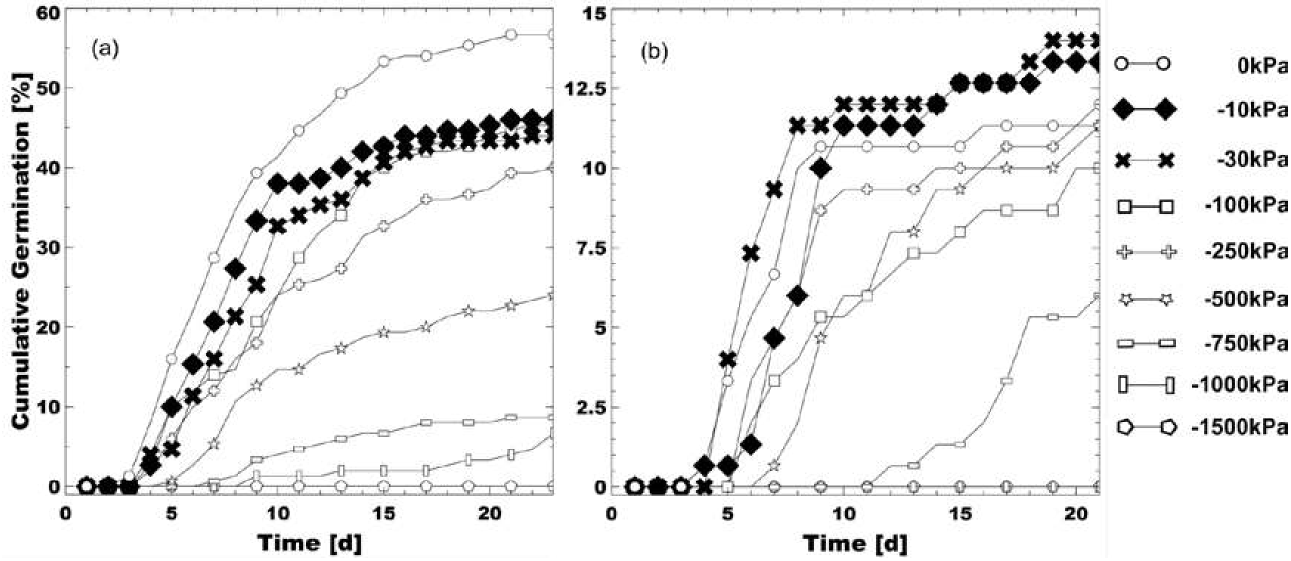


**Figure 2. Cumulative germination of (a) *E. populnea* and (b) *C. cristata* for values of soil water potential ranging from 0 kPa (saturated soil) to -1500 kPa (permanent wilting point). Note the different scales for cumulative germination. No germination was observed for *C. cristata* at a soil water potential of -1000kPa.**




**Implications for Ecohydrological Modelling as a Restoration Tool**

The results will be useful for physiological parameterisation of ecohydrological models. A recent study of ecohydrological interactions of plant communities in the Brigalow Belt (Arnold *et al.* 2012a) indicates that the co-dominant species such as *C. cristata* rather than the dominant brigalow (*A. harpophylla*) play a critical role in ecosystem functions (e.g. regulation of evapotranspiration). However, ecohydrological interactions such as soil-water-controlled germination and plant evaporation protection from bare soil have not yet been investigated. The functional relationship between the occurrence of rainfall events, soil water dynamics and short-term establishment of vegetation, for example, can be investigated using the results of this study to derive species specific empirical equations of germination success controlled by soil water potential. This is an important step within the cycle of scientific discovery (Savenije 2009) where empirical data facilitate the re-calibration or re-structuring of (ecohydrological) models (Arnold *et al.* 2012a), which will eventually improve the predictive power of those models, for example for application as restoration tool. In this context, other co-dominant overstorey and understorey species (e.g., *Eucalyptus cambageana*, *Geijer*a sp., *Eremophila* sp. or *Myoporum* sp. (Gunn 1984; Johnson 2004)) but also the dominant overstorey species *A. harpophylla* should be considered for experiments on the effect of water potential on seed germination.

With regard to re-establishment strategies in the face of erratic rainfall events, the timing of sowing is critical, particularly for *C. cristata*, because optimal soil water conditions for germination correspond to soil water potentials at field capacity ($\psi_{fc}$) (Fig. 2). That is to say, germination rate drops for both very wet conditions (saturated soils) and moderate to dry soil water conditions ($\psi$ = -1000 to -750kPa). On the other hand, no optimal soil water conditions could be found for *E. populnea*, but germination could still be possible under very low soil water conditions ($\psi$ < -1000kPa). This indicates that seeds of *E. populnea* are much more water stress resistant, and that strategies considering the re-establishment of *E. populnea* on post-mining areas rather than *C. cristata* could be more robust in the face of prolonged drought periods driven by erratic rainfall events. However, given the elevated variability amongst the three replicates for the germination trials on seeds of *C. cristata* (grey dashed lines in Fig. 1) uncertainty is involved in concluding that optimal soil water conditions exist that maximise germination. Future experiments should aim to minimise this uncertainty by increasing the number of replicates for the critical levels of water potential (compare variability of grey lines in Fig.1), for example at high and moderate water potential for *E. populnea* and *C. cristata*, respectively.

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# Assessment of soil characteristics across Singapore central streetscape

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**Abstract**

Recently studies on urban soil is getting considerable attention world-wide as it is one of the most challenging frontiers of soil science. Comprehensive soil data relating to the changes in its properties with time are scant in Singapore. There is also an urgent need for a systematic approach to soil sampling and methodological protocols for soil quality assessment for urban soils, so that an efficient soil management programme could be implemented for plant growth along urban streetscapes of Singapore. To address these issues, firstly Singapore city-state was divided into five administrative regions (North, Northeast, East, Central and West) and four major roads for each region were selected randomly for sampling. Each road was divided into three locations, each location was again sub-divided into three sub-locations and the soil samples were collected from three different depths (0-0.30, 0.30-0.50, 0.50-1.0 m) in a quadrate from each sampling point. Results showed that the organic carbon content was significantly higher in the surface soil layers of central streetscape and the soil bulk density ranged between 1.06-1.33 g/cm$^3$. The sampling strategy followed here is helpful to understand the differences in other soil properties and could be adopted for soil surveys for urban streetscape soils. Management plays a significant role in determining urban soil characteristics. The nutrient (P and K) content was not significantly affected by soil depth, but was significantly different across sampling locations. This is the first ever systematic baseline comprehensive soil data collected for Singapore streetscape.

**Key Words**

Urban soil, soil sampling, nutrient, soil quality

## Introduction

Soils associated with urban land uses act as a component for various ecological functions. The characteristics of urban soils are mainly determined by the degree of disturbances they have undergone. These activities cause compaction of the surface soil layers as well as in the lower soil profile (Alberty *et al*. 1984). As a result of that, urban soils show great variability in profile distribution across the landscape, due to the cut and fill, backfilling and resurfacing that occur during the process of land shaping (Blume 1986). Assessment of soil quality may serve as a useful indicator of the long-term accrued effects of urban environments (Pouyat and Effland 1999). Few researches have assessed the unique physical, chemical and biological properties of urban soils; specifically soil bulk density (Jim 1998), soil microbial biomass (Carreiro *et al*. 1999), and soil organic matter (Pouyat *et al*. 2002) have been studied and found to be affected by urban conditions.

Singapore is a city-state tropical country at the southern tip of Peninsular Malaysia, located between latitude 1°09'N and longitude 103°38'E with a hot and humid climate. The geology of Singapore consists of four main formations viz. (a) igneous rocks consisting of 'Bukit Timah' granite and 'Gambak' norite in the north and central-north; (b) sedimentary rocks of the 'Jurong' formation in the west and south-west; (c) quarternary deposits of the old alluvium in the east; and (d) recent alluvial deposits of the 'Kallang' formation, distributed throughout the island (Leong *et al*. 2002). Several projects have developed in Singapore on urban environmental studies, but most of them focusing on the above-ground aspects or are confined addressing specific regions. The importance of ecological functions and services of soil in urban systems, however, is not yet fully recognized, although little work has been done to characterize Singapore residual soils (Rahardjo *et al*. 2004), but even less work has been done on soil microbiology in urban environment (Scharenbroch *et al*. 2005). Furthermore, comprehensive soil data relating to the changes in its properties with time are scant in Singapore. There is also an urgent need for a systematic approach to soil sampling and methodological protocols for soil quality assessment. Therefore it is essential to have quantitative information on soil properties (physical, chemical and microbiological) which would help in making informed decisions in a range of disciplines from engineering to horticulture, and the outcomes of this study will also enfranchise professionals and their ability to consider this important segment of the urban landscape.

## Materials and methods

The fragmentation of exposed surfaces and the variability of soil properties make it difficult to follow any established sampling strategy for urban soils. Therefore a sampling protocol was proposed to minimise the spatial variability. Soil samples were collected across Singapore streetscape following that sampling strategy.

Firstly, Singapore city-state was divided into five administrative regions according to Urban Redevelopment Authority (URA) master plan 2008 viz. North, Northeast, East, Central and West (Fig. 1). For each region, four major roads were selected randomly for sampling. Each road was divided into three locations (L1, L2 and L3), each location was again sub-divided into three sub-locations (R1, R2 and R3) and the soil samples were collected from both sides of the road (A and B) in a quadrate from each point. These sampling points were defined at ~200 m intervals along the entire accessible length of the street. At each point, composite samples were collected (5 specimens from 2 $m^2$) for each depth. GPS reference points were also recorded for each sampling point for future sampling (Fig. 2).

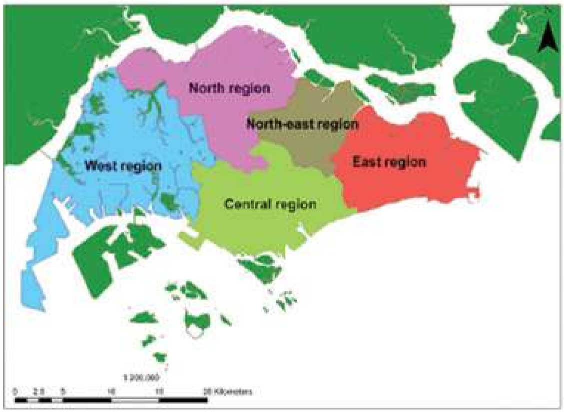


**Figure 1. Regions of Singapore city-state according to URA master plan 2008.**

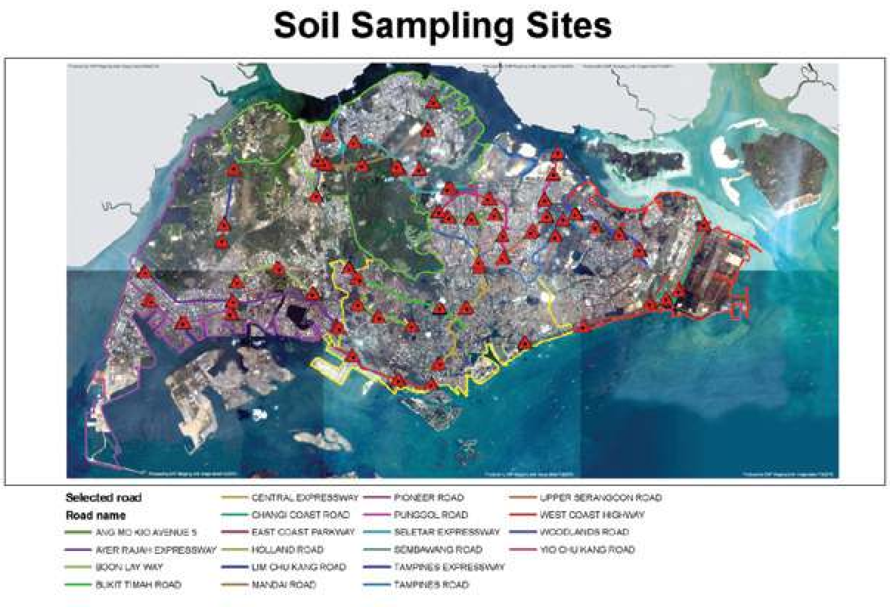


**Figure 2. Soil sampling sites across Singapore streetscape.**

Soil samples were collected from 0-1.0 m depth, with subsequent division of the samples into three viz. 0-0.30 m, 0.30-0.50 m and 0.50-1.0 m. Samples were collected using a 50 mm diameter corer and *in situ* bulk density was determined. Foreign objects e.g. large stones, pieces of glass or pottery etc. were removed and sub-samples (after the microbial analyses) were air-dried and then samples were sieved to < 2 mm for chemical analysis and >2 mm for soil physical analysis. Organic carbon content of the soil was estimated by the wet digestion method (Walkley and Black 1934), $NaHCO_3$ extractable P was determined by a colorimetric method (Olsen *et al*. 1954), and extractable K content were measured following Jackson (1973). In central Singapore, the sampling points were on the streetscapes of Bukit Timah Road, Holland Road, West Coast Highway and Central Expressway.

Significant differences in soil properties between the sampling points for each road were calculated based on the standard errors (SE).

**Results and Discussions**

Result showed that the bulk density of the central streetscape soils ranged between 1.06-1.33 g/cm$^3$ and the lower value was observed at the deeper layer (0.5-1.0 m). The low bulk density in these soils might be because of high clay content of the streetscape soils in the deeper layers. Management has a strong effect on urban soil properties. Because of continuous application of compost or mulch during and/or after plantation resulted in a high content of organic carbon in these soils, especially in the top two soil layers (0-0.30 m and 0.30-0.50 m) (Fig. 3). Similarly, Doichinova *et al*. (2006) observed a higher content of organic carbon in surface horizon in Sofia, Bulgaria. There were no significant differences on the bulk density and organic carbon content across both sides of the roads, however significant differences were observed between the sampling points along the roads. In urban conditions, trees along the streets are planted by using different soil-based media which significantly affect soil physical properties, especially soil structure and texture. These might result in the differences in organic carbon content and bulk density along the roads.

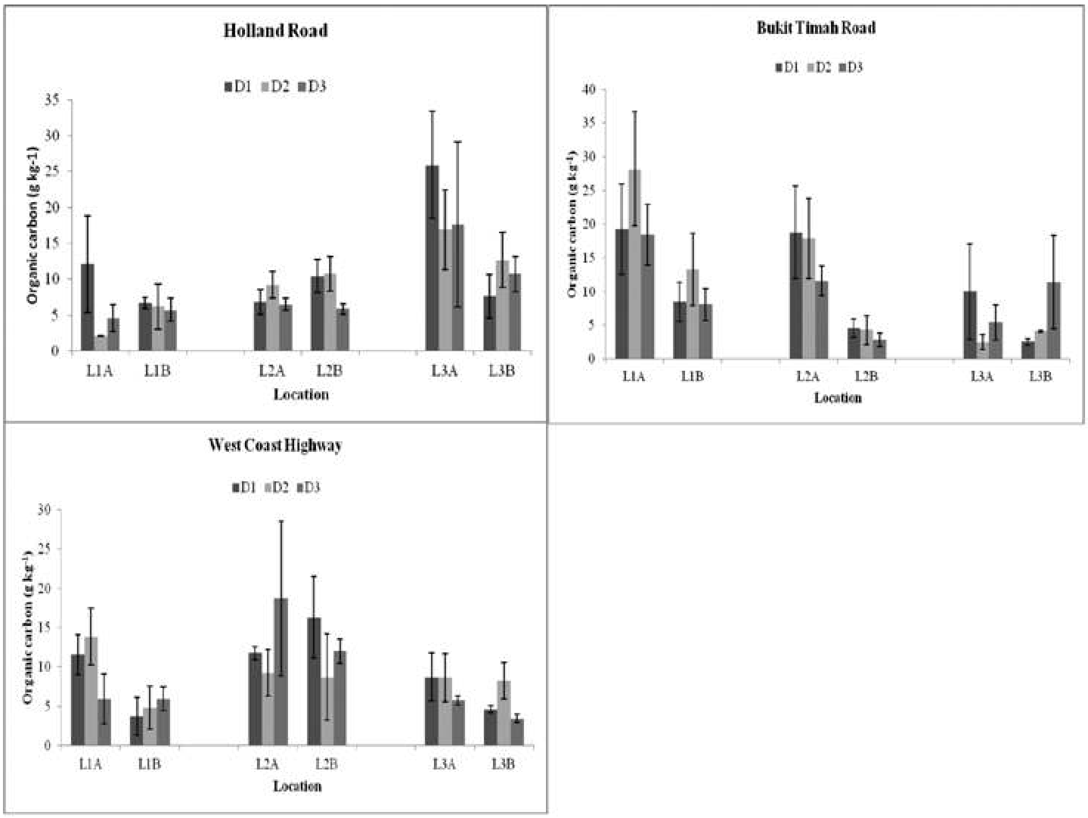


**Figure 3. Depth-wise distribution of organic C (g kg$^{-1}$) across three major roads in central Singapore. D1 = 0-0.30 m, D2 = 0.30-0.50 m, D3 = 0.50-1.0 m; L1 = location 1, L2 = location 2, L3 = location 3; A and B refers to each side of the road.**

For the soil nutrient status, extractable P and K content were not significantly affected by soil depth. Average P content of the surface soil was 1.77 mg/kg, whereas the contents were 1.91 mg/kg and 1.69 mg/kg at 0.30-0.50 m and 0.50-1.0 m respectively. Extractable K content of the central Singapore streetscape soils varied from 27.5-59.5 mg/kg. Bukit Timah road showed significantly higher quantities of K compared with other major roads of the region. The variations in nutrient content are strongly associated with fertilizer applications and intensity of use.

There is a need to understand the modifications of soil characteristics and their spatial distribution in urban ecosystems, so that traditional soil survey approach followed in agricultural environment can be applied to the urban environment (De Kimpe and Morel 2000). The sampling strategy followed here is helpful to minimise the heterogeneity of urban soil properties and also to understand the spatial distribution of various soil characteristics. Subsequent data will be generated from other regions as shown in Figure 2, which will support to establish the protocol. Urban soils have unique characteristics as modified by significant site disturbance for urban infrastructure and these factors play an important role in determining the spatial distribution of soil characteristics. This is the first comprehensive systematic soil data recorded, which can be used as baseline reference data to characterize streetscape soils of Singapore. The quantitative information on soil properties (physical, chemical and microbiological) help to identify the key soil quality indicators and the problems of current urban soils, so that a systematic programme for soil remediation could be implemented to improve the growth of plants along urban streetscapes of Singapore.

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# A potential remediation strategy for acid sulfate soil affected agricultural land in the Lower Murray Region

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The Lower Murray Reclaimed Irrigation Area (LMRIA) comprises over 5000 ha of agricultural land between Mannum and Wellington in South Australia. The construction of levee banks along the River Murray channel in the 1940's allowed this land to be developed for agriculture using gravity feed flood irrigation. From 2007-2009, drought conditions in the Murray-Darling system led to unprecedented low water levels below Lock 1. As a consequence, much of the LMRIA was not irrigated for substantial periods of time, allowing the water table to fall up to 1.5 m. The heavy clay soils cracked deeply which enabled oxidation of acid sulfate soil (ASS) materials and severe soil acidification ($pH<4$) 1-3 m below ground. When the water levels recovered and irrigation recommenced in late 2010, the stored acidity, and associated soluble metals, entered the shallow groundwater and the numerous drainage channels, which eventually discharge back to the River Murray. Following concerns regarding the potential for acidic drainage water to affect the environmental values of the River Murray, it was considered necessary to examine ways to treat acidity within LMRIA soils. The EPA, in conjunction with local landholders, is trialling a world's first technique which will attempt to raise the pH of the soil and groundwater by injecting a highly alkaline lime ($CaOH_2$) slurry at depth using a modified mole plough. The aim is to raise the pH of the soil and groundwater sufficiently so that sulfate reduction, a natural process of ASS remediation, can occur. In treating the acidity at the source, the concentration of acidity and soluble metals in the ground and drainage water is reduced, lessening the eventual harm to the River Murray and allowing the LMRIA to continue to be a productive agricultural area.

# Soil-for-life: Enhanced understanding and management of soil resources in UK agricultural systems via multi-scale, multi-stakeholder soil information system and environmental informatics approaches

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**Abstract**

The development of *Soil-for-life*, a multi-scale, multi-stakeholder soil information system facilitates the innovative integration of multivariate data pertaining to soil functions across expanding and potentially unprecedented spatial and temporal scales. The system aggregates the disparate data holdings of individual horticultural farms, currently predominantly based in the East Anglian region of the UK. These data are then integrated with existing soils and land information held within several national-scale soil inventories. This dynamic and expanding aggregated data holding represents a collation of information that was hitherto considered inaccessible in such an integrated manner, and allows an insight into the impact our agricultural systems have on the functioning of our soil resources. This paper outlines the development phases of *Soil-for-life* and introduces two studies conducted to demonstrate and test the proposed concepts, examining the concepts of soil health within the contexts of intensive brassica production and potato cropping.

**Keywords**

Sustainable intensification, soil management, soil information systems, environmental informatics, agriculture, horticulture.

## Introduction

Soil is a finite resource requiring proactive management in order to sustain it (Creamer *et al*. 2010). It provides the basic growing medium for crops whilst functioning to deliver a wider range of ecosystem services (Wall *et al*. 2012). The fundamental principles of sustainable agriculture promote the importance of soil health to continue the provision of such services. Integrated approaches are required to support these principles, acknowledging and monitoring the complex spatio-temporal interactions that exist between land use and soil functions (Kibblewhite *et al.* 2008).

This research is part of a collaborative research project between Cranfield University and a leading UK fresh produce supplier (Produce World Ltd.). Central to this project is the development of a spatially-explicit soil information system (‘*Soil-for-life*’) for the entire land-bank (c.17,000 hectares) of a horticultural grower group, enabling the collation and visualisation of a wide range of soil and crop attribute data from a varied range of sources across a multitude of scales. This approach provides a framework to standardise, document and normalise data sources to allow like-by-like comparison and integrated data analysis.

*Soil-for-life* facilitates the innovative integration of multivariate data pertaining to soil functions across growing and potentially unprecedented spatial and temporal scales, providing direct access, impact and value to soil-system scientists and land managers. A series of case studies illustrate how *Soil-for-life* has the capacity to facilitate the monitoring of soil attributes such as organic carbon, drive scenario-based assessments to protect against risks and ultimately enhance soil health and sustainable intensification. In addition *Soil-for-life* provides extensive data sources to form a powerful research tool to conduct data-driven, and ultimately evidence-based research into themes such as soil health indicators and sustainable management of this most fundamental resource. In this paper two of these studies are introduced, exploring the role of soil health within both brassica and potato production systems. These summaries outline the development stages, data sources, data analysis methods and project frameworks for each case study.

## System development

The *Soil-for-life* soil information system is based upon existing relational database management software (PostGres/MS SQL Server) which is coupled with software that allows geospatial data to be viewed and edited (GeoServer) via web-based services (OpenStreetMap/OpenLayers). The system is designed to provide a platform for growers, farmers, land managers and land owners to upload data for the land that they farm or manage, thus producing a dynamic, coherent and growing aggregated data holding pertaining to the modern status and management of soils within the context of economic and environmental sustainability of agricultural production. Uploaded data is georeferenced to individual field parcels, an intentional step to lock the resolution to the primary management unit and maintain relevance to, and buy-in from, users and stakeholders. System users have access to the soil information system through three web-based platforms:

- Data upload tool: Allows users to upload data into the system and assign appropriate levels of metadata.
- Land identifier: Provides users with the data required to select individual fields and access unique identifiers required for accurate georeferencing of data.
- Data viewer: Allows users to access data within the soil information system with the capability of searching, viewing, editing and querying the aggregated data holding on an anonymised basis.

A prototype system, tested with a group of 12 horticultural growers provided an opportunity to gauge insight from users and refine the design of the final system, focusing on the protection and promotion of soil health. A specific objective during this prototyping stage was to collate a list of soil attributes from this group, classifying these attributes by availability, accessibility, relevance and pertinence to specific systems. This research was conducted through a series of informal interviews with individual growers and with the grower group as a whole, at a dedicated grower meeting. In response to this prototyping stage, a scoping-scale study of soil organic matter was conducted with this grower group, resulting in the collection of 129 soil samples from a total of 44 fields. This study presented an opportunity to collect an initial swathe of data, test the agreed standard operating procedures and identify the most appropriate means of communicating results back to growers.

**Data analysis**

The aggregated data holding within *Soil-for-life* holds significant potential in terms of data analysis and interpretation. In its extant state (as of August 2012) the system requires 17.5 gigabytes of data storage and includes holdings from: 14 horticultural growers; a fresh produce packer-supplier; UK Governmental bodies including the Department for Environment, Food and Rural Affairs, the Environment Agency, the Rural Payments Agency and Natural England and the National Soil Resources Institute. Grower data cover a land bank of 650 field parcels (*c.*6,700 hectares; Fig. 1) and include data pertaining to farm management, trafficking, fertiliser regimes, pest control, cropping history and soil fertility.

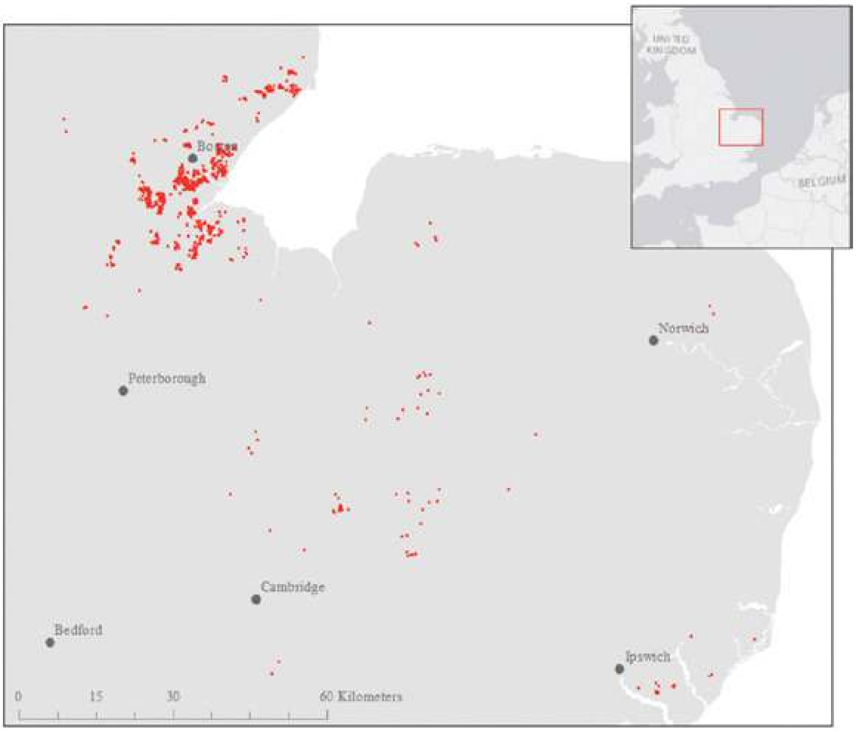


**Figure 1: Spatial extent of soils data held within Soil-for-life (individual red dots represent single field parcels).**

The approaches adopted by this research require discrete bounds around the system specifically being considered. For this reason the project adopts system-specific definitions of soil health, limiting the term to a bounded context such as, for example, '*soil health in relation to the production of a <prescribed> crop*'. A key challenge to this approach is the need to acknowledge and bridge the potentially contrasting objectives faced by food-security oriented perspectives of maximising food production in the first instance and promoting soil health more generally as well. Informatics approaches, specifically multivariate methods and network modelling techniques form the foundation of data analysis within this project.

Central to these approaches is a series of structural equation models which are used to represent individual definitions of soil health, identifying the primary response variable and creating the subsequent supporting network of influencing parameters, latent and observed variables (Fig. 2; Bollen 1989). In this context, *latent* variables are use to describe variables that are not directly observable, and hence are inferred; these approaches acknowledge latent variables and the impact they have upon *observed* variables, which can be at least qualitatively described and preferably quantified. A hypothetical range of such possible interactions is illustrated in Fig. 2, including the influence of latent variables upon observed variables [L(2) and O(3)], reciprocal effects [L(1)] and correlations between variables [O(2) and O(5)].

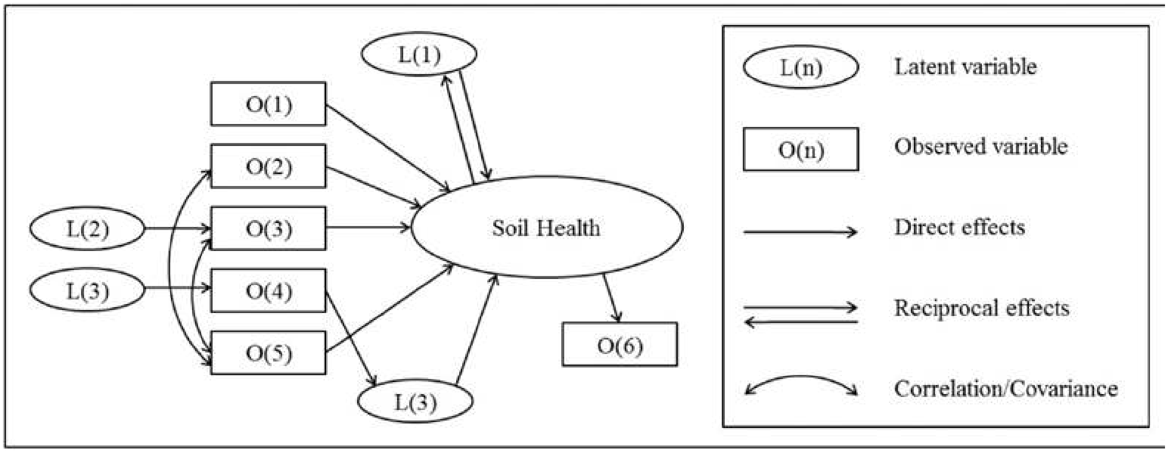


**Figure 2:An example structural equation model for the determination of soil health, illustrating the range of possible interactions between latent (non-observable) and observed variables.**

Once the model is developed, a series of extant datasets and proxies for each component can be incorporated and statistical analysis conducted to identify key relationships and influencing parameters. The initiation of two proof-of-concept studies, focused on utilising the data within the *Soil-for-life* system are due to be completed by Spring 2013. Overviews for each of these studies are presented below:

- *Soil health in brassica production*: This study focuses on intensive brassica production systems in East Anglia and explores the impact of continuous monoculture cropping on soil health. The study utilises a *Soil-for-life* dataset sample consisting of 16 years coverage of cropping history from a total of 96 fields (>1000ha) which was sourced from 10 land owners and growers. The structured equation model for this studied is focused on marketable yield as the primary response variable, a quantifiable value calculated as the volume of crop meeting a specified quality (a concept entrenched within the UK supply chain). Data and factors relating to the cropping environment as a whole have been integrated as variables, including factors such as fertiliser, pest and disease management; cultivation methods; soil type and fertility; crop type, variety and performance; water availability; and spatial variability of crop quality. Time-series analyses, used to filter constants from the structured equation model setup, are being developed for each variable.
- *Soil health in potato production*: This study utilises datasets from the pack-house of a large fresh produce packer-supplier and focuses specifically on potato crop defects and the linkages with soil properties. The study is intentionally narrowed to focus on one farm, investigating levels of potato crop defects including scab, bruising, dry matter content, skin finish, sprouting and black dot/scurf. These defects are used as the response variable in this study and will be integrated with field data including cropping environment variables similar to those listed in the previous example. In a similar approach to the previous example, the output of this study will be a modelled definition of soil health in potato production, with defects and marketable yield acting as the primary response variables.

## Discussion

*Soil-for-life* has been developed to provide a framework to standardise, document and normalise disparate data sources, allowing like-by-like comparison and an integrated approach to data analysis. This data can be used to better understand the complex spatial and temporal interactions between soils and cropping environments (Rutgers *et al*. 2012). The development stage alone of this project has demonstrated that these integrated approaches, underpinned by good agricultural extension mechanisms, can drive positive changes in management across the grower-base, identifying and promoting best practice. For example, a culture of openness and a mindset where (non-commercial) data-sharing is the norm has been encouraged, and manifest, by developing a platform for growers to attain a better understanding of the land they farm, benchmarking their key soil metrics against analogous systems in a structured and accessible manner.

The value of these data transcends a range of scales, providing both information on the sustainable use of land resources for agricultural production and a source of data for scientific investigation. The two case studies outlined in this paper provide insight into the power of such a system in providing data to populate unknowns and quantify latent variables within network models such as Fig. 2. The concurrent development of such models alongside the continuing accrual of data over time, presents a robust foundation for future research.

The challenge of optimising soil health for food security as well as maintaining sustainable and profitable agricultural production, demands a collaborative response from policy makers, scientists and land managers. The approaches adopted within *Soil-for-life*, specifically 'crowd-sourced' data capture, take this collaboration beyond dialogue into a paradigm of data-driven information, advice and agricultural extension.

## Acknowledgements

The authors would like to thank colleagues at the National Soil Resource Institute; project partners Produce World Ltd.; the *Soil-for-life* grower group, specifically Marshalls Farming, G & D Matthews Ltd., Allpress Farms Ltd., Tompsett Growers, TaylorGrown Organics, Elveden Farms Ltd.,3M's Ltd.; and the project sponsors: Knowledge Transfer Partnerships, Technology Strategy Board, National Environmental Research Council, Department for Environment, Food and Rural Affairs, Biotechnology and Biological Sciences Research Council.

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# Monday 3 December

## PEDOLOGY, SOIL STRATIGRAPHY AND QUATERNARY LANDSCAPES

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# Unconsigning the Dermosol to the rubbish bin of Australian soil classification

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The Dermosol order of the Australian Soil Classification (ASC) scheme includes those well-structured mineral soils that lack a texture contrast between the A and B horizons, and that lack the diagnostic features of Podosols, Vertosols, Hydrosols, Ferrosols and Calcarosols. While the etymology of 'Dermosol' suggests that illuviation to form clay skins is an important or typical pedogenic process for this soil type, evidence of this process is not a diagnostic criterion. The areal extent of Dermosols across Australia is estimated to be approximately 1.6% of the country, indicating that it is a relatively minor soil order. However, in our experience of surveying soils in eastern NSW, the Dermosol can be a dominant soil type in many agricultural catchments, belying its apparently small contribution to Australian soil coverage. In such localities, the often repeated cultivation of topsoils and the associated greater risk of erosion significantly increases the possibility that erstwhile texture-contrast soils have 'become Dermosols'. Using a large soil profile dataset from the Hunter Valley of NSW, we show that Red and Brown Dermosols are very common in a variety of landscape positions subjected to agriculture, whereas Chromosols and Kurosols are more common in undisturbed or pastoral locations. We hypothesise that a substantial number of Dermosol profiles in agricultural fields have a shorter *taxonomic distance* to nearby Chromosols and Kurosols than to nearby Dermosols in undisturbed locations. There is a clear pattern of the Dermosol order being a 'catch-all' for cultivated soils with clayey B horizons. Where evidence of this 'dermosolisation' is fairly clear, we contend that there should be flexibility within the ASC scheme to reflect this information. This may be achieved by introducing; (i) an Agric Dermosol suborder, or (ii) a Dermosolic Anthroposol suborder, although for the latter case, the author of the ASC explicitly excluded "…agricultural operations…which may change a soil from say a Chromosol to a Dermosol" from the "profound modification, mixing and truncation" requirement of an Anthroposol. However, there is little doubt that the main soil altering factor is anthropic.

## On the Way to the third Edition of WRB (2014)

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World Reference Base for Soil Resources (WRB) is a worldwide-applicable soil classification system that has been endorsed by the IUSS Council as the IUSS international correlation system. The first edition was published in 1998, the second in 2006 and the third is due to be presented in 2014 at the World Congress in Korea. The second edition (2006) avoids stronger hierarchies and is concentrated on the classification of pedons. For creating map legends an amendment allowing stronger hierarchies was published in 2010. One of the aims of the third edition is to have a common document for both purposes. Field testing of the second edition revealed the need for some (mostly minor) changes in the definitions, which is the second aim. A third aim is writing the definitions in a more didactical way. A fourth aim of the third edition is to ensure a better WRB classification of Australian soils. There was never before a broad testing of WRB in Australia and some soils, widespread in Australia and rare in other parts of the world, may be daifficult to be allocated according to WRB, among these being the sodic texture-contrast soils. Common excursions directly before this conference should help to find possible adjustments of WRB definitions to encompass these soils. Some examples will be given in this presentation.

# Testing the reconstruction of environmental and climatic change using ancient DNA preserved in buried allophanic palaeosols on tephras: Challenges and pitfalls

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Sedimentary ancient DNA (sedaDNA, DNA isolated from palaeosediments) is a powerful temporal tool that can be used to complement existing disciplines such as palaeontology and palynology to reconstruct late Quaternary palaeoecosystems. The majority of sedaDNA studies have focused on ice, cave, lake, and permafrost deposits, with only a handful of studies utilising modern soils or buried palaeosols. Our research is investigating whether ancient DNA (aDNA) isolated from buried palaeosols can be used to conduct palaeoenvironmental reconstructions. Specifically we are targeting palaeosols developed on late Quaternary sequences of weathered tephra (volcanic ash) deposits in the central North Island, New Zealand. The palaeosols in the upper parts of the tephra sequences usually have andic soil properties, being dominated by allophane, a nanocrystalline mineral comprising tiny Al-rich spherules (3–5 nm in diameter) with extremely large surface areas (up to 1500 $m^2g^{-1}$) that form very strong associations with organic matter. Allophane is hypothesised to play an important role also in DNA preservation. In this talk we will outline some of the challenges and pitfalls associated with deriving and analysing aDNA from tephra-derived palaeosols in contrast to sedaDNA studies on other materials. We discuss factors including pedogenesis, DNA leaching, DNA extraction efficiency, and exogenous contamination from modern plant material. We will also present preliminary aDNA data from buried palaeosols on Holocene tephras and how we are overcoming the challenges associated with this new and emerging research field.

# Characteristics of a paleosol and its value as an archive for the landscape history, Rocky Mountains Front Range, Colorado, USA

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Activity and stability phases as well as geomorphic processes within the Critical Zone are well known. Erosion and deposition of sediments represent activity; soils represent geomorphic stability phases. We present data from a 4 m deep sediment section including a paleosol that was dated by luminescence techniques. Upslope erosion and resulting sedimentation started in the late Pleistocene around 18 ka until 12 ka. Environmental conditions at the study site then changed, which led to the formation of a well-developed soil. Radiocarbon dating of the organic matter yielded ages between 8552 – 8995 cal. BP. From roughly 6.2 – 5.4 ka another activity phase accompanied by according sediment deposition buried the soil and a new soil, a Cambisol, was formed at the surface. The buried soil is a strongly developed Luvisol. Black colors in the upper part of the buried soil are not the result of pedogenic accumulation of normal organic matter within an A-horizon. Nuclear magnetic resonance spectroscopy (NMR) clearly documents the high amount of aromatic components (charcoal), which is responsible for the dark color. It is corroborated by charcoal pieces seen in thin sections. This indicates severe burning events at the site and the smaller charcoal dust (black carbon) was transported in deeper parts of the profile during the process of clay translocation.

The presented example documents the value of combining soil and sediment analyses to fully understand the interplay of surface processes and soil genesis within a sediment section in order to reconstruct the landscape history.

# Check your aquic spodosols for nasty sulfides and sulfur

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We have found sulfur, pyrite and other sulfides in wet subsoil horizons of aquic spodosols in inland swale locations in ancient siliceous dunes in south-western Australia.  These are therefore potential acid sulfate soils (PAS) but have not formed in coastal or estuarine deposits where most PAS occur.  Because the soils are very sandy, they are poorly buffered and very low pH values rapidly develop as soils are aerated Groundwaters are also affected  and this condition is not readily ameliorated.

We have developed techniques to recognise and quantify the acid-forming minerals in this type of aquic spodosol and would like to invite all pedologically minded soil scientists to consider if such soils exist in their region.  The sandy wet soils of Tasmania and New Zealand would seem to be candidates for such examination.  Indeed the curious mineralogy, chemistry and morphology ofsome very sandy spodosols throughout the world, including “giant podzols” may sometimes reflect a prior PAS condition.

# Monday 3 December

## PEDOLOGY, SOIL STRATIGRAPHY AND QUATERNARY LANDSCAPES

*Presented Posters*

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# Automated WRB classification – software tool and initial experience with its application

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Soils are described and classified using national guidelines and taxonomies in many countries of the world. For international exchange, the use of an international soil classification system is necessary. The World Reference Base for Soil Resources (WRB) has been adopted by the International Union of Soil Sciences and by the European Union as a reference system to enable communication on soils. Direct translations from the German soil classification system to WRB failed for about half of the German soil types. Due to the high number of already described soil profiles in various German soil information systems, a software application was designed to automate the procedure of assigning profiles to the WRB reference soil groups (RSG) and adding the appropriate qualifiers.

We reformulated the diagnostic criteria of WRB 2006 (update 2007) in a way that they relate directly to data fields of the German Soil Mapping Guideline. For each diagnostic horizon, material, or property, qualifier and RSG, a graphical algorithm shows criteria for data fields. We included proxies for parameters that are assumed to miss in many datasets.

The algorithms have been coded as a MS Access based application. Experience with the application on German and Swiss soil data regarding performance and quality of results is presented.

## Weathering of basic and ultrabasic substrates in soils of cold climates (Russia)

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The study of primary and secondary minerals in the recent soils from highly weatherable basic and ultrabasic rocks has been developing the knowledge of soil formation and allows to prognose soil behavior induced by global change. Mineral association of rocks and soils was analyzed in the thin sections, X-ray diffraction and IR-spectroscopy analysis. Outcrops of serpentinous dunite characterized by initial reticulate structure with fragments of olivine grains in loops and phyllosilicates resulting from metamorphism, are located in the Polar Urals, the Rai-Iz massif. Haplic Cryosols (Reductaquic) (WRB, 2006) are situated in the mountainous tundra. The inherited minerals in soil are represented by olivine, serpentine, talc, and chlorite. Two smectites (saponite and nontronite) and vermiculite are absent in rock and present in soil due to rock weathering in soil environment. In contrast to serpentinous dunite the flood basalt complex (traprocks) covered a large area in the Central Siberian Plateau playing the role of parent materials for the soils. The Epileptic Entic Podzols on dolerites with poikilitic structure, under larch-birch forest were studied in the central part of the basaltic province. Plagioclases, pyroxenes, iron (hydr) oxides, and smectites are identified in the rock and the soil skeleton. The share of inherited smectite(s) drastically decreases in the upper horizons that can be explained by the dissolution of smectites under acidic conditions in the upper soil horizons. Accumulation of smectite in the bottom horizons seems to be the result of rock disintegration and formation of the fresh mineral surfaces which are more sensitive to weathering.

# Three chronosequences in recessional glacial deposits in the Central Transantarctic Mountains, Antarctica

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Soil chronosequences on moraine deposits emplaced during glacial retreat in the Central Transantarctic Mountains, Antarctica, are described. Three 'windows' were studied in detail; the Ong Valley, the western margin of the Dominion Range, and a broad sweep of moraine beside Mount Achernar. The sites are all located between 83° and 85° South, at altitudes of 1600-2200 m.

Descriptions of soils were used to construct soil-landscape maps for the three study windows. Chemical and physical characterisation of each soil horizon, and characterisation of microbial communities in surface horizons, is underway.

The Dominion Range and Mount Achernar sites exhibited similar soil patterns, both being areas with extensive suites of lateral moraine. Both chronosequences have a zone of recent undeveloped Glacic Haploturbels (with massive/glacial ice within 10 cm of the surface) closest to the glacier(s). Hummocky thermokarst topography was dominant adjacent to the glacier with limited or no patterned ground development. Soil age, soil and patterned ground development, and depth to massive glacial ice increased with distance from the glacier margin. Older soils comprised soil associations of Glacic Haplorthels and Haploturbels. Anhyorthels/Anhyturbels (no ice-cement within 70 cm) were less common and the Typic subgroup (no massive ice observed in profile) occurred infrequently. Soil age, by its influence on the extent of glacial ice sublimation, is the major determinant of soil taxonomy.

Ong Valley exhibited a similar chronosequence, although differing in that soils have developed in a series of terminal moraines within a relatively narrow confined valley.

# Australia versus the world … issues in soil classification

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Australia is unique in its particular suite of soils – both in terms of diversity and extent. The two currently used international systems of soil classification – World Reference Base and Soil Taxonomy – are not commonly used in Australia. They are perceived to be 'unnecessarily complex' and lacking in relevance – mainly because of difficulties in correlation with the Australian Soil Classification (ASC). This paper addresses the particular features of Australian soils that make them challenging for international classification systems and offers suggestions on Australian contributions to the development of a Universal Soil Classification system. Soil issues addressed will include: texture-contrast soils, pedogenesis and the argic horizon, sodic soils, wet soils, base status and analytical data requirements.

# Wetland acid sulfate soil distribution between Blanchetown (Lock1) and Wellington, Murray River – is there a pattern to investigate?

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## Abstract

Acid sulfate soil materials can have serious environmental consequences. During the drought between 2006 and early 2010 the wetlands below Lock 1 of the Murray River, between Blanchetown and Wellington, generally all dried exposing the normally subaqueous wetland soils. Soil surveys and laboratory analysis of soil samples were used to identify the acidification hazard for each of the wetlands. To date, the data has been interpreted on a wetland by wetland basis to provide an understanding of acid sulfate soil properties and hazards present within a wetland and to aid in risk assessment. This paper combines information collected at 59 wetlands with earlier work from 14 wetlands to evaluate the acid sulfate soil occurrence and distribution in a regional context, to identify if there is an observable pattern. Preliminary findings describing the regional variability presented in this paper indicate that the occurrence of acid sulfate soil materials and the acidification hazard was greatest in the wetlands downstream from Mannum town, and these materials were associated with cracking clay soils. Based on initial findings, further investigation of the data is warranted to identify if the occurrence of sulfuric and sulfidic material can be predicted by developing linkages between the acid sulfate soil properties with the surrounding landscape that may influence the soil forming processes. An improved understanding will assist with future predictions about the occurrence of acid sulfate soil materials.

## Introduction

The 77 wetlands along approximately 250 kilometre length of the lower River Murray between Blanchetown (Lock 1) and Wellington, South Australia (Figure 1) were severely impacted by unprecedented drought that occurred for approximately a decade up to early 2010. This drought led to significantly lowered river weir pool levels from about +0.75m AHD to -0.5m AHD causing disconnections of water between flood plain wetlands and the river channel. Nearly all of the 77 wetlands along this length of the river were dry, exposing acid sulfate soil materials that would normally be covered with water. Work by CSIRO Land and Water and others in subaqueous soil (lakes and rivers) and wetland environments in this region identified acid sulfate soils with various occurrences of sulfidic, sulfuric and monosulfidic black ooze materials (see review paper and key references in Fitzpatrick *et al*. 2009). Occurrences of these acid sulfate soil materials can have serious environmental consequences relating to soil and water acidification, de-oxygenation of water, emission of foul smelling gases ($H_2S$, organo-S compounds) and the release of heavy metals and metalloids (Shand *et al.* 2010).

There was limited information on the distribution, characteristics and processes of acid sulfate soils throughout this region apart from the work of Fitzpatrick *et al*. (2008a, 2008b, 2010) on 14 wetlands. The wetlands for these initial investigations were selected because they were identified as likely to pose a high acid sulfate soil acidification risk and therefore information discussed was from wetlands that best expressed the acid sulfate soil characteristics and processes. Given the concern about acid sulfate soils identified by these preliminary studies, combined with extensive drying of wetlands below Lock 1, a survey of the remaining 59 wetlands was commissioned by the Murray-Darling Basin Authority to provide a comprehensive and standardised baseline dataset on the wetland acid sulfate soil properties (Grealish *et al.* 2011).

These data have previously been interpreted on a wetland by wetland basis to provide an understanding of acid sulfate soil hazards within a wetland and the risk to the adjacent River Murray. This paper combines information from this study with the earlier work to evaluate the acid sulfate soil occurrence in a regional context for the wetlands below Lock 1, between Blanchetown to Wellington. Preliminary findings describing the regional variability in acidification hazard and soil type are presented in this paper, with the purpose of identifying if there is a pattern that warrants further investigation.

## Method

Between August and November 2008, 189 soil profiles in 59 wetlands were studied, from which 687 soil layers were described and 667 soil samples were collected. The wetlands ranged in size from 0.3 ha to 348 ha, with a median of 34.7 ha. The sample site location and number of sample sites placed within a wetland were determined by the type and size of the wetland. A number of factors were taken into consideration, and in general, 2 to 4 sites were located form a topographic traverse within the wetland. Sampling locations were typically selected at the centre (low), intermediate (mid) and edge (high) position in each wetland. For wetlands that covered a larger surface area, extra transects were added to provide a better spatial distribution of sites.

Three or four layers were typically sampled per soil profile and the layers corresponded with soil horizons if they were observed, otherwise they were sampled by depth, generally there was a soil sample collected from the surface (about 0 to 5 centimetres), subsurface (5 to 20 centimetres), subsoil (about 20 to 50 centimetres), and deep subsoil (50 to 100 centimetres). Laboratory analyses included $pH_{water}$, $pH_{peroxide}$, $pH_{incubation}$, and acid base accounting parameters ($S_{CR}$ (sulfide % S), $pH_{KCl}$, titratable actual acidity, acid neutralising capacity) and water-extractable $SO_4$ (1:5 soil:water suspension). The approach and methodology generally follows the protocols described in MDBA (2010).

The data were used to classify the soil samples into their acid sulfate soil material class – sulfuric, hypersulfidic, hyposulfidic, and other acidic or other soil material as defined in MDBA (2010). The sampled soil profiles were then allocated an acid sulfate soil type according to the Acid Sulfate Soil Identification Key (MDBA 2010).

The wetland acidification hazard rating was determined by assessing within the wetland the acid sulfate soil materials identified and the acid sulfate soil type with the following criteria: 1) the position of the sample in the soil profile, (i.e. if it was a surface sample it is more likely to be at the soil water interface and, therefore, to have an impact on surface water in the wetland than a sample deeper in the profile) and 2) the extent and distribution of the sample, (i.e. based on information available, such as whether the sample was representative of a widespread area of the wetland and therefore more likely to have an impact on the wetland water than an isolated local occurrence). The wetland acidification hazard was then rated as low, low to medium, medium, medium to high or high. For acidification hazard, a high rating generally indicates that sulfuric or hypersulfidic acid sulfate soil material was found. A medium rating generally indicates that hypersulfidic or hyposulfidic acid sulfate soil material was found. A low rating generally indicates that no acid sulfate soil material or occasionally other acidic soil material was identified.

A total of 59 out of the 77 wetlands in the region were assessed from the field data collected as part of this study. In addition, assessments of the data provided in previous CSIRO reports were evaluated for a further 14 wetlands, and 4 wetlands were not assessed because they were not surveyed due to their location.

## Results

The results identified that the wetlands were distributed across all ratings as shown in Figure 1, with 16 wetlands rated as high, 12 as medium to high, 20 as medium, 12 as low to medium, and 12 as low. One wetland was rated as low to high because of the large variation across the wetland. Therefore a total of 73 wetlands have a rating assigned, with 4 wetlands not assessed because data were not available. It should be noted that this assessment was based on the field and analytical data that was obtained during the August to November 2008 field survey, and takes into account the status of the wetland of the acid sulfate soil materials at that point in time. Acid sulfate soil materials change with time from sulfidic to sulfuric as the soil dries and when inundated from sulfuric to sulfidic, and these changes can occur relatively rapidly (Fitzpatrick et al. 2009).

Cracking clay soils were the dominant soil in 26 of the 28 wetlands rated as high or medium to high acidification hazard. Sulfuric cracking clay soils were present in 14 of these wetlands and hypersulfidic cracking clay soils dominant in a further 12 of the wetlands; examples of the wetland surface and soil profile are shown on Figure 2. The remaining 2 wetlands were dominated by organic soils. Cracking clay soils were less dominant in wetlands rated as medium and low hazard, but the soils in these wetlands were generally clayey and occasionally on the margins sandy soils were observed.

The regional distribution shown in Figure 3 indicates that the occurrence of acid sulfate soil materials was greater downstream (south) from Mannum town (located at about wetland number 29 in the figure), where there is a higher proportion of high and medium to high acidification hazard wetlands.

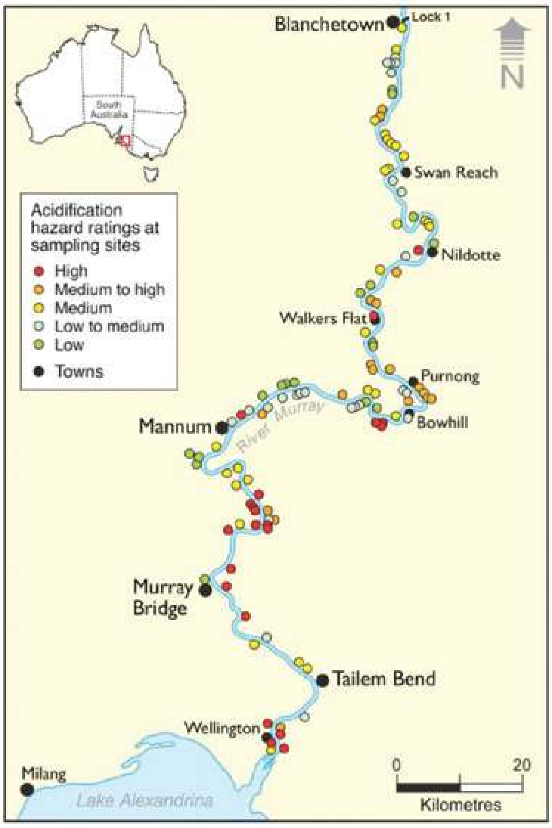


**Figure 1: Wetland acidification hazard assessment rating for wetlands between Blanchetown to Wellington.**


**Figure 2: Upper photograph shows the wetland surface of a hypersulfidic cracking clay soil. The lower photographs show the wetland surface and soil profile of a sulfuric cracking clay soil, where the surface columnar aggregates are breaking down and filling the cracks.**

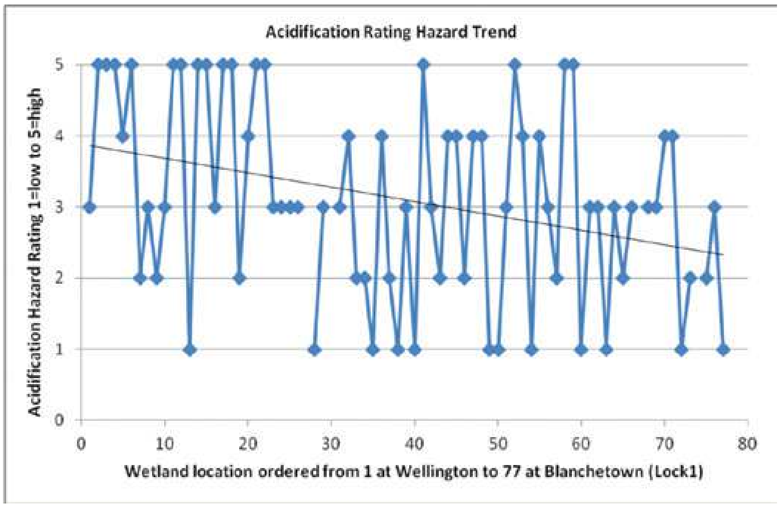


**Figure 3. Wetland acidification hazard ratings from Wellington to Blanchetown.**

**Discussion**

The data analysis using the acidification hazard to summarise the acid sulfate soil property information indicates that there is a general regional trend in the wetlands along the stretch of the River Murray below Lock 1 from Blanchetown to Wellington. This was further supported by the observation that the high acidification hazard wetlands where sulfuric and hypersulfidic soil materials were identified were often associated with cracking clay soils, whereas the medium acidification wetlands where hypersulfidic soil materials were identified tend to be associated with clayey soils.

These findings suggest that there is regional variability for the acid sulfate soil materials. Further investigation of the acid base accounting data, geomorphic, hydrological and biogeochemical data is warranted and will now be conducted to identify if the occurrence of sulfuric and sulfidic materials can be predicted by developing linkages with the surrounding landscape that may influence the soil forming processes. One possible soil-landscape process to better explain this trend may be that more wetting and drying occurs in the upstream wetland environments which could result in more oxidation of accumulated sulfides compared with the downstream wetlands that are mostly kept saturated especially adjacent to the irrigated areas below Mannum. Other models to help explain the trend may discuss the flow of carbonates into the system from the surrounding regolith and soils to enhance neutralising capacity. An improved understanding of the soil landscape dynamics will assist with future predictions on the occurrence of acid sulfate soil materials.

**Acknowledgements**

This work was largely funded by the Murray-Darling Basin Authority (MDBA). We would like to acknowledge the input of staff from: (i) CSIRO for laboratory and graphics work (Greg Rinder), (ii) South Australian Murray-Darling Basin Natural Resources Management Board for project support (especially Rebecca Turner, Kate Mason and Michael Fogg) and (iii) review comments and project management by R. Kingham (MDBA).

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# What is the average rate of soil formation?

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Knowing the rate of soil formation has become an important topic considering of soil as a renewable resource. We investigated the rate of soil formation of Australian soil and compared it with the rates in other parts of the world based on literature that used terrestrial cosmogenic nuclides. We found that the rate of soil production (conversion of rock to soil) mostly followed an exponential decrease with increasing soil thickness. The rates of soil formation in Australia appear to be in similar ranges as compared to other parts of the world. Soil production rate data from different lithologic conditions and various climate regimes vary between 10 to 100 mm/kyr. We also compare the data with rate of chemical weathering. The average soil production rate and its implication is discussed.

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## SOIL PHILOSOPHY, SOIL EDUCATION AND THE FUTURE OF SOIL

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# Capturing and communicating soil and landscape knowledge

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### Abstract

Knowledge management involves the conversion of implicit or tacit knowledge into explicit and accessible formats. During the past 17 years, land resource assessment staff in DPI, and predecessor Departments, have been working with retired specialists to develop information about Victoria's soil and landscapes. This approach has relied on partnerships with key soil and landform subject matter experts, including a 'knowledge network' of retired specialists. Since 1997 this has been done in parallel with the Victorian Resources Online (VRO) website as a route-to-market for this information. Initially, web content consisted largely of image-based maps and associated information documents. More recent incorporation of interactive visualisation products (such as videos, landscape panoramas and animations) is providing interactive rich-media content. These interactive products are providing an effective means for both capturing and communicating expert knowledge with spatial context.

### Key Words

Knowledge management, implicit knowledge, explicit knowledge, internet, visualisation, animation, soil/landscape, geomorphology.

### Introduction

Classifying, mapping and interpreting the nature and distribution of soils and landscapes are primary tasks for geologists, geomorphologists and pedologists. Although there are prescribed approaches and standards for consistent description and measurement of landform, lithology and soil properties, the integration of all these elements into a map depends on the experience and subjective skills of the surveyor who, invariably, has to develop a conceptual understanding of the landscape's evolution and its contemporary processes. These tacit understandings are spatially specific, as they are built within the individual's field-based experience, and can be brought to bear on re-interpretation of old data in the light of new technology, development of communication products, and succession capability within an organisation. This is achieved through knowledge management.

#### *Knowledge management*

Knowledge management was highlighted as a critical issue in Victoria's most recent catchment condition report (VCMC 2007). For organisations in the future, the cost of lost knowledge is to become an increasingly compelling issue (De Long 2004). Codifying accumulated experience is critical in the case of employee renewal or when turnover is substantial, and few firms make a systematic effort to uncover this knowledge (Boiral 2002). Raymond *et al*. (2010) reviewed different definitions of knowledge within the environmental management literature. They recognise various types of 'contexted' knowledge, including indigenous, local or situated, tacit, implicit, informal and expert. Knowledge derived from experience (i.e. experiential) can be broadly classified as 'explicit', 'implicit' or 'tacit' knowledge. 'Explicit' knowledge has been articulated in a form that is accessible to others (quantitatively or qualitatively), whereas 'implicit' knowledge can be (but has not yet been) articulated and belongs to individuals (Fazey *et al*. 2006). Explicit knowledge exists in a written (i.e. codified – numerical or graphical) and categorical form that is widely accessible. 'Tacit' knowledge cannot be articulated (Nickols 2000). 'Local', or situated, knowledge reflects an understanding of local phenomena (Robertson and McGee 2003) – contexted to a local situation. Knowledge management involves the conversion of implicit and tacit knowledge into explicit knowledge and information. There is a wealth of implicit knowledge of Victoria's soil and landforms held by experienced individuals, many who are now retired or may retire in the next 5 years.

*Information and knowledge sharing on the world wide web – Victorian Resources Online*

Many agencies have realised the value of the internet to enhance their knowledge capture, retention and sharing abilities (Denner and Diaz 2011). Knowledge repositories are usually intranet sites or portals that preserve, manage, and leverage organisational memory (Dalkir 2005). In Victoria, since 1997, the Victorian Resources Online (VRO) website (www.dpi.vic.gov.au/vro) has been the key means for capturing and disseminating knowledge of Victoria's soils and landscapes (their nature, distribution and associated processes) as web-based information products. Information presentation on the website is being continually enhanced with incorporation of rich-media elements such as videos, animation, interactive panoramas to support the more standard web content, i.e. text and graphics. These new approaches provide the means to both capture expert knowledge and communicate it to a broad audience. VRO is therefore a knowledge repository that preserves, manages and leverages both current and predecessor specialist knowledge as web-based information, particularly related to soils and landscapes (Imhof *et al*. 2011).

*Harnessing expert knowledge – 'new wine from old bottles or old wine into new bottles'*

A Geomorphology Reference Group (GRG) was established in 1995 to review the mapping and description of Geomorphological Divisions in Victoria. The group, convened by David Rees, includes seven retired pedologists, geologists and geomorphologists who, between them, have over 200 years experience in soil, landform and geological survey in Victoria. The VRO project has been working closely with the GRG since 1997, forming an informal 'knowledge network' to guide the development and delivery of the Victorian Geomorphological Framework (VGF) and associated web-based information. VRO web pages contain clickable maps depicting geomorphological units for each region of Victoria, linked to map unit descriptions developed by GRG members (http://vro.dpi.vic.gov.au/dpi/vro/vrosite.nsf/pages/landform_geomorphology). These descriptions are now being augmented with visualisations of landscapes and landscape processes developed in partnership with relevant experts. The GRG-VRO network and partnership is an example of how to harness the implicit knowledge of scientists, to create accessible information suitable for a broad range of users (Imhof *et al*. 2011).

**Approach**

The GRG has met a few times annually since 1995. New technologies and data, such as digital elevation models and gamma radiometrics imagery, have been utilised as part of this process, together with field visits. The regional and local area expertise of the GRG members has also been utilised opportunistically in regionally focused projects to provide more explicit spatial and process context to new work being undertaken and to educate existing staff.

Visualisations on the VRO website are used as a means of capturing and communicating the knowledge held by GRG specialists. Animations have been developed from 'storyboards' (a series of hand-drawn sketches that outline all the events in the animation) created by interviewing specialists. Experts have also been captured on video in the field explaining soil and landscapes and associated processes. Videos and animations are made available on the website using a browser plug-in that provides the functionality for flash-based content to be displayed. Three hundred and sixty degree panoramas of representative landscapes are captured in the field with camera and tripod and then 'stitched' together and converted to a series of 'faces' that can be displayed on the web (using the same flash-based browser plug-in) to provide an interactive landscape 'exploration' experience. Users can move from one panorama image to another (i.e. to different parts of the landscape) and access associated landscape related information (e.g. soil descriptions, observations, site data). Audio-visual elements such as video and audio are now integrated within these landscape models.

## Results

Two examples of web-based information products, created through the knowledge management techniques described above, are provided. These relate to explaining the nature and distribution of soils and landforms in the Maffra region as well as complex soil processes in agro-ecosystems.

*Soil/landscape mapping in the Maffra region – static maps, landscape panoramas and videos.*

Soil/landform mapping is based on a conceptual model that usually resides 'in the head' of the expert surveyor (i.e. tacit and implicit knowledge). Soil/landform mapping in the Maffra region of West Gippsland (Sargeant and Imhof 2000) has been available on the VRO website since 2000 as clickable image-based maps (e.g. Fig. 1). This mapping is based on a conceptual framework that explains the development of the soils associated with a series of terraces formed during the Pleistocene period. Diagrammatic representations of terrace development were provided in earlier web-based versions on the website. Recently, an animation has been created with Ian Sargeant (a GRG member who specialises in mapping the soils in the Gippsland region of Victoria) to depict this in an interactive way and to show the relationships to soil profile development (Fig. 2). Ian has also been engaged for visits to create audio-visual records of his explanations in the field.

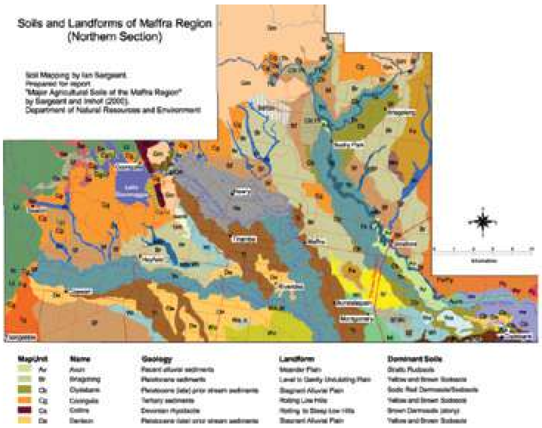


**Figure 1. Soil/landscape image map**

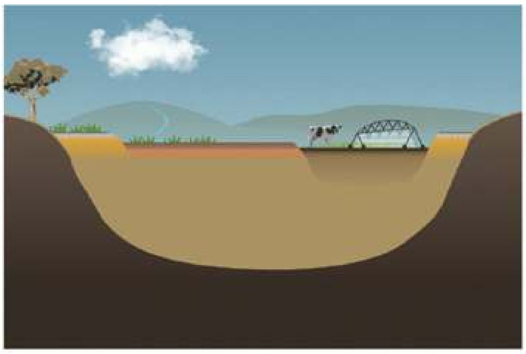

**Figure 2: Scene from terrace development animation**

A series of 360 degree panorama images of landscapes, representing key soil/landscape mapping units, are accessible online with zoom and pan functionality and clickable images and text boxes (Fig. 3). Videos of Ian Sargeant describing soils and landscapes in the field have been integrated into the panoramas as well as images and descriptions of representative soil profiles (Fig. 4).

**Figure 3: Web-based landscape panorama**


**Figure 4: Video of Ian Sargeant describing landscape features in the field**

*Soil processes*

Animations have been developed with scientists to visually explain soil processes. To date, animations have been published on VRO that depict the soil carbon and nitrogen cycles, and processes related to soil acidification and coastal acid sulfate soils, all within the context of an agro-ecosystem. In the case of nitrogen animation (Fig. 5), five soil scientists provided input, each with a different focus (e.g. farming system, soil biology, chemistry, greenhouse gas perspectives). This animation consists of twelve frames displaying graphical elements, with voice-over and transcript incorporated.

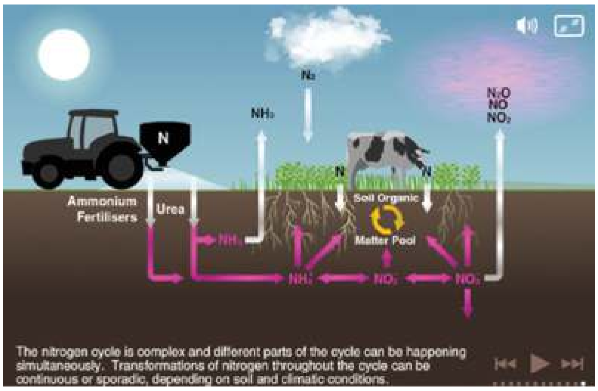


**Figure 5. Scene from N cycle animation**

### Evaluation

Unsolicited feedback to date has highlighted the value of animations to extension staff and University lecturers. User testing is currently underway to investigate whether it is easier to gather information from animations compared to basic text. Eye tracking testing will also be carried out on some users to record how they visually interact with web-based visualisations. User profiling (based on IP address tracking) indicates that 87% of usage of animations is not possible to attribute to specific organisations (much of it via national and international ISPs). The education sector is a significant user (accounting for around 6% of all visits) and 3% of visits are from Victorian Government users.

### Conclusions

In an era of corporate downsizing and loss of long-serving expert staff due to retirement, the need for harnessing knowledge and making it available via information and knowledge repositories is as strong as ever. Websites such as VRO provide opportunities to capture and facilitate access to this information and knowledge in forms that can be utilised by a diverse audience. There is an ongoing need to capture expert knowledge as specialists are aging and not being replaced with new sources of knowledge. Making this knowledge more explicit, and in ways understandable to a broader audience, is possible with appropriate visualisation techniques. Partnerships with subject matter experts and effective use of visualisation are two key elements of our current knowledge management activities. VRO now incorporates both static and dynamic visualisation products (with varying degrees of interactivity) that have been developed in association with these experts. The integration of video files, animation and interactive visualisations into virtual landscape models provides audio-visual richness to otherwise static web pages and supports more interactive and dynamic exploration of content. These are useful tools for both 'knowledge capture' and 'knowledge transfer' and allows for the legacy of these experts to be recognised.

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# The role of landholder education in adoption of soil health management systems

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### Abstract

Management for soil health has received increasing attention, but, despite this, adoption of soil health management plans (SHM) has been slow and is possibly affected by landholder education. This paper investigates the role of landholder education in the adoption of SHM systems, using salinity and sodicity as indicators. Through the use of a landholder response mail based survey consisting of Likert scale rank questions, categorical responses and open ended questions, education was shown to mildly affect the adoption of SHM programs, but was not considered an overriding impediment by landholders. However, there is a disparity between education as an impediment and landholder knowledge. This disparity is potentially overcome by a reliance on agronomists and extension officers to guide landholders through SHM issues that they find complex. In terms of managing soils for salinity, education was shown to be adequate, although for sodicity education is still a major limiting factor.

### Introduction

The term 'soil health' has become increasingly prevalent in scientific documents, advertisements for agricultural company services, departmental extension programs, government-based discussion and policy, and farming communities. However, the adoption of soil health management (SHM) programs in Australia has been slow. Farmers remain hesitant to implement structured management plans tailored to address soil health, despite accumulating scientific evidence for the credibility of certain soil health indicators, increased reporting of program benefits, and progress in communicating these. Lobry de Bruyn and Abbey (2003) observe that soil health programs and their indicators are often too complex to be implemented by farmers independent of external assistance and advice. Hence, landholder education is potentially one way in which to address SHM adoption.

With reference to two specific SHM issues, salinity and sodicity, Watson *et al.* (2000) reported that 54% of Australian local governments questioned believed that sodicity *was not* an issue in their area, with a further 22% unsure, while 72% of local governments believed salinity *was* an issue, with only 9% unsure. These are curious statistics considering that sodicity currently affects ~340 million ha (Murphy 2002) and salinity is comparatively forecast to affect only ~4 million ha of Australian land by 2050 (Robertson 1996). Furthermore, this disparity is concerning because local governments are usually comprised of landholders, are responsible for the local farming community, and should be up-to-date with issues affecting their local community. Irrespective of this concern, Hajkowicz and Young (2005) make the assumption that landholder awareness of sodicity is really quite high, resulting from educational/training programs in the decade prior to 2005. However, this was not further explored.

This paper investigates the role of landholder education in the adoption of SHM systems, using salinity and sodicity as indicators.

### Methods

The data used in this study was collected from landholders, via a mail-based survey, in the Lachlan and Macquarie Valleys of New South Wales. The survey questions used for this paper consisted of Likert-based scales (Likert 1932), with categorical selection used for salinity and sodicity definition testing. There was opportunity provided for an open response concerning the impediments for the adoption of SHM strategies. Technical terms used in the construction of questions were representative of those often used by agronomists, extension agencies and landholders. The survey was based on a survey template used in the studies of May (2006) and Mylek (2006), with further reference to the tailored design method (Dillman 2007). The information sent to each participant included a letter of explanation, the survey, a return addressed envelope and a stamp. A participant database was obtained from various Livestock Health and Pest Authorities (LHPA) within the survey region; this database constitutes rate payers within each individual LHPA region. This was supplemented, where necessary, through the use of the White Pages®, cross-referenced with a real-estate database for landholding size and title details. A mailing list was selected at random and stratified using demographics of council regions. The survey sample only comprised of landholdings equal to, or exceeding, 60 ha. The number of eligible participants was 719.

The response rate achieved after the initial send out of the survey and one reminder card was approximately 20% ($n$=144, $N$=719) following exclusion of ineligible participants. Non-response bias was not evident, and was assessed by obtaining a second sample ($n$=96, $N$=100) from non-responders and comparing the frequency of auxiliary variable distributions for respondents and non-respondents.

## Results

### *Education and training impediments*

Of the education and training impediments listed (Table 1), lack of research into broadacre SHM (3) and lack of expert advice or assistance for SHM, other than an agronomist (4), represent the greatest impediment to adoption of SHM plans; 47% (N=115) and 44% (N=125) responded with 'large impediment', respectively. Responses to the statement concerning the time taken to learn SHM skills (5) were generally spread evenly across the impediment scale, while not knowing enough (1) and not enough ongoing technical advice (2) were spread across the slight impediment to large impediment categories. It is noted that the majority of respondents to statement (2) indicated a moderate or large impediment (66%, N=127).

**Table 1: Education impediments to the adoption of a soil health management plan as ranked by landholders of the Lachlan and Macquarie Valleys**

| Education and training impediments to the adoption of soil health management plans | Frequencies (%) Not an impediment | Slight impediment | Moderate impediment | Large impediment | Sample size (N) |
|---|---|---|---|---|---|
| 1. I don't know enough about soil health management | 17 | 26 | 27 | 30 | 130 |
| 2. There is not enough ongoing technical advice on soil health management | 10 | 24 | 34 | 32 | 127 |
| 3. There has not been enough research into large-scale/broad-acre soil health management | 7 | 24 | 23 | 47 | 115 |
| 4. It is difficult to get expert advice or assistance for management of soil health, other than an agronomist | 18 | 18 | 21 | 44 | 125 |
| 5. It takes too much time to gain the knowledge and skills needed | 25 | 30 | 19 | 25 | 130 |

Various landholders suggested that they now knew what it was they had to do in the future to manage for soil health based on their past experience, or past experience of others. For example:

> *"My wife and I have been running our farm for 9 years taking over from my parents… when it does rain we will have the hindsite (sic) we need to take advantage of every drop of rain and so improve soil quality"* (L24)
>
> *"Blindly following what dad did. Lesson learnt. I now know what I need to do"* (L120)

Others indicated that they didn't know what some of the soil health characteristics used in this study were, or what their impact on soil productivity was:

> *"…a lot of the issues you raised I am unfamiliar with, I am sure I have most of the other problems, we have very little help with & reduction of soil deficiencies – I have no idea in \$ terms what they cost me in production."* (L16)

### *Soil health factors*

Landholders were asked to rate the importance of various SHM factors (Table 2) to the management of their properties. All of the factors listed received a vast majority of responses in the 'highly important' category, with the exception of sodicity, slaking and electrolyte. The percentage of respondents who selected 'don't know' for sodicity, slaking and electrolyte was also notably higher than the remaining factors (17%, 34% and 33% respectively; N=144). Additionally, landholders were questioned on salinity and its definition. Given the direct relationship between salinity and electrolyte concentration, there was an interesting difference in numbers of respondents being unfamiliar with either factor. Compared to the 33% unfamiliar with electrolyte, it was found that 12% of landholders were unfamiliar with salinity and a further 1% selected an incorrect definition (N=135); 87% correctly identified the definition.

Organic matter content and soil structure were represented as the factors that the majority of landholders placed as most important to their management (86%, N=132 and N=131 respectively). Despite the relationship between sodicity and soil structure, sodicity was only considered to be highly important by 41% with a further 32% suggesting sodicity to be of no importance to their property management (N=110). When questioned about the definition of sodicity, 36% of landholders selected the correct definition, while the remaining 64% either did not know, thought sodicity was the same as salinity, or confused the definition of sodicity with its consequences (N=143).

**Table 2: Landholder ranked importance of selected soil health management factors for the Lachlan and Macquarie Valley**

| Soil health factor | Response as a valid percent of N (%) | | | Sample Size (N) | Don't know** (%) |
|---|---|---|---|---|---|
| | Not important | * | Highly important | | |
| Organic carbon | 10 | 14 | 77 | 125 | 9 |
| Water infiltration | 5 | 9 | 87 | 129 | 5 |
| Sodicity | 32 | 27 | 41 | 110 | 17 |
| Nitrogen | 2 | 15 | 84 | 135 | 3 |
| Microbial diversity | 5 | 14 | 81 | 129 | 6 |
| Phosphorus | 2 | 17 | 81 | 136 | 3 |
| Slaking | 40 | 32 | 28 | 88 | 34 |
| Organic matter content | 1 | 10 | 89 | 132 | 4 |
| Electrolyte | 11 | 34 | 55 | 85 | 33 |
| Soil structure | 2 | 9 | 89 | 131 | 5 |
| Soil erosion | 16 | 10 | 74 | 136 | 2 |

* Those selecting a category between 'not important' and 'highly important'

**Those who selected 'don't know' reported as a percentage of the total response NT=144

## Discussion and Conclusions

*Education as an impediment*

Education as an impediment to SHM was shown to have a moderate influence. With the exception of broadacre research and expert advice, responses were relatively evenly spread over the impediment categories. While at least a quarter of landholders indicated education as a large impediment, the general consensus is that education does not have an overriding influence on the implementation of SHM programs. Comments made by landholders suggest that past experience provides adequate knowledge to continue with SHM. However, there appears to be conflict between the influence of education as an impediment and landholders' knowledge; i.e. there may be a propensity for landholders to think they understand adequately, irrespective of whether or not they do. This is highlighted in the current research through: (i) landholders' tendency to select 'not important' for soil health factors, such as slaking, that also had the highest proportion of landholders select 'don't know'; (ii) the number of landholders who actually know what sodicity is, compared to those who said it was important to their management; and, (iii) direct comments made by farmers with reference to their knowledge of soil health factors. Therefore, it is quite possibly the case that soil health education is still a major hurdle to consistent SHM.

Although landholders may have overrated their understanding of soil health issues, this does not necessarily make education an impediment to adoption. In the same way that Kelly *et al.* (2009) suggests that there is an over-reliance on agronomists and extension agencies, it is quite likely for those with a lesser SHM education to feel comfortable in implementing such a program through relying on the supervision and advice of their agronomist or local extension officer.

The extent to which education is perceived as an impediment to SHM adoption is also influenced by those who design and communicate the innovation. Landholders generally want and seek to understand processes and information that will aid them in their farming enterprises, although reason provides that simple innovations are likely to be adopted over those that are complex (Guerin and Guerin 1994). The current results indirectly show that the complexity of SHM is still a major concern for landholders. Almost half of the farmers indicate that there is a requirement for more ongoing expert advice or assistance for SHM, while approximately a third believe more ongoing technical advice is required. This shows a reliance on experts and technicians in order for landholders to be able to sustain a structured and consistent SHM program. Subsequently, it can be deduced that SHM is complex. It is not necessarily possible to make the soil physical, chemical and biological systems and interactions less complex, so it should be kept in mind by those promoting structured SHM programs that adoption longevity is reliant on ongoing external advice. While we agree with Kelly *et al.* (2009) that farmers are at risk of losing connectivity with their land, the only apparent way around a reliance on external advice is through further and higher education, which is not necessarily an option. Therefore, this requirement for external advice must continue, but landholders should be encouraged to remain involved on all levels, from on-the-ground decision-making through to the conduct of research, contrary to the beliefs of Sojka and Upchurch (1999).

*Sodicity versus salinity*

Environmental salinity campaigns such as "Halt the Salt" continue to endure, having received much public attention and concern in the past three decades. So, it was not surprising to observe that 87% of responding landholders correctly identified the definition of salinity. In fact, it might be asked why the remaining 13% did not understand salinity as an environmental issue. Comparatively, sodicity has received less public exposure (Hajkowicz and Young 2005) and the results in this study reflect this, with only 36% of landholders correctly identifying the definition of sodicity. It may be suggested that only those who have sodic soils could be expected to understand the issue. However, given the relative affected land estimates, there is still an obvious disparity between the knowledge of salinity and salinity affected land as compared to sodicity and sodicity affected land. Northcote and Skene (1972) estimate that 47% of NSW is affected by sodicity, with the majority of affected land west of the Great Dividing Range (McKenzie *et al.* 1993). This increases the likelihood of sodicity being a SHM concern for the Lachlan and Macquarie Valley landholders.

Hajkowicz and Young (2005) further suggest that farmers have an intimate knowledge of their land and how it responds to treatment, such as applications of sodic ameliorants. Their argument extends to reason that if farmers are well aware of solutions, then it is possible that the marketplace has identified an optimum level of treatment; i.e. the decision to not address sodicity is a private investment based one, rather than a function of information failure. While it is plausible that farmers may not address sodicity, even if they are aware of it, they must be aware of it to make this decision. The results for the understanding of sodicity versus salinity do not support this notion. Hence, education is still a limiting factor where sodicity is concerned. Once again, the role of scientists, extension agencies and agronomists will be important in addressing this.

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# Soil knowledge management: Needs and ways of knowing for a more resilient rural landscape

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### Abstract

As a boundary organisation involved in a research programme we offer a space that allows for non-linear discussion of research findings and the potential for farmer experimentation/expressionism of acquired soil knowledge and its application. The ability to transform scientific knowledge to specific contexts is often highly dependent on willingness of farmers to co-operate, to take an idea on and place it into action. As a boundary organisation are we able to mediate between accepted soil knowledge and local knowledge systems to improve the level of practice change for a more resilient farming system? Also are we meeting farmers' soil knowledge needs and preferences for knowledge interaction especially in terms of maintaining soil fertility which is important to the resilience of a farming system? We analysed 8 field days (about on-farm research) and 1 forum with a total of 138 participants (57% farmers, 11% agribusiness, 12% researchers, 17% NRM and 2% other) to examine if we are creating the "space" for such discussion of soil knowledge needs and ways of knowing.

### Key Words

Soil knowledge, resilience, farmer experimentation, local knowledge, scientific knowledge, interaction

### Introduction

Research into the value of indigenous/local knowledge has occurred where subsistence agriculture still exists, mainly in Africa, South America and Asia (WinklerPrins and Sandor 2003). However, in nations with well-developed production agricultural systems there is less recognition of or research into local knowledge and its role in improving: soil fertility decline recognition or land management practices, with some exceptions (Romig *et al.* 1995; Lobry de Bruyn and Abbey 2003). The recognition by land managers of a decline in soil fertility is paramount as it is a critical slow variable that can affect the resilience of a farming system and its ability to adapt to future shocks and perturbations. It is well accepted amongst scientists that maintaining soil fertility for agricultural production is important, and it is also widely known that Australian soils are inherently infertile. Yet, this accepted knowledge of Australian soils is not reflected in land management practices when in 2009-10, 39% of 32 680 farm businesses across Australia did not apply any form of fertiliser, with the study region close to the national average (ABS 2011). In addition, for those farmers in the study region who did apply fertiliser, less than half did so based on soil testing (ABS 2011). In this paper we examine the soil knowledge needs, ways of knowing and awareness of risks to the farming business in a National R&D program – Grain and Graze II. The programs overall aim is to make a significant contribution to "increasing knowledge, capability, respect and confidence of mixed farming communities, thus enabling them to manage a more viable and environmentally sustainable mixed farming system that can adapt quickly to changing climatic, market and policy conditions" (Project Specification Document DEEDI/GRDC DAQ00162, p2 unpublished). The northern New South Wales component plans to deliver on practice change by increasing farmer adoption in: planting perennial species; maintaining ground cover at 40% (cropping) and 60% (grazing systems); diversifying crops/pastures and forages; planting of ley pastures and use of reduced/zero tillage, largely in priority sub-catchments of the Border Rivers-Gwydir Catchment Management Authority (BR-G CMA). How effective we have been as a mediating structure in promoting these land management activities, forming local networks with local farmers, building knowledge and capacity, and co-operating with farmers in conducting on-farm research, has all been assessed through evaluations of the 8 field days and 1 forum held in 2011-12.

### Methods

*Data Collection*

The monitoring and evaluation approach and survey was approved by the University of New England's Human Ethics Committee (approval code HE10-212). The research team designed an entry and exit survey that can be used repeatedly at all field days and forums to address the key objectives of the Grain and Graze II programme. A number of field days and forums were held over the 2011-12 that collected both entry and exit data (Table 1). All of the field days were focussed on a co-operator (farmer) who was conducting paddock-scale experiments of particular practices that address the sustainability of the mixed farming enterprise. The forum, specifically addressed the nature of an ideal farming system for the Delungra area. The field days and forum are summarised in Table 1. The monitoring and evaluation examined 4 key questions through a combination of likert questions and accompanying follow-up open-ended questions with a written response. Several responses to likert questions were combined to address the 4 key monitoring and evaluation questions which are shown in Table 2. The likert scale was from 1 to 5 with a value of 1 suggested low level of agreement with the statement whereas a value of 5 indicated high level of agreement with the statement. To reflect on the meaning of the likert responses the follow-up open-ended question allowed for farmers to clarify their response. In some of the field days the attendance was too low to be worthwhile in terms of analysis, and the other difficulty we had was that some people still returned an incomplete form, while others make no attempt to provide written feedback. In future field days we made more certain that evaluation surveys were returned complete to reduce the incomplete survey percentage and increase the level of written feedback.

**Table 1: Field days and forum as part of Grain and Graze II Northern NSW in 2011-12.**

| Location | Topic | date | No of Agribusiness participants | No of Farmers | Respondents to entry survey | Respondents to exit survey | Total attendance at event |
|---|---|---|---|---|---|---|---|
| Glenwood, Gravesend, | Effect of gypsum, and nitrogen on pasture production | 29 March, 2011 | 1 | 4 | 4 | 4 | 7 |
| Weegowrie, Inverell, | Role of summer cover cropping with Lab Lab. | 6 April, 2011 | 1 | 4 | 4 | 4 | 7 |
| Claire, Elsmore, | Effect of gypsum, potassium and nitrogen on pasture production | 6 May, 2011 | 1 | 5 | 5 | 6 | 10 |
| Weegowrie, Inverell, | Manure on forages | 15 September, 2011 | 1 | 3 | 0 | 3 | 6 |
| McMaster Field Day | 5 topics including compaction, pasture-crop transitions and legume varieties | 22 September, 2011 | 5 | 16 | 14 | 20 | 35 |
| Nullamanna, Inverell, | Phosphorus and sulphur on legumes | 25 October, 2011 | 1 | 9 | 7 | 7 | 14 |
| Forest Hill, Inverell, | Using Coolatai Grass | 27 October, 2011 | 1 | 14 | 10 | 10 | 20 |
| Utah, Delungra, | Soil nutrition specifically magnesium | 22 March 2012 | 3 | 10 | 9 | 10 | 19 |
| Delungra Forum | What is an ideal farming system for the Delungra area? | 30 March 2012 | 1 | 12 | 9 | 8 | 20 |
| TOTAL | | | 15 | 77 | 62 | 72 | 138 |

### Study area and area of field day impact

The BR-G CMA services the entire Gwydir Catchment (approx 26,500km$^2$) and the NSW portion of the Border Rivers Catchment (approx 24,000km$^2$) (Fig. 1). Both of these catchments are located within the Murray-Darling Basin, in south-eastern Australia. They are bounded by the Queensland border in the north and west, the Great Dividing Range in the east, and the Namoi Catchment in the south (Fig. 1). The area covered by the field days approximated 59 182 hectares of farming land, and the percentage of cropped land varied on average was 31% (10-50%) and the percentage used for grazing averaging 81% and varied from 67 to 100%. Hence there is a dominance of graziers attending our field days. Farmers indicated the areas of interest for future field days were mostly soil issues (66%, n=72) which included soil fertility, nutrient management, soil Carbon and land capability, pasture (52%, n=72), and pasture-crop transitions (25%, n=72).

## Results

The analysis is from 8 field days (about on-farm research) and 1 forum with a total of 138 participants (57% farmers, 11% agribusiness, 12% researchers, 17% NRM and 2% other) to examine if they are achieving the goals of the Grain and Graze II programme, and developing networks in priority sub-catchments that have a legacy beyond the research funding period. The composition of field day attendance also reflects that there is a genuine 'space' where farmers, scientists and agronomists can interact and debate the merits of what they are hearing.

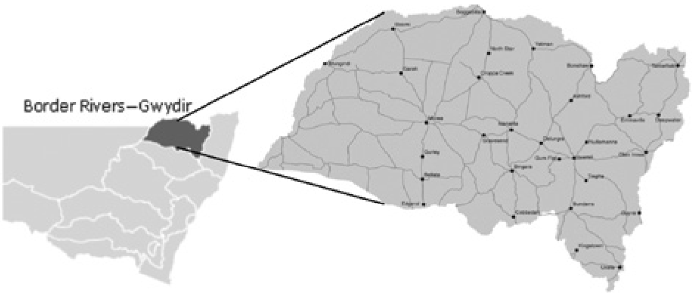


**Figure 1: Localities within the Border Rivers-Gwydir Catchment Management Authority where field days and on-farm experimentation occur as part of Grain and Graze II in Northern region of New South Wales.**

The Grain and Graze II program sought to highlight the areas of risk and uncertainty as identified by farmers as well as the perceived difficulties in implementing the practices demonstrated in the field days or discussed at the forum and as a counterpoint how willing they were to try some of the practices demonstrated at the field days. Content analysis of the entry surveys indicated that only a minority of farmer participants consider nutrient loss or soil health decline (12% of the 62 farmers who responded to this item) a risk to their farm business. They also focussed on the fast variables such as cost of production (13%, n=62) and commodity prices (10%, n=62), which for the most part are beyond their control and are always likely to fluctuate. It is the slow critical variables, like soil fertility, that are more important to manage in terms of improving farming system resilience. Other natural resource variables that farmers can manage that were also nominated as risks to the farm business were weeds (10%, n=62), and soil erosion (6%, n=62). The focus on production is also reflected in the farmers' responses to the question: What are you trying to achieve in your farm business? Most farmers indicated more than one of the four categories which were: Achieve whole farm sustainability (70%, n=62); Improve production per hectare (62%, n=62): Optimise profit (52%, n=62); and Other (3%, n=62).

The entry survey is a useful way of examining what segment of the farming community we were attracting to these activities and how they learnt of the event. From our entry survey the majority of farmers attending are over 50 yrs old (52%, n=62), which closely aligns with the average age from the ARMS 2009-10 of 55 yrs old for the BR-G CMA (ABS, 2011). The majority of the attendees had heard of the field days through the post (45%), and very few had learnt of the event through the internet (6%). This seemingly low use of the internet contrasts with the ARMS 2009-10 which reported 38% of farmers in the BR-G CMA used the internet for information and advice, but the question did not allow elaboration on specific use (ABS 2011). When farmers were asked how they would like to receive information for future Grain and Graze II events there was a much higher preference for email (57%) or mail (40%), with very few farmers indicating telephone (11%) or fax (3%). When asked why they attended the field day; 84% said the "topic", 51% said "location", 10% said "social opportunity", and 13% said "other" (n=72). Our survey respondents confirm a recent national survey of farm managers that highlights 57% of respondents attend field days and 34% attend training courses or workshops for land management practice advice (Ecker *et al.* 2011).

Table 2 summarises the four key questions targeted by the evaluation survey from the past 8 field days and forum. In brief there was strong agreement with questions 1 to 4 (Table 2). The potential for these days to provide a "space" for discussion and interaction was confirmed by a number of farmers who commented in the survey along the lines of:

- *great discussion environment,*
- *Locally based studies very pertinent to our style/methods of farming, hard to find such targeted information anywhere else,*
- *because it is very informative and it provides the opportunity to discuss issues with people who know solutions,*
- *a good way to exchange ideas, and keep up with new developments,*
- *it was interesting and allowed everyone to talk and give their opinion,*
- *good to get such great local knowledge appropriate to our climate and soil,*
- *exchange of expertise with locals,*
- *good learning environment,*
- *the best part was being able to see theories in action and hear speakers explaining their trials,*
- *conversation with the other farmers very informative*

**Table 2: Likert responses to the 4 key evaluation questions from Grain and Graze II field days and forum in 2011-2012 (n=72).**

| Field day /Forum | Q1. Did the event achieve its objectives? (rate from 1 to 5) | Q2. Was the event useful for participants? (rate from 1 to 5) | Q3. Did it contain new information? (rate from 1 to 5) | Q4. Likelihood of implementation and potential for practice change (rate from 1 to 5) |
|---|---|---|---|---|
| Weegowrie, Inverell | 4.1 | 4.0 | 3.7 | 4.7 |
| Weegowrie, Elsmore and Gravesend Field days | 3.7 | 4.1 | 4.0 | 4.0 |
| McMaster Field Day | 4.1 | 3.6 | 4.4 | 3.1 |
| Nullamanna, Inverell | 4.5 | 4.2 | 4.0 | 4.0 |
| Forest Hill, Inverell | 4.0 | 4.1 | 4.1 | 4.7 |
| Utah, Delungra | 4.3 | 4.0 | 3.4 | 3.9 |
| Delungra Forum | 4.4 | 4.1 | 4.1 | 4.5 |
| TOTAL | 4.16 | 4.01 | 3.96 | 4.13 |

The forum on farming systems asked participants whether it caused them to question their current thinking on what an ideal farming system is for the Delungra area, on a scale of 1 to 5, with 5 indicating a high agreement with the statement, and on average they were 3.8. So for the most part the audience did question what is an ideal farming system for the area? The areas of the farming system they questioned their management of the most was undoubtedly first soil and fertility management followed very closely by pasture and its management through grazing practice. Inevitably the two - soils and pasture - were mentioned in tandem, and were never unhinged but coupled in their responses to this question. When asked what part of the farming system are you now more appreciative of than you were before attending the forum, most responded with soil fertility or soil nutrients, and are more determined to undertake soil testing to know what limits their plant growth. In terms of how confident they feel in adapting farming systems research to their own farm the forum participants were equally divided between those who thought it could be done and those who thought it was not easy to adapt research findings to their circumstances – their soils are "unique". Nevertheless after hearing the panel speak most forum attendees said they would implement the findings they had heard about at the forum to their own place. When asked specifically what particular part of the farming system do you feel most confident in changing forum participants usually responded with the soil fertility or more generally pasture and crop management. A few were more specific in what they may try such as undertaking pasture and winter cropping for the winter feed gap. Interestingly these are the same areas that they are now more appreciative of after attending the forum. Most of the forum attendees felt they had learnt something new and rated it as 4.1. We can identify from the survey responses (Table 2) that those farmers who attend the field days find the experience very positive, and 96% (n=72) would recommend the field day to others. The success of a programme that seeks to use boundary organisations for the purpose of interaction, discussion and promoting practice and behavioural change needs to recognise that sustained effort over multiple occasions is required to gradually build trust and attendance numbers. Our own experience has recorded repeat attendance by individuals which is a sign that trust is being built, because as the farmers say once they are there the results are "obvious" and very rewarding. The ARMS 2009-10 (ABS 2011) also recorded 19% participation in programs/initiatives for the BRG CMA, but for those who did participate the benefits were multiple: new skills/information (67%), on-ground works implemented (52%), improved understanding of land management and environmental issues (45%), and improved community interactions and networks (22%).


## Acknowledgements

I would like to acknowledge the project team of Dr Chris Guppy, Mr Bruce Gardiner, Ms Nicole Gammie and Mr Simon Jasper, the attendees of the field days and those farmers with OFR, as well as GRDC for the project funding.


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# Stakeholder suggestions for the soil science curriculum

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### Abstract

This paper reports on a two-year soil science higher education learning and teaching project funded by the Australian Learning and Teaching Council. Collecting data from soil science stakeholders though an action learning process resulted in the development of a set of unique soil science teaching principles, which also included a proposed structure of outcomes for graduates with soil science expertise. A key challenge is for providers to meet the learning needs of competent graduates that represent the soil science community.

### Key Words

Action learning, teaching principles, volume of learning, graduate outcomes

### Introduction

The project was concerned with bringing together all relevant stakeholders to develop a 'national curriculum' which is defined as: *a curriculum that includes stakeholder considerations and is applicable at any higher education institution teaching soil science* (Field *et al*. 2012). This was an inclusive approach that aspired to synthesise the broad range of perspectives internal and external to academia. The stakeholders included:

- Graduates in the workplace
- Undergraduate students
- Employers in industry
- Australian Society of Soil Science Inc.
- Professional accreditation body (CPSS)
- Academics

An initial framework (Field *at al*. 2010) was used to indicate the relationships between learning and teaching and stakeholders, as shown in Figure 1.

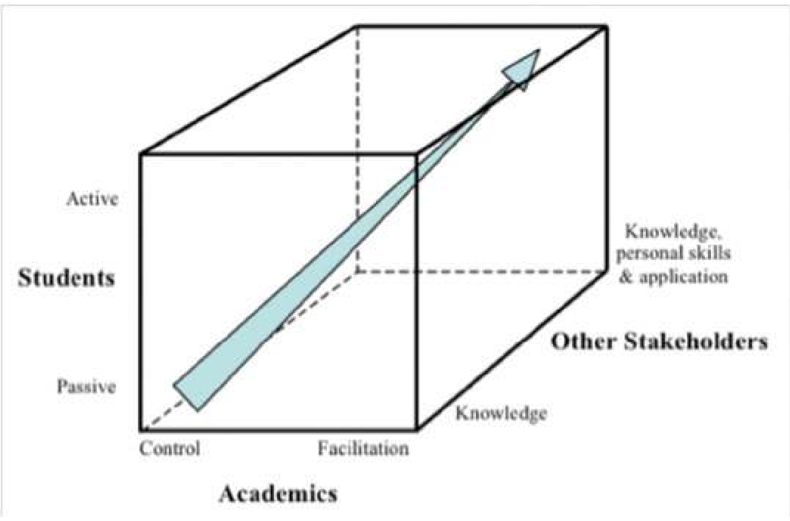


**Figure 1: A framework for stakeholder inclusion in soil science education (adapted from Field *et al*. 2010).**

Figure 1 indicates that the third dimension includes all stakeholders outside academia and that the outcome of soil science education is given by the direction of the arrow towards the right rear corner where students are active autonomous learners with the knowledge, personal skills and application to be productive soil science graduates.

### Methods

The project used a progressive action-learning model (Figure 2) where the participants moved through successive cycles building on the outcomes of the previous cycle. Such a model has been used in a related science discipline (Kelly *et al*. 2006). Figure 2 also represents a map of the project activities and outcomes, such as the development of Soil Science Teaching Principles with stakeholder input, and the creation of surveys for students, employers and graduates in the workplace during Cycles 1 and 2. The third cycle was concerned with the design and construction of joint teaching units based on the newly developed teaching principles. The forums were key focal points attended by representatives of all stakeholder groups and maintained the direction of the project.

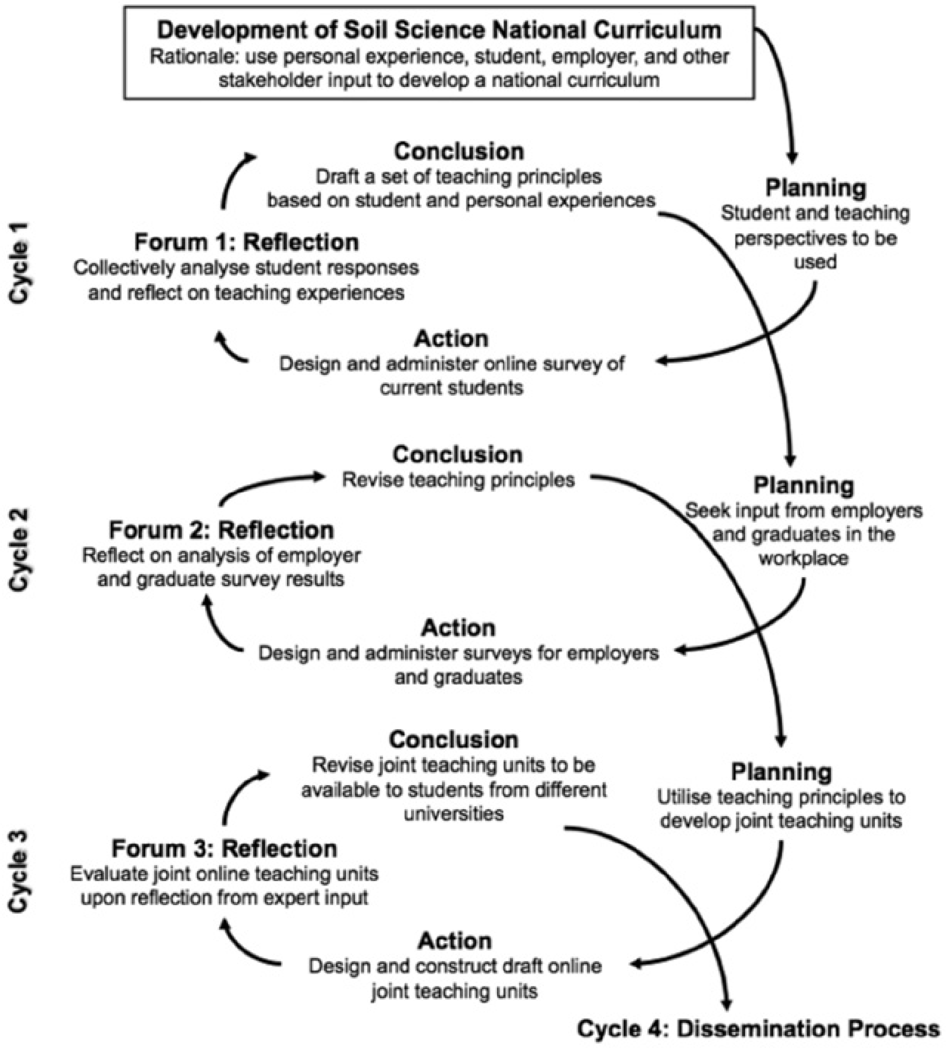


**Figure 2. Progressive action learning cycles used in the development of the project with each cycle forming the basis of the subsequent cycle (sourced from Field *et al*. 2012).**

### Results and Discussion

As shown in Figure 2, the development of the Soil Science Teaching Principles went through two cycles (two forums), and also incorporated feedback from academics and the surveys from undergraduate students, graduates in the workplace and employers. Forum stakeholder attendees were concerned with two key aspects of the teaching principles: they had to reflect the unique nature of soil science, and they had to have outcomes.

The eleven soil science teaching principles, including what it takes to teach the unique discipline and outcomes from applying the principles, were published in Geoderma (Field *et al*. 2011). A summary of the principles is given in Table 1 and illustrates the stakeholder input from the cycles, in both specific soil science teaching considerations and more generic teaching principles that are applicable to soil science.

**Table 1: Teaching principles for soil science as developed by the community of stakeholders (adapted from Field *et al.* 2011)**

1. Uniqueness

Soil Science is a scientific discipline that should be taught by people experienced in Soil Science who appreciate the uniqueness and functions of horizons (which define profiles), aggregates and colloids and are able to make connections within the discipline and with other disciplines. (See Churchman (2010) for unique aspects of Soil Science.)

2. Fieldwork

To demonstrate relevance and real-world connections and engage hands-on learners, use field and practical learning activities wherever possible and appropriate. Field activities are an important component of Soil Science as they help students to comprehend soil as part of the landscape and functioning ecosystem.

3. Jargon

With students new to Soil Science, use every-day language, relate to familiar or current issues, and introduce Soil Science jargon gradually.

4. Active learning

Assist students to derive Soil Science theory by using current real problems, scenarios and case studies.

5. Connections

To encourage the creation of connections, synthesis and integration, allow students to revisit concepts in different situations.

6. Systems

To assist students develop systems thinking and transfer knowledge laterally and vertically and to appreciate that soil is part of larger systems, emphasise the nature and role of soil in various natural, managed, social and economic systems at local, regional, national and global scales.

7. Communication

Allow students to interpret and present information and ideas in a variety of formats that resemble real-life scenarios where possible.

8. Authentic problems

Allow students to solve contemporary, authentic, challenging problems in groups to enable them to apply their abilities and experience, learn from the multiple perspectives in the cohort, reinforce concepts, and develop personal skills.

9. Feedback

Provide students with timely, constructive and plentiful feedback to aid their learning.

10. Assessment

The assessment regimes are aligned with the desired learning outcomes and group assessments are fair to all group members.

11. Outcomes

The outcomes resulting from the application of these principles are that graduates are proficient in 5 areas:

1. Identification, understanding and application of the unique features of Soil Science
2. The role, context and relationships of Soil Science to other disciplines and society as part of interrelated systems
3. Identifying problems and designing relevant contextual solutions
4. The ability to coordinate and function within and between relevant groups and effectively communicate results
5. Manage self for personal development and lifelong learning

The Outcomes given in Table 1 (Principle 11) reflect the initial model of stakeholder inclusion (Figure 1) in that graduates with expertise in soil science: should have a volume of connected knowledge about soil science (Outcomes 1 and 2); be able to solve problems (autonomously and in teams) in an academic or real-world context (Outcome 3); be able to communicate effectively with people (clients) who require soil science knowledge (Outcome 4); and have the attitude that they are responsible for on-going personal development and growth in a rapidly changing world.

Feedback from stakeholders (at forums and via surveys) enabled several recommendations to be made. These include the value of including industry professionals in teaching to provide further evidence to students of soil science relevance, practical application, and provision of real-life scenarios to enable problem-based learning. There was repeated emphasis on the importance of practical application, scientific rigour and the ability to communicate findings to a variety of different audiences.

## Conclusion and Future Challenges

Soil science is a unique discipline that requires teachers with the appropriate knowledge and skills able to apply the specific and relevant teaching principles to attain the outcomes required of graduates with soil science expertise. Apparently there is no BSc Soil Science available from any provider in Australia, and any soil science taught is part of some other degree such as Agriculture, Horticulture, Earth Sciences, or Environmental Sciences.

Figure 3 illustrates that different providers – in a higher education environment of proliferating providers – in different degree (or other qualification) programs impart different volumes of soil science knowledge, practical application and outcomes. A key challenge is for providers to know whether or not they have achieved a baseline (or minimum standard) to enable the graduate to be recognised (by employers of all kinds) as having achieved an acceptable outcome of credible soil science expertise. For example, how many hours of teaching by suitably qualified soil scientists (perhaps another challenge), learning activities, field-work, practical application, and interaction with industry will produce the desired outcomes for a graduate with acceptable soil science expertise as summarised in Table 1 (Principle 11)? Who can credibly answer such a question and based on what evidence?

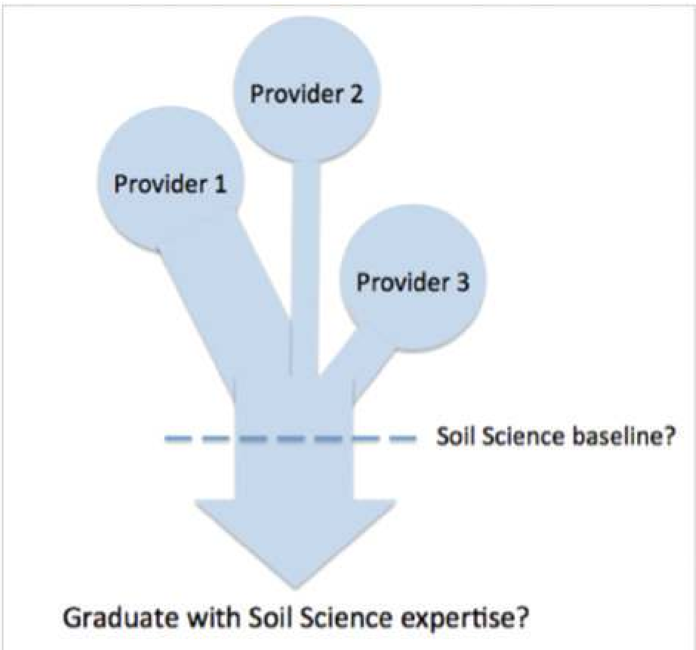


**Figure 3. Illustration showing that a graduate may have different volumes of soil science teaching by different providers in any degree program with an unknown soil science baseline resulting in an unknown level of Soil Science expertise.**

The challenge for us in safeguarding the discipline of Soil Science is to be able to define that baseline with the core body of knowledge required to ensure competent graduates.

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# Soil windows – bridging the gap between existing soil maps and the need for detailed farm-scale information

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Conceptual models of soil landscapes are used by field pedologists for soil survey. Many, but not all of these models have been presented in reports such as Singleton (1991). Advances in geospatial technology enable land managers to access soil information digitally reducing the demand for hard copy maps. These technologies present an opportunity for soil information to be applied directly to farm management decisions, potentially increasing farm profitability and enhancing environmental outcomes. A national soil mapping initiative (S-Map) is underway but may take more than a decade to complete. In the meantime gaps in soil information are reducing the effectiveness of farm management decisions. The Soil Windows concept aims to bridge the gap between soil information available at coarse scales and the land manager's requirements by providing detailed insights into soil landscape relationships that can be applied at the farm scale. A project is underway to develop the Soil Windows concept in the Waikato region, New Zealand. This project involves collaboration between experts, council, consultants and farmers through workshop participation to explore soil information knowledge and use. These findings will be used to develop and refine the Soil Windows concept.

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# Monday 3 December

## SOIL PHILOSOPHY, SOIL EDUCATION AND THE FUTURE OF SOIL

*Presented Posters*

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# Use of real-life problem solving to stimulate student thinking

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Many students find it extremely challenging to move from learning concepts and theory to the application of soil science in real-life situations. At La Trobe University we have for a number of years delivered a component of third year soil science as a problem-based learning project. This project incorporates an experiment, commenced in 1978, in which used a range of lime additions were added to an acidic silt-loam surface soil. The design is fully randomised with 3 replicates. The plots are planted with two species of legumes that do not normally perform well in acid soils. The students measure the effect of lime addition on soil pH, pH buffering, exchangeable cations and available P. The plants are harvested late in the semester, and plant biomass and nutrient concentrations in shoots are measured. Students are then expected to analyse the data, search for relevant literature and write a formal scientific paper, in addition to delivering a short presentation. Assistance during this process is provided in the form of lectures on the relevant soil science concepts, and importantly, how to critically analyse a scientific paper, and what makes a good presentation. As a result of this project, students are exposed to a number of soil science concepts including soil acidity and its effects on available nutrients, symbiotic nitrogen fixation and plant growth. Furthermore, students develop skills to solve reasonably complex issues within a multi-dimensional approach. Student feedback on this project has been extremely positive, with some of the final papers being of a very high standard.

# On old soil scientists, old soil and old lessons

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While the basic principles of field pedology may be acquired from textbooks and by attending university courses, much of the expert knowledge is learnt on the job through practical experience. Mentoring by experienced field pedologists is vital in the acquisition of tacit knowledge which new pedologists need to develop through practice of field pedology. Planning for succession of pedologists is important for organisations in order to maintain and build their capability in pedology. Capturing, documenting and archiving soil information is equally important to maintain and build knowledge and to allow for future research requirements.

Using an example from a project investigating changes in soil C&N by comparing current data to historic data from the same site, the importance of succession planning for pedologists and of both physical and paper/electronic archives will be illustrated. The national soils database (NSD) and the NSD soil archives are invaluable resources for investigating current status and changes in New Zealand soils. To develop the skills needed to practice field pedology, and to interrogate legacy pedological information, senior pedologists working at Landcare Research and other pedologists retired or otherwise employed with connections to the former DSIR Soil Bureau, have also been invaluable in locating NSD sample sites dating back to the 1960s.  Strengths and weaknesses of procedures employed by the current holder of soil information and expertise (Landcare Research) and also the historical provider (DSIR – Soil Bureau) will be presented, as will the lessons learnt.

# Valuing and using Soil Ecosystem Service Benefits (SESB) for catchment planning

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Catchment Action Plans spatially target what and where hundreds of millions of dollars of natural resource work will be done in NSW over 10 years. Soil ecosystem services concerning provision of water quality, primary productivity and soil habitat have been separately valued using a benefits approach within a resilience analysis framework to help support New South Wales government catchment action planning.

Maps of main variables that effect provision of quality water namely: surface clay and nutrients from modelled sheet erosion; salinity donation from hydrogeolandscape mapping; and soil and nutrients from gully erosion by soil type were spatially modelled and compared against surface water quality requirement maps for sub catchments using multi-criteria analysis shell for spatial decision support software (MCAS-S).

Similarly maps of soil productivity SESB were derived using; land and soil capability; intrinsic soil fertility; available water quality; land use and economic yield maps within MCAS-S

Value of soil habitat provision was also determined by assessing the relative rareness of lowest disturbance land uses against soil type and land and soil capability.

The three resulting ecosystem service maps were then combined to extract a map of maximum collective SESBs. Interventions were then planned for areas with highest maximum overall benefit according to maximum SESB type. This was done by using NSW Land and Soil Capability and Land Management within capability assessments along with analysis of MODIS satellite ground cover dynamics to target key land management actions to help achieve maximum soil ecosystem service benefit.

# Soil health surveys as an education tool for landholders

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Since 2007, Northern Rivers Catchment Management Authority (NRCMA) has conducted soil health surveys across 500 sites in north east NSW. The region has 8676 landholdings. These include dairies, grain and soybean farms, grazing properties, potato farms and subtropical lands producing sugarcane, coffee, fruit and nuts. Over 90% of the survey sites were rural lands. The remainder were natural bushland. A key aim of the program is to increase knowledge, skills and capacity of farmers to interpret soil sample results and manage their soil sustainably.

Soil characteristics measured included soil physical, chemical and biological attributes. The proforma from the Northern Rivers Soil Health Card was used to record attributes including ground cover %, ease of soil penetration, diversity of soil macrolife, root density and plant vigour.

Landholder selection covered a range of landuses and management styles. Landholders choose the sites and were encouraged to select a ‘typical’ site and one which was not doing so well or which they wanted to ‘improve’. Where possible landholders were present and participated in the sampling and soil health interpretation. Following laboratory analysis, landholders received a report comparing their results with other sites. The data was in a format that ensured confidentiality. Seminars were held in nearby towns, and landholders were encouraged to bring their individual reports and discuss their results. Feedback from landholders was generally positive. They were initially cautious of the CMA and how site data would be used. Over 5 years considerable trust has developed among project operators, the CMA and landholders.

# Estimation of carbon stock and emission in peatland areas in Kien Giang and Ca Mau provinces of Vietnam

**Vu Tan Phuong**

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The study was done in the framework agreement for cooperation between World Agroforestry Center (ICRAF) and Research Centre for Forest Ecology and Environment (RCFEE) on "Reducing emissions from all land uses, REALU". The objective of this study was to assess the potential for reducing emissions from peatland management in Vietnam. Research was carried out in Kien Giang and Ca Mau provinces, mostly U Minh Thuong and U Minh Ha National Park. Estimation of emissions from peatlands focused on emissions sources: biomass burning (forest and peart fires), oxidation of peat, agricultural production and peat extraction. Estimation of emissions follows IPCC guidelines (IPCC 2006). Data sources used come from statistical data on forest fires, land use changes and survey data. Study results showed that within 33 years (1976-2009), peatland area decreased at the rate of 530 ha/year and peat stock reduced at 6.59 millions tons/year (or 3.29 million tons C/year). The common land uses in peatland are peatland conservation (mainly in U Minh Thuong and U Minh Ha National Park), forest planting (mainly melaleuca plantation), agricultural production (*Deris elliptical*) and small sacle of peat exloitaion for fertilizer production. The study indicated that total emissions from peatland uses in two provinces is 12.76 million tons $CO_2$/year, in which emissions from buring of biomass and peat was the biggest, about 12 million tons of $CO_2$/year, accounting for 95% total emissions. Emissions due to oxidation of peat is about 0.6 million tonnes of $CO_2$/year, estimating at 4.7%. Peat exploitation for fertilizers production causes small emision, just about 107 tons of $CO_2$/year. Emission from peatlands in Ca Mau province is 8.01 million tons of $CO_2$, occupying 63% of total emissions of two provinces.

# Monday 3 December

## SOIL FERTILITY AND SOIL CONTAMINANTS (NITROGEN)

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# N supply and demand: Can controlled release fertiliser simulations help create the perfect match?

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Nitrogen (N) fertilisers play an important role in increasing crop productivity, but when the timing of N supply and demand are not well matched there may be a risk of N losses to the environment. The development of polymer coated fertiliser products may provide an opportunity to move towards more environmentally sustainable increases in productivity as these products allow the release of N from fertiliser to be controlled by the characteristics of the polymer coating. To explore the opportunities to better match N supply to demand and inform the optimal product usage, we implemented a routine for controlled release fertiliser into the APSIM model. The modelled N release mimics that of polymer coated fertilisers with an absorption stage (no release), a steady-state release stage, and a declining release stage, all of which have a $Q_{10}$=2 temperature effect. Simulations were performed using ‘products’ with different release patterns for a range of crops (wheat, barley, sugarcane, and vegetables), soil and climatic conditions. Key factors affecting the potential benefits of controlled release fertiliser were found to relate to the soil leaching potential, soil fertility level, rainfall distribution, crop rooting depth, and N demand and response patterns, as well as understanding the fertiliser release patterns. Climatic variability, uncertain seasonal forecasts and the complexity of the system make it impossible to ensure a perfect match between demand and supply, but the simulations through their potential for unlimited ‘experimentation’ do provide useful lessons that will inform the testing and use of these products under field conditions.

# Nitrous oxide emissions and the role of nitrification inhibitors to improve nitrogen management under elevated $CO_2$ conditions in legume based cropping systems

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Legume based cereal rotations are considered better in terms of a sustainable source of nitrogen (N) and less dependence on chemical fertilisers compared to non-leguminous rotations. However, legumes have a potential disadvantage of higher nitrous oxide (N2O) emission compared to non-leguminous crops. But, there is no evidence under elevated CO2 conditions. It was hypothesised that N2O emissions under elevated CO2 conditions in legume based rotation system (wheat-pea) will increase N2O flux as a result of increase in root biomass and root exudations that are used by denitrifiers. Nitrification inhibitors (NI) were included in the N management strategy as the experimental site has a high nitrification potential. The gas fluxes were measured using closed static chambers at the Australian Grains Free-Air Carbon dioxide Enrichment (AGFACE) facility in southern Australia. The targeted atmospheric [CO2]s were 380 (ambient) and 550 (elevated) mmol mol–1. Gas measurements were conducted at various times during the growing season. The detailed conclusion of results is under process but a significant interaction is found in the emission of N2O for crop type and different growth stages.

# Improving nitrogen fertiliser management by taking account of positive charge commonly found in highly weathered acid soils of southern Australia

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### Abstract

Highly weathered soils often contain kaolinite and sesquioxides of iron and aluminium. These minerals become positively charged under acidic conditions in subsoils where concentrations of specifically sorbed phosphate, organic and other anions are low. Positive charge, common in Western Australian (WA) cropping soils is likely to occur across southern Australia where these minerals occur, and affect nitrogen management. This work aimed at characterising the amounts of nitrate retained by positive charge under varying soil solution concentration and to measure the effect of charge on nitrate and chloride leaching. Charge was measured on two Arenic yellow orthic Tenosols. Leaching was measured using monolith lysimeters of these Tenosols and a Haplic Eutrophic Red Kandosol with no measurable positive charge. Positive charge of the Tenosols increased linearly with increased equilibrating solution concentrations. The 30-50 cm layer of these soils could potentially hold 12.5 and 45.5 kg $NO_3$-N $ha^{-1}$ at 4.0 mM nitrate concentration. This increases to 51 and 110 kg $NO_3$-N $ha^{-1}$ at 25.0 mM. In comparison, a database of WA cropping soils shows an average of 100 kg $NO_3$-N $ha^{-1}$ held in the 1 m soil layer. This can potentially meet the N requirement of a 2-3 tonne crop. Charge is likely implicated in profile nitrate retention. Positive charge delayed nitrate and chloride leaching by a factor of 2-3 compared with the uncharged soils. Delayed nitrate leaching and nitrate retention in acidic soils are currently ignored within N fertiliser decision systems for southern Australian crops. This may be leading to an overestimation of N requirement.

### Key Words

Anion exchange capacity, leaching delay, nitrate retention, nitrate adsorption coefficient, pore volume

### Introduction

Clay-sized fractions of highly weathered acid soils of southern Australia commonly contain kaolinite and sesquioxides of iron and aluminium. These minerals exhibit variable charge properties. The magnitude and sign (positive or negative) of the charge depend on soil pH and soil solution composition (Uehara and Gillman, 1981). They typically become positively charged under acidic conditions especially in subsoils where concentrations of specifically adsorbed anions such as phosphate and organic anions are low. Appreciation of these principles led us to measure and to confirm the widespread occurrence of positive charge in subsoils across the wheatbelt of Western Australia (WA) where these minerals dominate the clay-sized fraction of the soil (Wong and Wittwer, 2009). Positive charge common in WA cropping soils is likely to occur across southern Australian acid weathered soils due to similarity in clay mineralogy. It also often occurs in weathered acid tropical soils (Donn and Menzies, 2005). Positive charge is expressed as the anion exchange capacity (AEC) of the soil.

Because a hectare of soil 1 m deep contains about 13,000 tonnes of soil, a small AEC can retain a large amount of nitrate and influence nitrogen (N) fertiliser requirement. If we assume that (1) AEC is uniform in a 20-100 cm layer and (2) only 50% of the AEC is occupied with nitrate ($NO_3$), then a soil with 0.1 $cmol_c kg^{-1}$ can hold 73 kg $NO_3$-N $ha^{-1}$. This is potentially enough to meet the N requirement of a 2-tonne $ha^{-1}$ cereal crop (~60 kg N $ha^{-1}$). The AEC of a soil can often exceed 0.1 $cmol_c kg^{-1}$ when measured in a 2 mM calcium chloride solution (Donn and Menzies, 2005, Wong and Wittwer, 2009). A database compiled for WA cropping soils suggests that on average, about 100 kg $NO_3$-N $ha^{-1}$ is held in the 1 m layer (Yvette Oliver, personal communication). Charge is likely implicated in profile nitrate retention. Capacity to retain nitrate is elastic since AEC increases with increasing soil solution concentration (Donn and Menzies, 2005). In addition to retaining nitrate, an even smaller AEC of 0.01 $cmol_c kg^{-1}$ can, according to chromatographic theory delay nitrate leaching by a factor of 1.5 compared with an AEC of 0 (Wong and Wittwer, 2009). Our aim was to assess the implication of positive charge on N fertiliser management by (1) determining how nitrate retention by charge varies with soil solution concentration and (2) measuring the effect of charge on nitrate and chloride leaching in order to assess the need to modify leaching routines used in crop models such as APSIM to simulate response to fertiliser N and to assess fertiliser requirement and fertiliser practices such as split application under variable soil, climate and management scenarios.

## Methods

Monolith lysimeters were collected for two Arenic yellow orthic Tenosols (deep yellow sands) and a Haplic Eutrophic Red Kandosol (red sandy earth). The lysimeters used 10 cm diameter PVC pipes placed on a tight fitting cutting ring to collect the soil monoliths with a “dig around and push down” technique to minimise soil disturbance. The soil monoliths were 60 cm (red sandy earth) to 70 cm (deep yellow sands) deep. Leaching was carried out in 6 replicates and under a tension of 10 kPa to ensure that soil water only flowed through pores < 30 uM diameter to avoid bypass flow. Resident soil nitrate was leached with 2 mM calcium chloride applied at the rate of 10 mm per day to ensure that the infiltration rates of the soils were not exceeded. The volume of calcium chloride applied (with peristaltic pumps) to each lysimeter was measured by weighing. Leachates were collected regularly under tension at the bottom of the lysimeters, measured for volume and analysed for nitrate and chloride using rapid potentiometric techniques with ion specific electrodes to determine delay in nitrate and chloride leaching.

Soils along the sides of the lysimeter collection trench in the field were sampled at 10 cm intervals to 30 cm depth and then at 20 cm intervals to 70 cm depth and analysed for pH, Al-saturation, nitrate, KCl-extractable sulphate, total N and organic C. Charge was measured at soil pH and varying equilibrating calcium chloride / nitrate solution concentration using the leaching method described by Wong *et al.* (1990).

## Results and Discussion

The Kandosol had the highest soil pH of 5.4 to 6.1, no aluminium saturation and the highest ECEC of the three soils examined (Table 1). It had no measurable AEC throughout the 0-70 cm layer and the lowest KCl-extractable sulphate and nitrate when sampled in late August 2011. The two Tenosols had pH values between 4.1 and 4.8 but aluminium saturation <41% that is unlikely to inhibit root growth and water and nitrate uptake down the soil profile in low ECEC soils. Abundant root growth was observed down to at least 70 cm at all three soil locations in the field during lysimeter collection. Tenosol 1 (Maddock) had lower AEC (in 2 mM $CaCl_2$) of 0.001, 0.033 and 0.049 $cmol_c kg^{-1}$ in the 0-20, 20-45 and 45-70 cm soil layer than Tenosol 2 (Alvaro). The corresponding AEC in Tenosol 2 were 0.033, 0.086 and 0.090 $cmol_c kg^{-1}$ respectively in the 0-20, 20-45 and 45-70 cm soil layer. Lower AEC in Tenosol 1 appears to result in less KCl extractable sulphate and nitrate when sampled in late August 2011 than measured in Tenosol 2.

**Table 1. Selected properties of soil sampled along the lysimeter trench in late August 2011**

| | pH in $CaCl_2$ | Al-sat (%) | Organic C (%) | Total N (%) | ECEC ($cmol_c kg^{-1}$) | $NO_3$-N (kg $ha^{-1}$) | KCl-S (mg $kg^{-1}$) |
|---|---|---|---|---|---|---|---|
| Kandosol depth (cm) | | | | | | | |
| 0-10 | 5.7 | 0 | 0.58 | 0.05 | 4.18 | 2.8 | 4.5 |
| 10-20 | 5.4 | 1 | 0.29 | 0.04 | 3.06 | 2.5 | 3.7 |
| 20-30 | 5.9 | 0 | 0.15 | 0.06 | 3.07 | 1.5 | 4.7 |
| 30-50 | 6.1 | 0 | 0.17 | 0.03 | 3.42 | 5.0 | 6.4 |
| 50-70 | 6.1 | 0 | 0.13 | 0.02 | 3.69 | 5.6 | 6.0 |
| Tenosol-1 | | | | | | | |
| Maddock depth (cm) | | | | | | | |
| 0-10 | 4.7 | 3 | 1.20 | 0.06 | 2.41 | 1.0 | 5.9 |
| 10-20 | 4.2 | 23 | 0.74 | 0.05 | 1.73 | 2.7 | 7.3 |
| 20-30 | 4.1 | 41 | 0.36 | 0.03 | 1.53 | 4.6 | 10.1 |
| 30-50 | 4.2 | 39 | 0.22 | 0.02 | 1.57 | 8.7 | 29.1 |
| 50-70 | 4.2 | 31 | 0.14 | 0.03 | 1.68 | 4.8 | 30.0 |
| Tenosol-2 | | | | | | | |
| Alvaro depth (cm) | | | | | | | |
| 0-10 | 4.8 | 5 | 0.91 | 0.06 | 2.57 | 9.1 | 4.8 |
| 10-20 | 4.2 | 37 | 0.60 | 0.06 | 1.80 | 6.7 | 10.0 |
| 20-30 | 4.2 | 40 | 0.31 | 0.03 | 1.71 | 6.2 | 17.3 |
| 30-50 | 4.2 | 35 | 0.13 | 0.04 | 1.93 | 16.0 | 33.1 |
| 50-70 | 4.1 | 36 | 0.12 | 0.03 | 1.98 | 6.7 | 42.1 |

AEC increased linearly with the square root of the equilibrium solution concentration, which is in line with theoretical expectation (Donn and Menzies, 2005). The example in Figure 1a shows that both the gradient and intercept of the relationship between charge and equilibrium solution concentration increase in soils with more charge. The maximum amount of nitrate that AEC in a 20 cm soil layer (30-50 cm) can retain can be up to 200 kg N $ha^{-1}$ (Figure 1b). In practice the amount retained will be less due to competition by other anions such as chloride and sulphate for the charged surfaces and lower soil solution concentration leading to less charge.

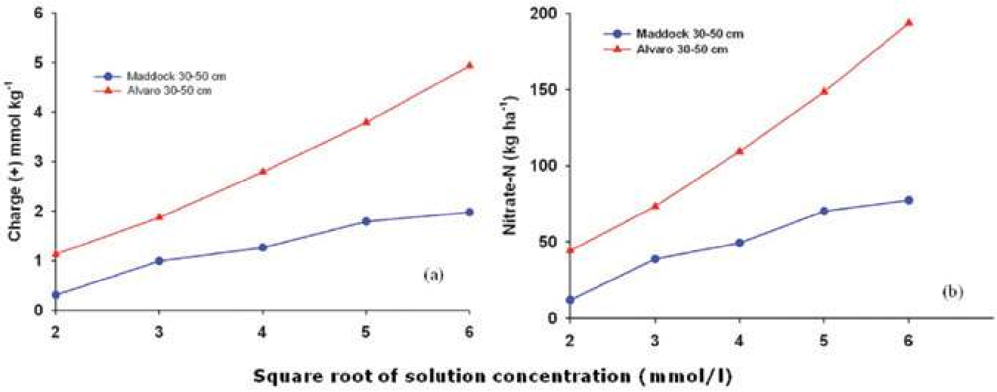


**Figure 1. Relationship between equilibrium solution concentration and (a) charge (anion exchange capacity, AEC) and (b) maximum amount of nitrate that can be retained by the AEC for a 20 cm layer (30-50 cm) of Tenosol 1 (Maddock) and Tenosol 2 (Alvaro).**

Chloride and nitrate leached fastest through the Kandosol. In the examples shown in Figure 2, the half maximum concentration of the chloride front in this soil occurred at approximately 1 pore volume (Figure 2a). This is expected as this soil had no measurable AEC and hence did not delay chloride movement. Nitrate displaced from this column was equivalent to 23 kg N $ha^{-1}$ and was partly derived from mineralisation causing its peak close to 1 pore volume (PV) to stretch to over 2 PV presumably due to continuous mineralisation on N during leaching.

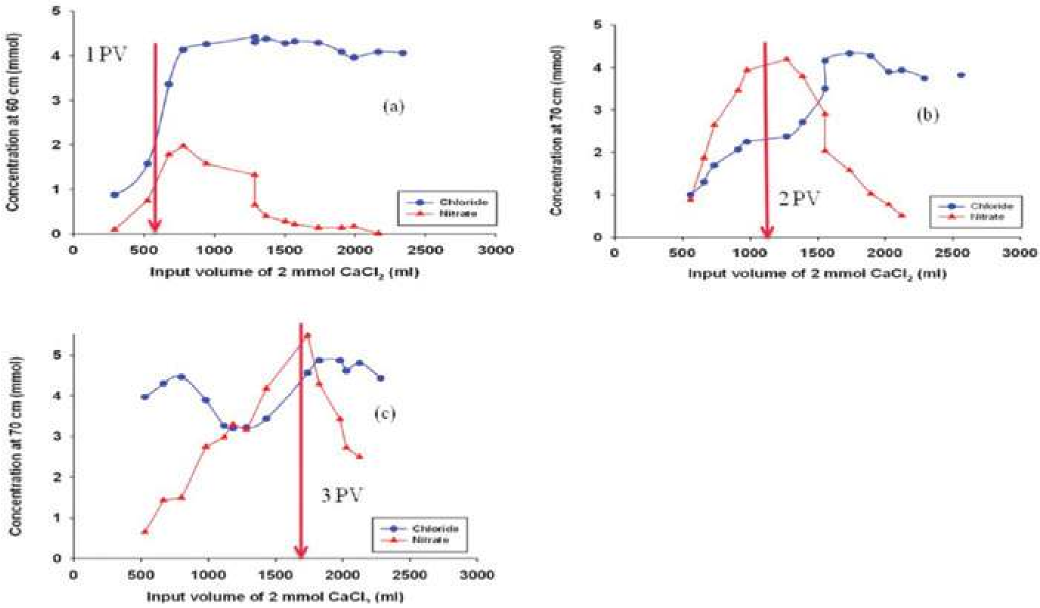


**Figure 2. Examples of nitrate and chloride leaching measured at the bottom of the Kandosol (a), Tenosol 1 (b) and Tenosol 2 (c). Locations of 1, 2 and 3 pore volumes (PV) are given for reference. The amounts of nitrate displaced were equivalent to 24 kg N $ha^{-1}$ from the Kandosol, 55 kg N $ha^{-1}$ from Tenosol 1 and 91 kg N $ha^{-1}$ from Tenosol 2.**

Nitrate and chloride leaching delay increased to about 2 PV and amount of nitrate displaced increased to 55 kg N ha$^{-1}$ in Tenosol 1 (Figure 2b). Delay in nitrate leaching increased further to about 3 PV and amount of nitrate displaced increased to 91 kg N ha$^{-1}$ in Tenosol 2. For this soil, delay in chloride leaching could not be determined due to high background soil chloride concentration similar to the input concentration (Figure 2c).

The amount of leaching delay due to AEC depends on the nitrate adsorption coefficient (□, nitrate adsorbed / nitrate in solution) of the soil layer (Wong and Wittwer, 2009). Although change in AEC with soil solution concentration is perceived to be a problem in modelling nitrate leaching in positively charged soils, in fact the problem is simpler since □ is almost constant resulting in constant gradients (□) shown in Figure 3. Using these gradients and the calculation of delay described by Wong and Wittwer, (2009), the 30-50cm layers of Tenosols 1 and 2 are expected to delay leaching by 1.7 and 2.7 PV respectively. These adsorption coefficients are being applied to the crop model APSIM to test its ability to simulate nitrate leaching in soils with AEC and determine the outcomes of scenarios to improve N management for more profit in these soils.

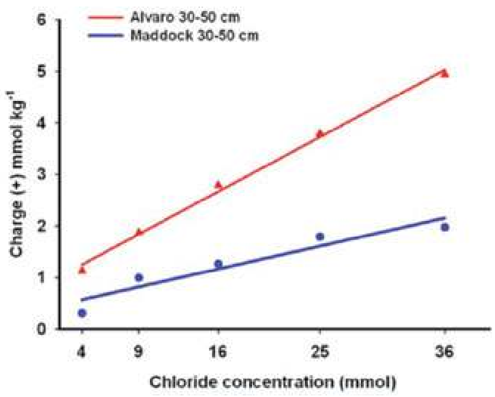


**Figure 3. Regression of charge (y) on equilibrium solution concentration (x) for Tenosol 1 (Maddock, y = 0.38 + 0.05x, $r^2$ = 0.90) and Tenosol 2 (Alvaro, y = 0.78 + 0.12x, $r^2$ = 0.995).**

## Conclusions

This work demonstrates that even small AEC values reported for WA soils can have large effects in retaining significant amounts of nitrate and presumably other anions such as sulphate in the soil profile. This would affect fertiliser requirement. Small AEC can also delay nitrate leaching by a factor of 2 to 3 compared with soils with no AEC. Delayed leaching will affect both the fertiliser requirement, as leaching losses would be smaller, and fertiliser management practice such as split fertiliser application, aimed at minimising leaching. In spite of likely widespread occurrence of AEC in Australian agricultural soils, its implication in fertiliser management is ignored. Crop models such as APSIM assume that nitrate leaching is not delayed. Simulated crop response to N fertilisers, fertiliser requirement and response to split applications are likely to be inaccurate for soils with AEC. There is therefore room to improve profits from better N management on these soils.


## Acknowledgements

We thank GRDC for co-funding this work as part of project UMU00035 and its “More Profits from Crop Nutrition” Program.



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# Nitrate removal in a denitrification wall after 14 years

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Denitrification walls are a low-cost approach for removing excess nitrate ($NO_3^-$) from shallow groundwater. They are created by mixing a soluble carbon source, generally woodchips or sawdust, with the soil to intercept shallow groundwater. Denitrification walls need to be maintenance-free for a number of years to remain cost effective, but little is known about the longevity of these walls. In this study, a denitrification wall constructed on a New Zealand dairy farm in 1996 was monitored to determine $NO_3^-$ removal by the wall 14 years after installation. After 14 years, the denitrification wall removed 92% of $NO_3^-$ input, which ranged from 2.2 to 3.7 mg N $L^{-1}$. The $NO_3^-$ input to the wall had decreased since first constructed, which was attributed to a change in upslope irrigation practices on the farm. Denitrifying enzyme activity (DEA) remained high after 14 years and the wall remained $NO_3^-$ limited. However, total C and microbial biomass C in the wall had decreased by approximately half, while available C remained relatively constant since year 2. By applying a first order decay curve, it was estimated that total C in the denitrification wall would not be depleted for 66 years, but it is unclear at what amount of total C that denitrification would become limited. This long-term study suggested that denitrification walls are cost effective solutions for remediating groundwater $NO_3^-$ pollution, as they can be effective for a number of years without any maintenance.

# The effects of previous management and urine patch interception on nitrate leaching from grazed dairy pastures

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A dairy farmlet trial comparing two contrasting farm systems was established in the Waikato (New Zealand) in winter 2012. Because this was a transition year, we hypothesised that effects on N leaching would be related to the previous management of the paddocks. Porous cups were installed (20 per ha) to measure N leaching.

With 440 mm drainage during winter 2012, the $NO_3$-N leaching losses were 115 ± 16 and 84 ± 11 kg N/ha for the two farmlets. There was a strong curvilinear relationship between $NO_3$-N leaching from individual paddocks and N fertiliser applied the previous year, which ranged from 110 to 540 kg N/ha/yr. To investigate further, we allocated a porous cup as intercepting a urine patch if $NO_3$-N > 100 mg N/l was measured in the drainage. From c. 260 porous cups per treatment, 33 or 24 patches were intercepted where we measured 115 or 84 kg N/ha leached, respectively. Over 75% of the variance in $NO_3$-N loads was explained by N rate applied and number of urine patches intercepted by the suction cups ($NO_3$-N loss (kg/ha) = 0.3 + 0.1759 * N applied (kg/ha) + 36.70 * No. urine patches per paddock; $R^2 = 0.785$***).

This highlights the significance of previous N management on $NO_3$-N losses, but also the impact of variability in urine patch interception by suction cup collectors on measured N leaching. This has implications when comparing management strategies that change the number of urine patches in a paddock, such as stocking rate or wintering off.

# Monday 3 December

## SOILS AND INFRASTRUCTURE DEVELOPMENTS

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# SoilMapp for mobile delivery of Australian soil data

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CSIRO soil, agricultural and IT scientists have collaborated to create a mobile app called "SoilMapp" for iPad tablets. This app exploits recent, substantial advances in mobile information delivery. SoilMapp provides a user-friendly interface that allows users of Australian soil information to interrogate and store location-specific soil data (i.e. mapping, data and reports) from national soil databases. The databases include soil mapping from the Australian Soil Resource Information System, soil site data from the CSIRO's National Soil Archive, and soil water characterisations from the APSoil database (e.g. for crop simulations). Data interrogation is achieved by geographic intersection of the databases via web services at the user's location using iPad's global positioning, or at pre-determined locations from maps on the tablet. Examples of the functionality of SoilMapp will be presented, as will ideas for future development.

# Historical soils information still relevant for Coal Seam Gas (CSG) projects soil assessment and management planning

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From the 1950s through to the 1990s the Queensland government and CSIRO described landscapes, soils and vegetation across the state at scales ranging from land systems studies at 1:500,000 or more to land suitability assessment at 1:100,000 or finer. These land resource data were supported by mapping by geological survey at 1:250,000 and 1:100,000. These data were used initially mainly to inform agricultural development but in more recent years the emphasis has changed to land management and land use planning.

These data have proved to be remarkably resilient and soils and landscape data collected over 50 years ago are being reinterpreted to address current issues. The CSG industry in Queensland has tenements over 3.7m ha collectively and construction activities could cover around 210,000 ha. Many of the soils in the region have properties such as being strongly sodic, strongly alkaline or acid, saline, highly dispersive and low fertility that make them vulnerable to degradation through soil erosion and loss of productivity. Understanding the distribution of vulnerable soils and their relationships within the landscape are essential to developing soil management plans that deliver desirable environmental outcomes.

The broad scale of the historical information means that it cannot be used directly to inform management measures about land at the scale required for intensive development. The descriptions, however, provide a valuable framework within which to base a systematic approach to land management. The site data including soil profile and landscape descriptions and accompanying analytical data allow assessments to be made about soil responses to disturbance and rehabilitation. These data are particularly valuable given the currently high costs of procuring comparable data.

# Soil amendment with biosolids exposed to silver nanoparticles: Is silver bioavailable?

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Rapid expansion of the nanotechnology industry has occurred over the previous decade. Manufactured nanomaterials (MNM) are commonly defined as materials having at least one dimension between 1-100 nm (BSI 2007a). Compared to the bulk material, MNM exhibit unique and enhanced electronic, optical and chemical properties. Such materials are used by numerous industries and are currently incorporated into products such as medical supplies, personal care products, consumer goods and microelectronics (Farré *et al*. 2011). Manufactured nanomaterials encompass a variety of classes but the most common category is nanoparticles (NPs).

Silver (Ag) NPs are the most widely used NPs in consumer products. However, the anti-bacterial properties that render Ag NPs desirable may also lead to increased environmental risks. Studies on the behaviour of Ag NPs in the environment are particularly limited for terrestrial systems. In addition, the bioavailability of Ag NPs may be overestimated because it is usually studied in hydroponic systems using pristine Ag NPs that are known to be retained by soil solids (Cornelis *et al*. 2012). Silver NPs may enter wastewater streams through the washing and use of nano-containing products (Benn and Westerhoff 2008; Geranio *et al*. 2009; Kaegi et al. 2010) and partition predominantly to the biosolids during wastewater treatment (Kaegi *et al*. 2011). Soil exposure to Ag NPs is thus a realistic scenario considering that wastewater treatment biosolids are often applied to soil as an agricultural amendment.

During a simulated wastewater treatment process we demonstrated that the majority (> 90%) of Ag NPs were captured by biosolids and transformed to Ag2S. Based on this scenario, we investigated the availability of Ag NPs to lettuce (L. Sativa). Biosolids containing transformed Ag NPs were produced and incorporated into a soil with weak Ag+ sorption. This paper will report the Ag concentrations in the plant parts of lettuce grown for 6 weeks in the biosolids-amended soil. Preliminary analysis indicates that the plant available Ag may be affected by the presence of macronutrient fertilisers used in commercial crop production.

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# Effectiveness of andesitic tephra soil filters to remove dissolved reactive phosphorus from wastewaters

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The discharge of wastewaters from town sewage treatment plants (STP) is a major cause of elevated dissolved reactive phosphorus (DRP) concentrations at various river sites within the Manawatu-Wanganui region of New Zealand, particularly during low river flows. Wastewater treatment options for DRP are either cost-prohibitive or largely ineffective, particularly for smaller towns (Keplinger et al., 2004). Therefore, there is a need for a new, low cost method for DRP removal.

Soils formed from andesitic tephra with relatively high P absorbing capacities are abundant in the central North Island; therefore, they have potential as relatively low cost filter substrates for DRP removal. In addition, once the DRP adsorbing capacity of the soil is exhausted, it has potential for re-use as a soil amendment.

The P absorption capacities of 7 different tephra subsoils from the Taranaki and Ruapehu regions were assessed in a laboratory column study. The quantities of DRP removed from solution by the subsoils ranged from 1.9-5.5 mg P/g soil at an average removal efficiency of 95%, or 2.5-8.7 mg P/g soil at an average removal efficiency of 75%. These results demonstrated that these materials show promise as filter substrates to remove P from wastewaters. Tephra soil filters are being evaluated at the pilot plant scale to further assess the feasibility of these treatment systems.

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# Soil assessment and management for Coal Seam Gas (CSG) pipelines in southern Queensland

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The recent expansion in CSG in south-west central Queensland has seen four companies separately developing tenements totalling more than 3.7m ha (Queensland Curtis LNG 407.8m ha, Australia Pacific LNG 571.2m ha, Arrow Energy LNG 848m ha and Gladstone LNG 1,884m ha) (WorleyParsons, 2011). WorleyParsons (2011) estimated that within these tenements there would be construction impacts over 139,072 ha with final operation impacts being 67,977 ha for gas wells, borrow pits, gas and water gathering pipelines, trunklines and major collection pipelines, field compression stations and central processing plants, water treatment facilities, access roads / tracks, brine ponds / basins, environmental dams and construction camps.

CSG is delivered to processing facilities in the gas fields then to the liquefied natural gas (LNG) facilities on Curtis Island in steel pipes ranging from around 450 mm to over a metre in diameter. These are buried at depths of 750-900 mm below the surface. These larger diameter pipes are laid in trenches excavated using track trenchers at widths of approximately 2 m. Up to 5 $m^3$ of soil material per linear metre may be excavated, exposed on the surface for up to 3 months and then reinstated around the pipes. The pipelines traverse a wide range of soils and many have physical and chemical properties that make them vulnerable to degradation, such as soil erosion, or are not suitable to support rapid revegetation.

Delivering salient soil management advice is critical if desirable environmental outcomes are to be achieved. We developed a system of describing unique pipeline sections (UPSs). Each UPS has a unique identification and includes soil and land resource information and management recommendations.

# Monday 3 December

## SOILS AND INFRASTRUCTURE DEVELOPMENTS

*Presented Posters*

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# Application of shallow geophysics within an artificial water catchment in the lignite mining area of NE Germany

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Ground penetrating radar (GPR) and electric resistivity tomography (ERT) have been applied to study the shallow subsurface of an artificial water catchment. The catchment is part of the  Collaborative Research Center of the German Research Foundation (DFG SFB-TRR 38) 'Structures and processes of the initial ecosystem development phase in an artificial water catchment' within a remediated site of the mine. The so called 'Chicken Creek' catchment is built up by mechanically deposited sediments reconstructing an initially emerged landscape. In order to monitor its' geomorphological as well as eco-functional evolution with time it is essential to incorporate comprehensive geophysical applications dealing with soils and sedimentology in the shallow subsurface.

GPR was applied with adequate resolution to detect the primary deposition structures within the technogenically bedded sediments. By means of this evaluation subsurface structures can be interpreted with regards to their relevance in controlling soil water movement, slope hydrology, redox interactions, root distribution etc. Furthermore, alterations of these primary structures resulting from bio- and pedoturbation processes can be monitored.

ERT was applied to monitor changes in water contents at a 2D scale across the catchment a at a 1-3 hour resolution. Infiltration of water after singe rain storm events could be traced and displayed on a 2D-view using time inversion lapse techniques. Preferential flow paths have been identified. Frost and thaw processes could be portrayed using ERT-techniques.

Methods of geophysical prospection are minimum invasive and therefore they are ideal techniques for monitoring subsurface processes.

# Monday 3 December

## PEDOLOGY, SOIL STRATIGRAPHY AND QUATERNARY LANDSCAPES

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# Airborne electromagnetics predict soil drainage and aid irrigation design: Weaber Plain, East Kimberley, Western Australia

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Airborne electromagnetic (AEM) data assist soil survey, and have important applications in irrigation design and predictive soil mapping. As part of environmental planning for expansion of the Ord River Irrigation Area, a detailed soil assessment was carried out in the eastern Kimberley, Western Australia. This assessment aimed to distinguish areas of better drained Cununurra clay (black and brown Vertosols) from poorly drained Aquitaine clay (grey Vertosols) on a relict alluvial plain to guide irrigation planning. Baseline data on soil characteristics, soil salinity hazard and drainage were collected prior to land use conversion. One hundred and sixty backhoe pits, excavated into the C horizon (2-4m), were positioned across the main soil, landform and vegetation units determined from previous soil surveys and recent AEM data. Fifty saturated hydraulic conductivity tests were also conducted on a range of substrates below the cracking clay horizons.

The AEM data was positively correlated with field EC and substrate clay content, improving prediction of soil drainage and suitability for irrigation. Cununurra clay had relatively low inherent EC and friable self-mulching topsoils on coarse to medium textured substrates (sandy gravel to fine sandy clay loam). In contrast, Aquitaine clay had relatively high subsoil EC on medium to heavy clay substrates.

The combination of traditional soil description and mapping with modelling using environmental parameters improved the reliability of the products and interpretations. As the models have been extended to similar potential areas, there is an opportunity for validation of these models and developing a cost effective assessment process.

# Electromagnetic conductivity image (EMCI) across a prior stream channel using DUALEM-421 and EM34 data and inversion software ($\mathrm{EM4}_{\mathrm{soil}}$)

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The alluvial plains of the Murray-Darling Basin (MDB) were formed by a system of prior stream channels which are susceptible to significantly larger levels of deep drainage as compared to the clay plains. In order to characterise the connectivity of the prior stream channels with underlying migrational channel deposits (i.e. palaeochannels) direct current resistivity or airborne EM systems are potential options. Whilst each has advantages major drawbacks include the time consuming nature of the former and the expense of the latter. One of the most popular geophysical methods currently being used to provide information about the near-surface is electromagnetic (EM) induction. One reason it has not been used for the same purpose is because of the lack of software to invert the measured apparent soil electrical conductivity ($\sigma_a$). In this paper, we describe how a next-generation DUALEM-421 and EM34 can be used in conjunction with joint-inversion algorithm software (EM4Soil) to generate 2-d models of σ (i.e. EMCI - electromagnetic conductivity image) across an irrigated cotton growing field located on Quaternary Alluvial clay plain northwest of Moree, New South Wales (NSW-Australia). The general patterns of the EMCI are shown to compare favourably with existing pedological, physiographic and stratigraphic knowledge. On the clay alluvial plain the main features associated with the depth of clay alluvium as well as delineating the juxtaposed sandier alluvial sediments, which characterize prior stream channels.  In addition, we resolve the location of buried migrational channel deposits (i.e. palaeochannel). However, modelled σ is found to be more strongly correlated with measured $EC_{1:5}$ ($r^2 = 0.61$).

# The role of regolith and soil development with respect to assessing heavy metal contamination in urban soils with particular reference to iron

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**Abstract**

Environmental assessors investigating brown and green development areas in inner and peripheral urban land in Australia routinely collect soil samples at prescribed depths in the soil profile, e.g. 0.1 – 0.5 – 1.0 m, etc. The protocols do not require the sampling depths to take notice of the natural horizonation of a soil profile and hence are blind to geomorphological and weathering history of the site. Australia has largely been spared the wholesale removal and re-deposition of soil and rock materials by Pleistocene glaciers or erosion following recent uplift. There are landscapes that probably date back to the Tertiary, 5-10 M years ago, without much change, as well as the Quaternary landscapes that are 4 M years old. The vertical and lateral movement of heavy metals, including iron, nearly always explains the occurrence of elevated concentrations of As, Ba, Cr, Co, Cu, Ni, Pb, V, and Zn in certain strata of the soil profile. The localised accumulation of these metals is normally controlled by changing redox potentials, which in turn are affected by translocation of clay and differences in soil hydraulic conductivity between A, B and C soil horizons. In other cases, the soil profile has operated like a chromatogram over many thousands of years.

**Key Words**

Heavy metals, weathering, iron.

**Introduction**

Environmental site assessments of both green and brown sites of future residential land in the Greater Melbourne Metropolitan Region (GMMR) frequently encounter elevated concentrations of heavy metals and metalloids in the soil, especially in the subsoil. These are difficult or impossible to attribute to man-made contamination. In such cases, there are costly delays because the Regulator will not sign off on the safety of the land and development stands still.

Australia is an old continent that has a large variety of lithologies which have undergone prolonged geochemical weathering, resulting in the relocation of many elements in the weathering mantle, the regolith. They have normally finished up in zones of the regolith where they have achieved maximum stability and minimum mobility and often, also, are in equilibrium with the hydrological regime of the soil profile and remainder of the regolith. The perspective of heavy metal contamination in Europe, the "old World" but a geomorphologically young land mass, is largely on man-made effects brought about in the last two thousand years, and this has also affected the environmental mindset in Australia, a "New World" but a geomorphologically ancient land mass. Environmental site assessors in Australia should know more about deep weathering processes.

The lack of a widespread understanding of how the geochemistry in the regolith affects the solubility and mobility of these metals also leads to insufficient appreciation of the "aging" factor of man-made metal contamination in the soil. Natural soil chemical processes can reduce the solubility and mobility of a number of these metals over time, often by forming inert compounds with the natural ferruginous matter in the soil (Trivedi and Axe 2007). Apparently arsenic at 160 mg/kg and chromium at 150 mg/kg in ferruginous concretions in constant contact with potable water on the shore of Melbourne's Sugarloaf Reservoir does not affect drinking water quality. Iron plays a major role in locking up metals and metalloids.

Because of its huge spread, the greater Melbourne Metropolitan region extends 70 km from its western extremity with 450 mm annual rainfall to its eastern extremity with 900 mm or more. A similar rainfall gradient may well have operated during much of the Quaternary, although during the Glacial phases the climate is thought to have been much drier. Likewise, the region spreads out over several very different "hard rock" lithologies from Quaternary basalt to Silurian sedimentary rocks and Devonian granites. However, there are landscapes in the region that probably date back to the Tertiary, 5-10 M years ago, without much change, as well as the basalt landscapes that are up to 4 M years old. The geochemical inheritance of this long period of weathering and soil formation on such different parent materials must be understood, or at least appreciated, to interpret the results of soil chemical analyses for environmental assessments.

The present climate of the GMMR has a slight Mediterranean influence in that summers are dry but significant rainfall still takes place and winters are cold and have the highest monthly rainfalls. There is a strong rainfall gradient from the west to the east: annual rainfall around the western edge of the GMMR is about 450 mm, but this increases to about 1000 mm at the eastern end. Annual evaporation is around 1600 mm in the west and 1100 mm in the east. This has affected the weathering and soil formation greatly. Rainfall in the west is far too low to achieve removal by leaching of the more soluble products of weathering including salts brought in by the main rain-bearing winds. Subsoils have considerable salinity, as measured by electric conductivity (EC), high sodicity and high pH. In the east, all soils have been leached and are fundamentally free of soluble salts and have become acidic throughout.

Research into palaeo-climatic conditions strongly suggests that mean annual temperatures during the Miocene may have been about 5$^{o}$ C higher than is the case at present in Melbourne (15$^{o}$ C). Annual precipitation was also considerably higher at 1500 mm in an area that today receives 800 mm and there was a pronounced wet season. During the Pliocene, it is thought conditions became warmer and rainfall more uniform but still remained somewhat seasonally humid. In south eastern Australia, the mean annual temperature was around 20$^{o}$ C and annual rainfall about 1000 mm.

The Pleistocene was affected by the 17 or 18 glacial cycles of the Northern Hemisphere, which also affected Australia by corresponding cycles of aridity and humidity. During the last glacial epoch only a very small area in the Snowy Mountains, New South Wales, was glaciated and small areas of the Victorian section of the Great Dividing Range were affected by peri-glacial conditions. The GMMR suffered cold but dry conditions. The Holocene Climatic Optimum (HCO) dominated the last ten thousand years between 9000- 6000 BP with warmer and wetter conditions than today. The climate subsequently, towards 2000 BP, became drier and cooler.

**Lithology and Geomorphology**

The GMMR area has been relatively stable since the beginning of the Tertiary, but has undergone climatic changes from tropical to cold during glacial epochs. The Silurian sedimentary rocks are very thick, up to 10 km, and underlie most of the GMMR. Within much of the metropolitan area, the Silurian sediments have been eroded to form an undulating terrain with isolated hills that have rounded tops and smooth slopes (Presland 2009). Further east and north, towards the Great Dividing Range, the land is strongly undulating, hilly, with steeper slopes, rising to 500-600 m ASL (Presland 2009). The granite and granodiorite areas in the eastern parts form high hills with broad summits. These summits mostly still have a cover of regolith that weathered during a warm and wet climate and has relatively deep red soils, but the flanks have been stripped of this material and have soils that may be considered to be contemporaneous. The summits of the Silurian Hills have been considered a dissected plateau-like palaeosurface, called the Nillumbik Surface, which slopes downwards in a southerly direction and continues below sea level in Port Phillip Bay. The depth of soil on the preserved and dissected parts of the Nillumbik Surface varies enormously. For example, a remnant plateau in the eastern suburb of Nunawading has 2 m deep soils, whereas a few km away on the dissected slopes soils typically are no deeper than 0.5-0.7 m.

The Pliocene basalts form broad-crested hills on a deep weathering mantle of soft regolith, often prone to land slips. The typical soil on these older basalts is a deep, friable, strongly structured red gradational profile, silt loam texture at the surface and light, kaolinitic clay at depth. Towards the drier north and west, where erosion has been able to remove more soil and regolith, some of these basalt outcrops have contemporaneous black expansive clays with lime-rich subsoils.

The lower parts of the GMMR have been covered with Pliocene to Miocene sediments which are partly unconsolidated; otherwise, they are soft sandstones containing much ferruginous material. These Tertiary sediments have been exposed to later erosion and continued weathering.

The recent basalts flowed out over the southern and south eastern areas of the GMMR less than 4.5 M years ago, the most recent flow being about 820,000 years ago. The depth of the regolith and the degree of rock outcrop varies with the age of these flows. The oldest soils are rock-free, deep, red-brown heavy clays with lime at depth; the youngest are shallow, dark grey-brown or black heavy clays with much rock outcrop.


**Figure 1. Geomorphic surfaces of the GMMR (Joyce and Webb 2003).**

Kinglake surface

Nillumbik surface S of Kinglake Plateau

Coastal plains on Tertiary sedimentary rocks

Quaternary basalt plains, varying soil formation ages

This paper will give some examples of misinterpreted contamination scares in relation to As, Ba, Cr and V that sometimes caused large financial budget overruns at developments in Melbourne.

### Arsenic and Chromium Enrichment in ferruginous Material

During the excavations for the new Melbourne Museum the environmental site assessors found numerous 'soil' samples with elevated arsenic in the highly weathered Silurian strata underlying the area. The result was that the Victorian Environmental Protection Authority decreed that all soil had to be stockpiled on site and re-sampled for chemical analysis to allow disposal as either clean fill or as contaminated soil. The workforce at the site demanded increased pay to compensate for the increased risk to health. In site investigation, it was discovered that the strata had undergone varying transformations-–the majority having become kaolinised and creamy white, where other strata had accumulated a great deal of ferruginous matter (Figures 1 and 2).

When these were sampled separately and analysed in the laboratory for Total As and Fe, citrate/dithionite extractable Fe and then the As concentration in that extract, it could be proved:

(1) that the elevated As occurs always in the ferruginised strata,

(2) none of the kaolinised strata had significant As, and

(3) when the Fe was solubilised, the As also came out, but

(4) not all high Fe samples had also high As.

Arsenic and chromium have also become enriched along with iron in nodules forming a beach gravel along the shores of Melbourne's potable water reservoir, the Sugar Loaf Reservoir. In other ferruginous nodules found in soils around Broadford chromium levels are highly related to iron levels ($r^2 = 0.956$). The iron coatings forming the red colour of red-brown earth subsoils around the Military Barracks area at Barranduda in Northern Victoria have become enriched with chromium also. In all these cases the enrichment has increased the metal concentrations to levels higher than the EPA criteria for clean soil.

### Barium Enrichment in Basalt-derived Clay Subsoil

A very large area on Quaternary basalts on the western periphery of Melbourne was being assessed for contamination as it was being prepared for residential development. The area has a dry climate with low rainfall and high evaporation. The soils are also duplex profiles with expansive heavy clay subsoils with an illuvial calcium carbonate accumulation at about 60 cm depth.

Soil samples were taken at the surface, 0.5m, 1.0m, 1.5m and 2.0 m depth unless the sampling tube encountered hard rock. A number of sites tested showed exceedance levels for barium causing the project to be stalled. When the results were analysed it was found that all the exceedance values were at 0.5 and/ or 1.0 m depth. To evaluate the risk to humans and the environment it was essential to obtain information on the solubility and mobility of the barium. All the soils in the site are well-drained, aerobic and have supported a grassland with scattered trees. Clearly, there would be sulphur in the ecosystem and hence in the soil and the most common form of sulphur would be sulphate. Barium is present in basalt with an average concentration of 330 mg/kg (Reimann and de Caritat 1998) and therefore will be released as the rock weathers to form soil.

Two of the high Ba sites were revisited and pits were excavated in order to sample the profile every 10 cm down to the base of the pit. All samples were analysed for total Ba, leachable Ba, electrical conductivity (EC), pH and water extractable sulphate. The outcome of this investigation showed that the Ba was also distributed vertically in the profile, forming a "bulge" in the deeper subsoil. Of more significance for assessing risk was the result that the concentration of water soluble $SO_4^{2-}$ on a molar basis exceeded the Ba concentration by 3 or 4 orders of magnitude. Hence, the barium existed as its sulphate and could be considered to be insoluble and of no consequence for the environment. In the Australian climate and over very long time spans the soil profiles have acted as chromatograms with respect to the various solutes as shown in Figure 2.

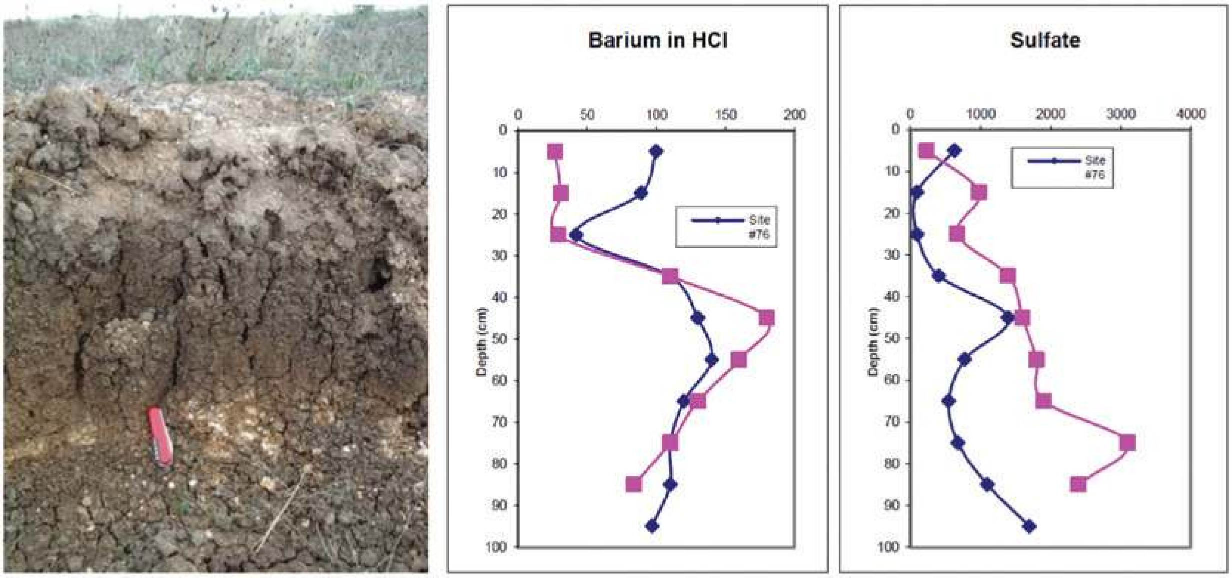


**Figure 2: Carbonate, barium and sulphate chromatograms in heavy clay soil.**

In many cases in which I have been asked to interpret heavy metal contamination in soils it could be shown that there was no true contamination and that the metals were indigenous to the regolith, explainable from a geochemical perspective and wholly inert. But they did cause unnecessary delays and considerable cost increases to the developer or the government.

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# Soil morphological and chemical characteristics of key research and demonstration sites of the Central Province, Papua New Guinea

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## Abstract

Knowledge of soil morphology and chemical fertility at key agricultural research and demonstrations sites is fundamental to the development and extension of agricultural production systems. Developing countries are often challenged by incomplete and/or sparse spatial data that does not capture the variability in soil type, fertility and other key limitations to productive and sustainable use. While it is not possible to overcome all land use limitations in a short term project, a strategic approach that targets areas that have potential for intensive agricultural development can be used to best focus limited resources. This approach has been used in Central Province of Papua New Guinea. This province consists of lowland and elevated areas with vastly differing soil types and land use limitations. Here we report on profile characteristics and soil fertility of potentially productive soil and site combinations identified at research stations and from organised farming cooperatives in the district. The data show some of the wide diversity in soil physical and chemical characteristics in the province. The implications of these soil data for agricultural development and productivity are discussed.

## Introduction

Understanding of the quality of soil resources is fundamental to development of sustainable agricultural production systems. In many developing countries, soil resources are described at an aggregated level, as in Bleeker (1983) and Hanson *et al*. (2001) for Papua New Guinea (PNG), but this is usually inadequate for decision making on cropping practices for long term sustainability. In this study, existing information on soils and land capability in Central Province PNG has been further supported by profile examinations of soils identified as having potential for sustainable agricultural development. Soil types in PNG are highly variable due to the landscape being affected by active geological processes, significant climatic gradients and variable topographic features in a high relief terrain. Land capability and land use potential is also highly variable (Hanson *et al.* 2001). This paper reports on profile characteristics and chemical fertility of soils identified as having potential for intensified vegetable production in coastal and elevated regions of Central Province, and identifies practices needed for sustainable production.

## Materials and Methods

Soils with potential for agricultural development were identified in the coastal lowlands (National Agricultural Research Institute [NARI] at Laloki, Pacific Adventist University at Koiari Park and farmer cooperatives near Kwikila in Rigo district) and in the more elevated areas of the Sogeri Plateau and Tapini. Soil pits were dug at all sites to at least 1 m depth (or by soil auger) and soil profile morphological characteristics were described and soil horizons sampled. Standard chemical analyses were undertaken by NARI Chemistry Laboratory using analytical procedures outlined in Rayment and Higginson (1992).

## Results

Summaries of soil profile descriptions are provided in Table 1 and these data show considerable variation in physical characteristics of profiles, with impeded drainage or high water-tables indicated by the presence of mottling (Mo) and manganese nodules (Mn) at differing depths in several soils. Texture class is dominated by clays (LC – MC) and clay loams (CL) which are commonly silty (Z). Soil structure is moderate to strong in most upper profiles. The presence of carbonate in lower horizons of the Koiari Park and Rigo soils indicates high base status and low leaching regimes.

**Table 1. Profile characteristics for selected soils in the Central Province of PNG**

| Laloki | Koiari Park | Rigo 1 | Rigo 2 | Sogeri | Tapini |
|---|---|---|---|---|---|
| Alluvial Dermosol | Colluvial Vertosol | Colluvial Vertosol | Alluvial Vertosol | Colluvial Ferrosol | Colluvial Dermosol |
| ~40 m ASL | ~50 m ASL | ~80 m ASL | ~85 m ASL | ~400 m ASL | ~900 m ASL |
| Imperfect Drainage | Imperfect Drainage | Well drained | Moderately well Drained | Imperfectly drained | Moderately well drained |
| **A11** 0 – 5 cm 10YR 4/2, ZCL M-PO | **A11** 0 – 10 cm 10YR 2/1, MC, S-PO, F, C, G | **A11** 0 – 5 cm 10YR 2/1, ZLC, S-PO, F, A | **A11** 0 – cm 10YR 2/1, ZLC, S-PO, W | **A1** 0 – 10 cm 7.5YR 3/3, LC, S-PO, F | **A11** 0 – 15 cm 7YR 2.5/1, ZCL, M-PO, W, MG |
| **A12** 5 – 12 cm 10YR 4/2, ZLC, M-PO | **A12** 10 – 25 cm 10YR 2/1,MC, S-PO, VF, C, F-MG, G | **A12** 5 – 25 cm 10YR 2/1, ZLMC, S-PO, F, D | **A12** 5 – 20 cm 10YR 2/1, ZLMC, S-PO, W | **A3** 10 – 25 cm 7.5YR 4/3, FSLC, S-PO, F, Mn, G | **A12** 15 – 25 cm 7.5YR 3/1, ZCL, M-PO, W, MG |
| **B11** 12 – 30 cm 10YR 4/3, ZLC, M-PR+M, | **B21g** 25 – 40 cm 10YR 4/1, LMC, M-AB, F, Mo, O, G | **A13** 25 – 45 cm 10YR 2/1, ZLMC, M-PO, VF, G | **B1** 20 – 35 cm 10YR 3/1, ZMC, MS, F | **B1** 25 – 35 cm 7.5YR 4/4, SCL, M-PO, F, Mn, | AB. 25 – 35 cm 5YR 3/2, ZLC, M-AB, W, CG |
| **B12** 30 – 50 cm 10YR 4/3, ZLC, S-PO+GR, Mo, G | **B22g** 40-60cm, 2.5Y 5/2, LMC, W-AB, F, Mo, S | **B21** 45 – 65 cm 10YR 3/2, LMC, M-PO, VF, $CO_3$, D | **B21** 35 – 50 cm 10YR 3/2, ZMC, MS, VF | **B21** 35 – 50 cm SCL, MS+M-AB, F, Mo, Mn, D | **B21** 35 – 45 cm 5YR 4/4, ZLC M-AB, W, MG |
| **B21** 50 – 75 cm 10YR 4/2, ZMC, S-PO+AB, Mo, G | **C1g** 60 – 80 cm 2.5 Y 5/2 , MC, MS, F, $CO_3$ | **B22** 65 – 90 cm 2.5 Y 3/1, MC, MS, VF, $CO_3$, FG, D | **B22** 50 – 65 cm 2.5Y 3/2, ZMC, MS, VF | **B22** 50 – 85 cm 7.5YR 5/2, SLC, MS+W-AB, F, Mo, Mn, D | **B22** 45 – 65 cm 10YR 5/6, ZL,MC, M-AB, W, MG |
| **B22** 75 – 95 cm 10YR 4/2, ZLC, Mo, | **C2g** 80 – 95 cm 2.5Y6/2, MC, MS, F, Mo $CO_3$ | **B23** 90 – 110 cm 2.5Y 3/1, MC, MS, F, $CO_3$, FG | **B23** 65 – 80 cm 2.5Y 3/1, ZMC, MS, VF | **B3** 85 – 120 cm 7.5YR 5/6, SLC, MS, F, Mo, Mn | **B23** 65 – 80 cm 10YR 5/6, ZMC, M-AB, W, MG |
| **B31** 95 – 105 cm ZLC | **C3g** 95 – 115cm 5Y7/8, MC, MS, F, Mo, $CO_3$ | | **BC** 80 – 95 cm ZMC, MS, VF | | **B24** 80 – 100 cm 10YR 5/6, ZMC, W-AB, MG |
| **B32** 105 – 110 cm+ ZLC | | | | | **B25** 100 – 110 cm 10YR 5/6, ZMC W-AB, MG, Mo |

Soil order from Isbell (2002)
Texture: CL = clay loam, LC = light clay, MC = medium clay note Z in front = silty, S = sandy, FS = fine sandy, L= light
Structure: PO = polyhedral, PR = prismatic, AB = angular blocky, M = massive
Gravel: FG = fine, MG = medium, CG = coarse,
Moist Strength: W = weak, F = firm, VF = very firm
Mottles, Nodules and Cutans: Mo = mottles, CO3 = carbonate nodules, O = Organic cutans, Mn = manganese nodules
Boundaries: Clear unless shown as A = abrupt, G = gradational or D = diffuse boundary.

Figure 1 provides selected analytical data for the profiles described and they show considerable variation in soil chemical characteristics. The three Vertosols (Mafic soils) at Koiari Park, Rigo1 and Rigo 2 are high in exchangeable Mg, CEC and C/N, but are generally low in P. However most profiles have high exchangeable Ca and moderate to high exchangeable Mg levels. Overall total N values are low (mostly <0.2%). All the soils have relatively high pH, except at Sogeri where the profile appears more leached. Available P as measured by the Olsen test ranges from low in the sesquioxide rich soil at Sogeri, to relatively high at the government research stations of Laloki and Tapini.

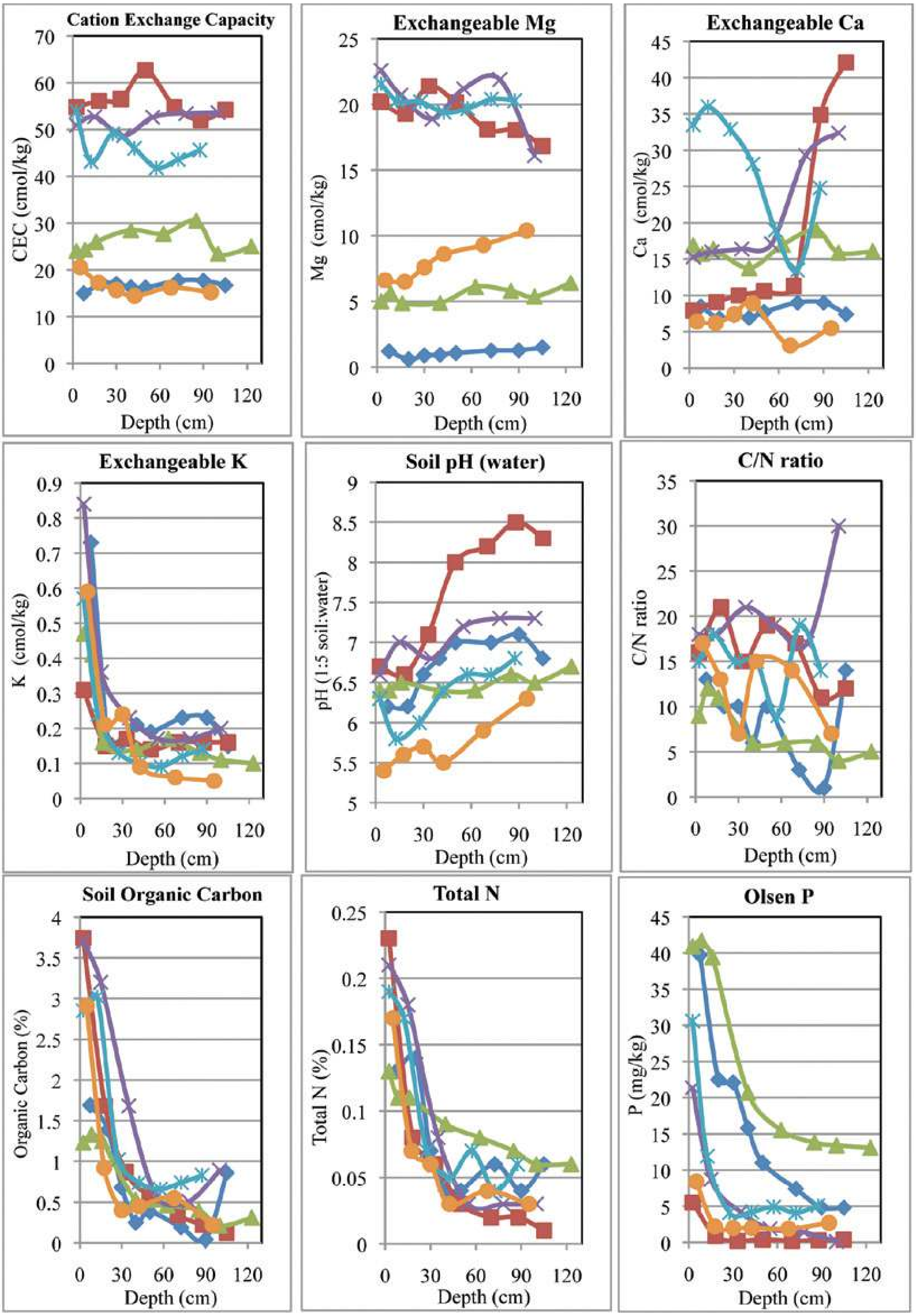


**Figure 1. Cation exchange capacity (CEC), Exchangeable Ca, Mg, K, Soil pH, C/N ratio, Soil Organic Carbon, Total N and Olsen P. Sogeri (orange circles), Tapini (dark blue – diamonds), Laloki (green triangles), Koiari Park (red squares), Rigo 1 (purple x) and Rigo 2 (light blue *).**

Total N and organic C concentrations are variable with values at Koiari Park and Rigo 1 and 2 (all Vertosols) being higher than at other sites. Both measures decline rapidly in the B and C horizons. C/N ratios are highly variable (ca 5 – 20), with some quite high (>16, above which immobilisation of N is likely). Exchangeable K values are low at Koiari Park and moderate to high at all other sites. Extractable

P and exchangeable K, organic C and total N were concentrated in the upper profile, though significant concentrations of P were present in the upper layers of the B horizon at Laloki and Tapini. In the three Vertosols (Koiari Park, Rigo 1 and Rigo 2) exchangeable Ca increased markedly in the lower profile associated with the presence of $CaCO_3$ segregations.

## Discussion

Profile physical conditions were variable, however, the overriding interpretation is that careful management will be needed to maintain favourable conditions at all sites, though some are more susceptible to degradation than others. Specifically, the mottling and manganese we observed suggests periodic water logging which may inhibit production during the wet season, while irrigation would be needed during the dry season. Weak moist soil strength, clayey textures and massive structure suggests that care will be needed in the use of machinery, where available, to minimise the risk of compaction, while high dry soil strength is likely to make soil cultivation when dry impractical. Thus, there is likely to be a narrow 'window' when soil conditions are suitable for use of machinery without degrading the soil.

The variation in chemical soil fertility means that assessment of soil fertility and thus the need for fertiliser inputs or other soil and crop management strategies will be needed on a site-specific basis.

To assist soil N and P economy at least, strategies to retain soil organic carbon will be essential. Management of the unfavourable C/ N ratios will require additional inputs of N from either organic (imported organic matter, compost, manures), biological (legumes) or fertiliser sources. Although regular burning of fields may in part be the cause of these high C/N ratios. How this is best addressed will depend on the local availability of suitable materials and crop rotations that are acceptable and profitable. Low to medium concentrations of extractable K can only be addressed by inputs of K in imported organic matter, ash or chemical fertilisers. An unfavourable Ca/Mg ratio (<1) has already been expressed as blossom end rot (Ca deficiency) of tomatoes at Koiari Park.

Land supply in Central Province for agricultural development should include analyses of soils to identify suitable and economically viable sites; the most suitable are likely to be in alluvial areas and toe slopes in mafic terrains with easy access to water for irrigation in the dry season.

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# A formative role of eucalypts in the generation of Bt and Bk horizons

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## Abstract

Our recent studies show that certain species of eucalypts are actively involved in pedogenetic processes across a number of Australian semiarid ecosystems. We highlight the role of lateral roots of such species in generation of clay pavements or horizons grossly enriched in calcium carbonate in the contemporary setting of Pleistocene dunes in southwest Western Australia. Morphology and composition of these layers are described and evidence presented, from xylem sap of tap and lateral roots, that key cationic constituents of clay (Al and Fe) and carbonate (Ca) are abstracted and uplifted by deep roots and subsequently delivered in the xylem stream to foci of clay or carbonate deposition in root platforms of the eucalypts concerned. Marked mineralisation of profiles in and below signature layers under the eucalypts contrast with the acidic depauperate profiles typically present under adjacent non-eucalypt vegetation.

## Key Words

Biomineralization, clay, pedogenetic carbonate, roots, xylem transport, hydraulic redistribution.

## Introduction

The landscapes of southwest Western Australia are characterised by widespread occurrences of variously developed intact and denuded laterites, clayey duplexes and highly calcareous soils vegetated by flora displaying extraordinary endemism, species diversity and specificity for soil type. Verboom and Pate (2006a) suggested that in each of these and other situations, idiosyncratic changes in soil profile characteristics are directly related to control of resources of water and nutrients by the dominant, deep-rooted, woody players involved. Acting at landscape scales, such activities have been suggested to explain the planiform distribution of soil types, lateral facies changes, overprinting and soil variation at very fine scales (see Verboom and Pate, 2003, 2006ab, and 2012 in press). We have coined the term 'phytotarium' to connote the cohort of higher plants and microbial players deemed responsible for engineering a specific pedogenetic outcome (see Verboom and Pate 2006a).

In this paper we consider and discuss the mechanisms involved in formation of dense clay pavements and carbonate-inflated upper profiles under eucalypts following colonisation of Pleistocene sand dunes by plants.

## Methods

### *Experimental sites and principal study species*

The first site comprised a lunette of quartzitic sand blown out from Lake Chillinup in the late Pleistocene (see Bowler, 1976 and Pate and Verboom 2009). It is currently vegetated by pristine mature mallee woodland intermixed with myrtaceous:proteaceous heath. Vegetation zones, species listings and the highly restrictive effects of clay pavements on understory vegetation were as described earlier by Pate and Verboom (2009). The principal study species is the mallee *E. incrassata*. The second site consisted of a dune of siliclastic materials flanking Lake Taarblin and vegetated at the project area by *Casuarina obesa* (swamp sheoak) and *E. longicornis* (red morrel).

### *Rooting morphologies and neoformations of clay and carbonate layers.*

These were examined using a combination of pit excavations, air and water spading of upper horizons of soil profiles and evaluations of the spatial and mass relationships between roots and respective layers. Observations were compared to those for other eucalypts in similar situations elsewhere in southwestern Australia.

*Chemical and physical composition of soil profiles under eucalypt and non eucalypt vegetation*

The elemental composition of soil was assessed within profiles passing directly through signature clay or carbonate layers in lateral root catchments of the relevant eucalypt and adjacent vegetation. Chemical and physical analyses of samples down profiles included determinations of total Fe, Al and Ca by Inductively Coupled Plasma (ICP) Optical Emission Spectrometer (OES) or ICP: Mass Spectrometer (MS) and bulk densities. Methodologies as listed by Verboom and Pate (2010).

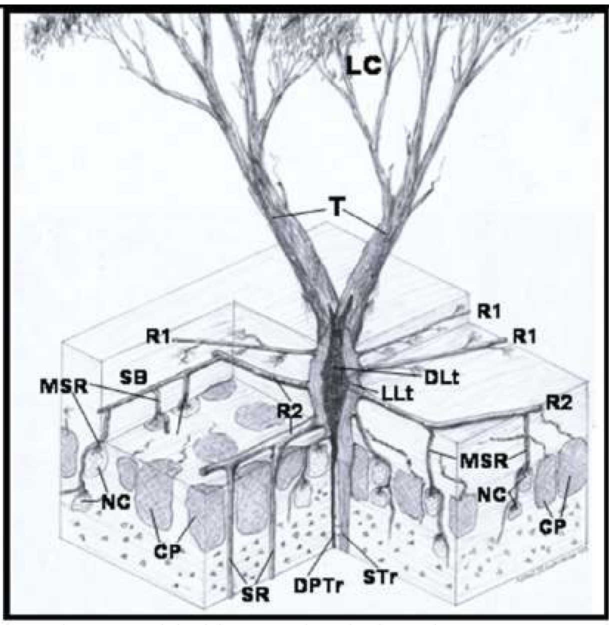
*Xylem* (tracheal) *sap collection and analysis*

Sap was obtained by vacuum extraction of clean, bark-free 30-40 cm long xylem cores cut from freshly excavated healthy lateral or tap roots of the study species using the techniques described by Pate *et al.* (1994). Parallel samples were collected similarly from principal non eucalypts at the sites. All samples were immediately membrane-filtered and frozen after collection. Analyses were conducted for Al, Fe, Ca, Mg, K, Na and Si using an ICP:OES.

**Results**

Fig.1A is a stylistic representation of a mature tree of the study mallee *Eucalyptus incrassata* showing principal morphological features (see legend to figure) in relation to clay columns of the developing pavement. As shown in Fig. 1B, clay biogenesis occurs in close association with clusters of fine rootlets borne on lower level (R2) lateral roots. Columns generated over the life of a tree, eventually coalesce into a more or less continuous, consolidated pavement.

Chemical analysis of cores through columns of pavemented profiles under *E. incrassata* showed substantial increases in bulk density and concentrations of Al and Fe (Fig. 1C), and great increases in pH, Mg and Ca status of lower profiles in comparison to cores through non eucalypt heathland (data not shown).




Drawing provided by Noel Schoknecht

Figure 1A legend-LC- leafy canopy; T- trunks; LLt- living lignotuber; DLt- decayed first-formed lignotuber; STr- secondary tap roots; DPTr- decayed primary tap root; R1- superficial lateral root lacking sinker roots; R2- lateral roots which may develop side branches SB), deeply-penetrating sinker roots (SR) and shallow mini 'sinker roots' (MSR), with new columns (NC) on their side branches. Columns of already-formed pavement (CP) are depicted in relation to decayed remains (broken lines) of a previous generation of R2 roots.

**Figure 1B: *In situ* view of juvenile column forming within a cluster of fine branch roots**

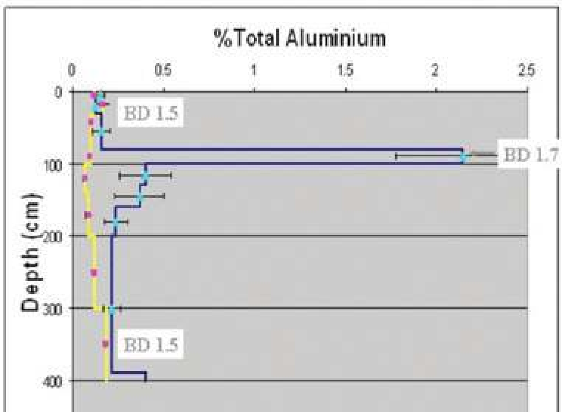



**Figure 1C. Bulk density (BD) and Al concentration profiles through nascent columns under E. incrassata (blue plot) versus that under neighbouring myrtaceous-proteaceous heath (red plot).**

Airspading of the lateral root catchments of the calcium carbonate depositing species *E. longicornis* revealed complex lateral root systems consisting of a) attenuating roots densely grouped close to the trunk and b) highly characteristic non-attenuating laterals extending horizontally for 15 m or more in linear fashion and each arising from major laterals bearing sinker roots. Sites of carbonate deposition across the catchments were consistently associated with terminal fine roots of lateral root systems (Fig 2B).

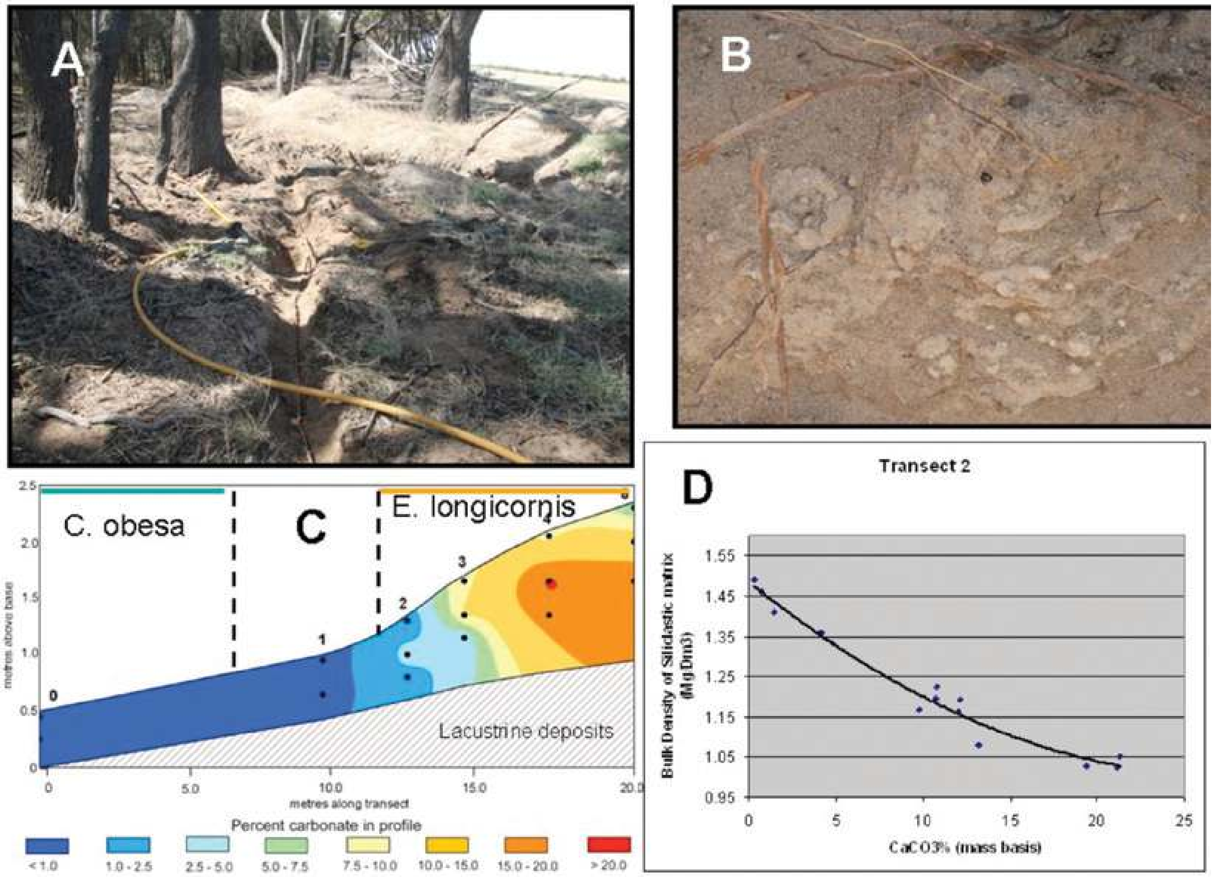



**Fig 2. A. Airspade excavation of root catchment of *E. longicornis* exposing a long non-attenuating lateral root characteristic of the species. B. Airspade excavation exposing carbonate lamellae at surface of B horizon and juxtaposition of fine lateral roots of the parent tree. C. Krig-sponsored interpolation of carbonate concentrations along a transect incorporating rooting platforms of *C. obesa* and *E. longicornis*. Depths of soil profiles above lacustrine deposits are indicated. D. Correlation plot showing an apparent second order polynomial dependency of bulk density of siliclastic matrix on percentage of deposited carbonate.**

Sampling of the transects between *C. obesa* and *E. longicornis* showed dramatic accumulations of carbonate and associated swelling in regions of the dunes profile under the eucalypt Fig. 2C and associated inflation of the siliclastic matrix as evidenced by reductions in bulk density proportional to amounts of deposited carbonate (Fig. 2D).

Note that all of the above eucalypt-related effects at both Chillinup and Taarblin have clearly been superimposed upon the original dunes matrices.

Data for xylem sap of the clay forming *E. incrassata* in Table 1A show that collections made in the dry season averaged at more than 100 times the concentration in Al ($P$= 0.008) and 5-15 times more concentrated in Si, Mg, Fe and S, ($P$= 0.008, 0.024, 0.310 and 0.992 respectively) than sap obtained in the wet season. Concentrations of Ca, K and Na were similar between the seasons.

Corresponding information for winter and summer samples for the carbonate depositing *E. longicornis* (Table 1B) showed saps of both species to be much the same during winter. However, summer samples from *E. longicornis* showed significantly higher enrichments of Ca, K and Na but extremely low levels of Al and Fe.

**Table 1: Season comparisons of xylem sap analyses of the clay pavement-forming *E. incrassata* and of the calcium carbonate layer forming *E. longicornis*.**

| **(A)** *E. incrassata* -Chillinup | | Al mg/L | Ca mg/L | Fe mg/L | K mg/L | Mg mg/L | Na mg/L | Si mg/L |
|---|---|---|---|---|---|---|---|---|
| Wet season (n=5) | Mean | 0.18 | 20.7 | 0.32 | 59.4 | 5.0 | 18.2 | 0.30 |
| | SE | 0.12 | 3.1 | 0.13 | 8.6 | 0.5 | 2.0 | 0.34 |
| Dry season (n=5) | Mean | 29.7 | 32.0 | 1.51 | 57.4 | 89.6 | 20.8 | 2.40 |
| | SE. | 16 | 8.6 | 1.1 | 8.5 | 80 | 3.4 | 0.37 |
| | | | | | | | | |
| **(B)** *E. longicornis* -Taarblin | | | | | | | | |
| Wet season (n=7) | Mean | ND | 22.1 | ND | 32.6 | 4.2 | 25.9 | 0.2 |
| | SE | ND | 6.4 | ND | 4.6 | 1.1 | 3.2 | 0.1 |
| Dry season (n=22) | Mean | 0.6 | 64.6 | 0.4 | 80.9 | 15.5 | 74.7 | 0.7 |
| | SE | 0.1 | 19.2 | 0.1 | 15.6 | 4.7 | 16.9 | 0.2 |

## Conclusions

Depositions of Al and Fe in developing Bt horizons eventuate in compacted clayey pavements in lateral root catchments of the mallee eucalypt *E. incrassata*. By contrast, precipitation of high levels of carbonate and increases in soil K characterize the B horizons under lateral roots of *E. longicornis.* Evidence from xylem sap analysis is consistent with key mineral elements in the above signature layers being supplied in the xylem to sites of deposition following hydraulic uplift of such materials from deep regions of the profile. Evidence of higher concentrations of the above elements in dry season than wet season samples is consistent with observations made by other workers on other woody species indicating that hydraulic lift and associated transport of mineral elements from lower parts of a profile is more likely to occur when stomata are closed and upper soil layers are dry (see Prieto *et al*., 2012)

We are aware of instances where other eucalypts of southwest Australian ecosystems engage in similar layer forming activities. Wider implications of bioengineering of this nature by plants are discussed at length in our recent experimental papers and reviews (Pate *et al*., 2001, Verboom and Pate 2006ab, Pate and Verboom, 2009, Verboom *et al*., 2010 and Verboom and Pate, 2012) including evidence that microbial associates of plants might play key roles in formation of layers on receipt of appropriate materials from their hosts.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# Monday 3 December

## SOIL PHILOSOPHY, SOIL EDUCATION AND THE FUTURE OF SOIL

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# Is it time for soil science to greet the open access revolution?

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## Abstract

There is a disconnection between research and application and we suggest this is about how our profession transfers knowledge and the current limitations of the processes used to make this transfer. Research students find they no longer have access to published articles in their field of expertise when they work within the commercial sector, and professionals need to enhance practical applications. Other interested parties, such as gardeners, schools and land managers, also do not often have access to soil science research. Open Access (OA) is an electronic method for communication with broad possibilities for soil science professionals to communicate within applied management, and for broad educational and collaborative outcomes. This paper seeks to highlight the existing problems with restricted access research in terms of availability and accessibility to soil science professionals, land managers and the general public. We propose the use of a broad range of OA communication methods to address this disconnection between research, practical application and public interest. We argue that an OA approach has numerous benefits for soil science research, application, education, and public profile and that it can exist as an extension to the traditional research publication process.

## Introduction

There is a problem in soil science publishing because the readership is limited by traditional database access and library subscription, or to partial ‘grabs’ generated as an internet search. Reading research articles and interpreting knowledge are the tools of trade for educated researchers and soil professionals. Yet many trained researchers will work outside an educational or research institution and therefore outside access to their traditional tools-of-trade - research literature - and many will have changed demographic to that of the consultant community (Cattle 2008).

Open access (OA) is an electronic method by which information can be widely accessed. Technically OA literature is digital, online, free of charge, and free of most copyright and licensing restrictions (Suber, 2012). OA literature for the scientific community is as important a development as was the Xerox machine, a technology which brought to the general and business public greater access to information. Introduced in 1959 the ability to photocopy meant reproduction of documents was cheaper and more freely available (Fuji Xerox 1999). Research undertaken by Eysenbach (2006) and Hajjem *et al.* (2005) found that OA articles were more widely cited than were traditional publications. Eysenbach (2006) suggests that OA does have the potential to be a faster way of sharing and applying practical research. The amount of impact an OA journal has needs further thought and to consider a range of variables, including download counts and quality (Hajjem et al. 2005). OA has the potential to be a free way for young and established scientists, and applied soil practitioners to remain informed of technical developments, and provide a platform for information exchange and support across a wide communication network.

## Open Access Platforms

### *Traditional*

Currently there are about 10 fully open access internet journals in Europe, South America, India, and Australia specifically targeted for soil research. Traditional journals are also now allowing open access options at a cost to the authors averaging $2500USD (Drake 2011), or moving to a completely OA platform (e.g. EMBO Molecular Medicine) with information publically available under the terms of the Creative Commons Attribution Non Commercial License (Wiley-VCH 2012). The availability of freely accessible science is important for scientists, professionals and managers world-wide who would otherwise not have access to new science due to lack of institutional access or where the cost of journal access is out of reach.

Accessibility, by means of the audience to understand the content of the journals, still requires development. Of all the fully open access journals, only one journal as of May 2012 has been specifically developed and edited for a management audience. FLAMMA is a Spanish soil science journal for soil issues and management post-wildfires (FUEGORED 2012) also published under the terms of the Creative Commons Attribution Non Commercial License (Creative Commons 2012). A two-way journal between soil scientists and managers, it is written non-scientifically as a bulletin and targeted at action-based management outcomes from science and research. It is free to publish in and view the journal, and is run by a small number of dedicated and passionate OA soil researchers. Australia and New Zealand has the potential to also provide OA journals that are targeted at a broader audience, such as land managers, to improve engagement, collaboration and uptake in soil science. OA provides continual professional development, science uptake and integration, science-based management decisions, and improved research particularly in places where access to traditional journals is limited.

*Non-Traditional*

Social media are becoming an increasingly easy and accessible way of communicating and providing science information. Blogs, Twitter and Facebook are three of the largest forms on the internet for sharing information. There are currently 17 soil science blogs in English, with www.soilduck.com and www.turfhugger.com the only known soil science blogs in Australia. Blogs are used by gardeners, land managers, interested people, schools and universities. Blogs allow the human element to come back into science, as a way of gaining trust and interest from non-scientists (Drake 2011; Zivkovic 2010). This may be one of the reasons blogs are so broadly used as a source of scientific information.

Blogs can also be used as a way to share ideas, or to discuss research ideas and get open peer review, collaboration and development of ideas (Drake 2011). The open sharing of hardware and software in the IT industry is part of the open source revolution (Open Source Initiative, OSI). Open source is a term describing a means of developing and distributing software that ensures software is available for use, modification, and redistribution by anyone (OSI 2012). It has allowed faster development, collaboration, peer review, continuity, low cost and profitable business models, and improved quality (LSWG 2000; Portelli 2012). This form of informal peer review has the potential to be integrated within soil science research to considerable benefit. For example, peer review of soil science, informally through blogs, at the early stages of research can also benefit soil science development and innovation, and provide immediate feedback leading to more robust and useful ideas at the offset, rather than in post-research review after the work has been completed. Blogs also allow researchers, professionals and land managers to get to know each other, and create new forms of networking and trust which were previously unavailable.

Twitter is a resource for sharing information and meeting people with similar interests. There are more than 150 English speaking soil professionals and groups on Twitter, with only a handful of these people in Australia and New Zealand. Many of these people and groups interact with land managers, through hashtag (#) discussions such as #agchat and #agchatoz. Some #agchatoz and #education people asked @soilduck to write a blog post on open access and accessible soil science (Drake 2012), as well as asking soil science 'tweeps', such as @soilmalone, @danicarno and @helenpking to comment on particular soil science issues. Collaboration and knowledge sharing between the general population and scientists has happened through Twitter, and is continuing to increase (Moon 2012). This is further supported by soil science society Facebook pages including the Soil Science Society of America, the British Society of Soil Science, the New Zealand Society of Soil Science and Soil Science Australia. These pages have a lot of sharing of information, research and collaboration between researchers and soil professionals of all ages and backgrounds, and provide access and opportunity to network and share.

Wikis are great introductions and explanations to launch into further learning. They allow people to contribute information in a peer-reviewed platform in an accessible way, and have and can be used by non-soil specialists (Wilson *et al*. 2010). As of May 2012, there were three English language soil science wikis on the internet: New Soil Net, Soil Scientists and Soil Analysis Support System for Archaeologists. The first two have little contribution from the soil science community and are stagnating. This is in contrast to The Microbe Wiki, which has some great sections on all types of soil biology, including flooded soil microbiology, and reflects the broad applications that soil biology will support. The reasons that the Wikis have not had uptake is uncertain. There is potential for soil science societies to start and/or endorse a wiki which could be linked from other social media sources (i.e. blogs, Twitter and Facebook) and prevent stagnation with ongoing updates and referrals. Developing soil Wikis can promote access to all forms of soil science including methods, processes and analysis that are not in thorough detail in journals, and can allow professionals, scientists and managers to exchange methods and approaches, and improve soil science understanding (Wilson *et al.* 2010). This approach has great potential as periodic updates to method texts (e.g. the intervening years from Rayment and Higginson (1992) to Rayment and Lyons (2011)) can be considered too infrequent to keep pace with rapid progress in some areas of method development.

Collaborative platforms, such as Research Gate (www.researchgate.net) and Mendeley (www.mendeley.com), are another way for professionals, managers and interested people to connect, discuss, ask questions, share information and collaborate on soil research projects. Both are open to anyone to join and have a professional profile, people and interests which you can follow. It is designed for networking, asking questions, collaborating, and sharing research and papers. Land managers use Research Gate as a way of finding information from professionals on how to best manage their soils, which shows the potential for Research Gate as an OA platform for practical application of soil science research.

These non-traditional forms of soil science communication demonstrate an ability to share, educate and collaborate with a broader audience and also within scientific, manager and soil professional circles. Engaging in alternative media provides networking opportunities, which can help professional development, soil science uptake and management, education, and development of soil science itself. However, the uptake and use of these innovative approaches is still in the early stages in Australia and New Zealand highlighting the potential for greater use in the soil science community.

### Vetting and peer-review online

One of the biggest problems with online environments is the credibility of the source. Wikipedia is a widely used source, but the reader does not know the background of the contributor. Wikipedia, however, does have a strong editing and peer review system. References are required, and professionals will often edit or add to the content if they determine that it is not correct. This is not the same for blogs, comments on websites or other non-traditional OA sources where there is often a lack of transparency or a credible editing and peer-review system.

The Hypothes.is project (http://hypothes.is/) is hoping to create a real time vetting and contribution platform for any and all web pages. This will allow real life, real-time peer-review from people who are experts or are known to be involved in that specific subject area. It will also allow for vetting in comment areas of blogs and other online sources, where professionals in that field will have a stronger peer-review presence than other commenters. The Hypothes.is platform has the potential to improve the quality of information available on the internet. In addition, online newspapers and blogs such as The Conversation, The Global Mail and Scientific American Blogs, provide their readers with a profile of the writer. This allows the reader to then make an educated decision about whether or not the content of the article is written by an expert or not. Comments on blogs can also have a profile attached, which allows informal peer-review on blogs and online newspapers.

With these vetting and peer-review platforms, there is less reason not to get involved in the OA revolution. OA also means accessibility outside traditional educational models of lectures and workshops, allowing people to search and read independently. The internet has good and bad content, including bad content on soils. People will find this and will read it regardless of its quality. Thus, why not add to that pool and provide better vetting for internet content and education opportunities?

### OA does not imply accessibility

Making soil science openly available, through journals and other sources, does not mean that it is accessible to a wide audience. If soil scientists want to reach a broader audience, whether it be land managers, researchers, youth, gardeners or other interested people, there is a need to develop OA information that communicates soil science to this broad population. We need to be trained in science communication (Silva & Bultitude 2009), and consider language, content, as well as making it personal and bringing enthusiasm and interest into the work (Zivkovic 2010).

### The future of OA in Australian and New Zealand soil science

OA allows freely available soil science information for all. It has the potential to improve collaboration, which in turn improves science and management, and allows professional development. It increases the ability for people to put research into practice, and make science accessible as an educational resource. There is increasing evidence to demonstrate the positives of OA science, both by improving science itself and its practical uptake. There are few, if any, disadvantages to making soil science OA in some form.

Indeed, other science disciplines are already well ahead in adopting OA approaches to research. Some ways in which scientists, practitioners and land managers at Soil Science Australia/New Zealand can get involved in OA soil science may include making time to contribute to a currently available blog or online papers, joining Twitter or the Facebook groups, linking the Australian and New Zealand soil science society websites and Facebook pages, developing a blog, getting involved with online collaboration platforms, publishing more management and practical articles in OA journals and developing an Austral-NZ OA soil-management journal for researchers and managers.

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# Producing YouTube videos on soil erosion research: Combining data, farmer experience and eye catching visuals

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In our organisation, there has been a history of printing brochures to convey information about research projects to the target audience. These factsheets have more recently been hosted on a website for convenience, but surveys have shown that time poor and younger grower audience would benefit more from a visual package of information. The decision to create YouTube videos to explain the findings of our soil erosion research project in the macadamia industry was made easy due to the growers' previous exposure to videos for information delivery over several years on DVDs.

Research and extension staff underwent video production training by local professionals. The course covered scripting, filming, interview skills, technical training on camera equipment and basic editing. We developed scripts, decided on the best visual display to present data and interviewed the researcher and several growers using best practice soil management methods. In one video, soil loss around trees was displayed on a two dimensional map at the experiment's start and end. The pattern of soil loss due to different harvest machinery was represented with simple animation. Slow motion footage at ground level captured the soil movement spectacularly. The response to the videos has been very positive and number of hits is used as a measure of reach. Posters describing our research also display a QR code with a link to the videos for viewing on smartphones. The industry and state agencies in Queensland and NSW have since developed a website (MacSmart) solely for communication, which is heavily video based.

http://www.youtube.com/watch?v=ojtGQMrgo64

http://www.youtube.com/watch?v=Srfq8zE4k0E

# Small heterogeneous associations and water retention link soil quality to carbon sequestration in soils: Philosophical and practical implications.

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**Abstract**

Extensive microaggregation and clear micropores characterised a virgin soil. Cultivation led to fewer and smaller microaggregates, with textural particles occupying pores. Both soil and clay was lost as a result. A published measure of soil quality derived from the water retention curve promises to be useful to explain the loss of quality, which results from the loss of microaggregates and micropores. Microaggregates can protect organic carbon (OC) and the water retention curve could similarly be used to indicate the extent of sequestration of OC. Microaggregates, as small heterogeneous associations, are the fundamental stable entities of good quality soils and practices that lead to their formation and stabilisation also encourage carbon sequestration. Other philosophical and practical implications of this conclusion are discussed.

**Key Words**

Microaggregates, micropores, carbon storage, cultivation, erosion, degradation

## Introduction

Arguably, the two main global challenges or broad objectives for soil scientists at this time are:

1. To enable soils to provide enough food (and shelter, fibre and energy) to sustain a rapidly growing population while sustaining the soil base itself, and

2. To enable soils to provide a sink for rapidly increasing emissions into the atmosphere of carbon dioxide and other greenhouse gases.

Objective 1 requires measurement of the response of the soil under the increased strains brought about by the need for increasing plant productivity. For at least 20 years, the concept of 'soil quality' has been proposed as an indicator of "the capacity of a soil to sustain biological productivity, maintain environmental quality and promote plant and animal health" (Wienhold et al., 2005, p. 349). However, there is no consensus about the soil properties that constitute soil quality. Generally, it is considered to be indicated by a variety of soil properties or characteristics. There is also a lack of consensus on the methods to be used for its quantification (e.g., Bastida et al., 2008). Soil quality as a concept has also met strong criticism from some, such as Sojka et al. (2003), who label it "elusive and value-laden".

Even so, soil degradation occurs as a common result of agricultural practices. Reference to a comparison between sites nearby one another in South Australia which each contain the same soil type but which have differed in their type of agricultural management and its duration (Churchman et al, 2010) illustrates the need for measures of soil quality. Degradation has clearly occurred. It is deduced that one particular index that is relevant to this South Australian example also indicates the ideal requirements for meeting objective 2, i.e. the best requirements for soils to be able to sequester carbon. The dual relevance to both Objectives 1 and 2 of this particular index, which is obtained from whole soil samples, is discussed in this paper.

## Material and Methods

The studies of the effects of land management were carried out on a soil classified as a Calcic Haploxeralf by Soil Taxonomy (Soil Survey Staff, 1992) and a Red Chromosol by the Australian Soil Classification (Isbell, 1996) at (1-4) a virgin site, (A) an adjacent farmed site, and (Tc, Tn)) nearby rotation and tillage trial sites (Fig. 1). The soils at the tillage and rotation trial sites had been farmed by conventional practices for ~100 years prior to the establishment of the trials, which began 18 years before the soils were sampled. The virgin site and the adjacent farmed site had been part of a churchyard which contained a church building until 1949. While the virgin site has never been cultivated, the adjacent farmed site began to be used for agriculture in quite recent times and the local residents report that it has been cultivated more lightly than the land at the rotation and tillage trial site. Further descriptions of the sites and soils are given in Churchman et al. (2010). Methods of analysis used are given therein and also within the Results section herein, as appropriate.

**Results**


Figure 1. Aerial photo (taken December 2006; from Google Earth) of the virgin sites (Nos. 1-4) and the adjacent farmed site (A).

The trial sites (Tc – conventionally cultivated for ~120yr, and Tn – no-till for 18yr after ~100 yr conventionally cultivated) are located ~300m beyond the lower L.H. corner of the photograph.

Tc Tn

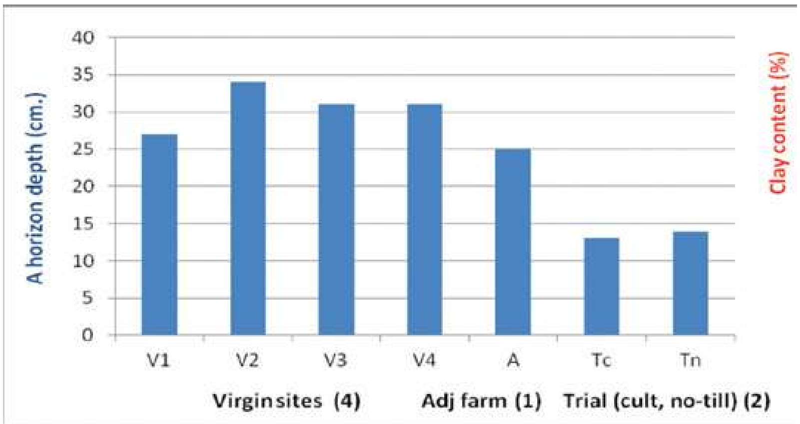


Figure 2. Changes in depth and clay content of the A horizon with changes in land use from virgin (V1-4) to adjacent farmed (A), conventionally cultivated (Tc) for ~120 years, to no-till for 18 yr. after con. cult. for ~100 yr. Shows erosion as total loss of A horizon (loss of depth) and as coarsening of soil (loss of clay). Clay content was measured in only one of the virgin soils.

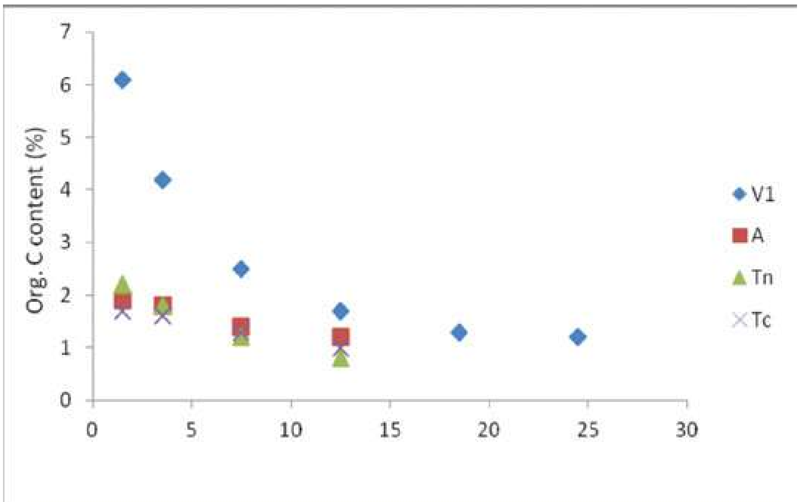


Figure 3. Profile of organic C (%) with depth in A horizon as affected by land use. OM contents in farmed soils are very similar to each other but lower than in virgin soil at similar depths.

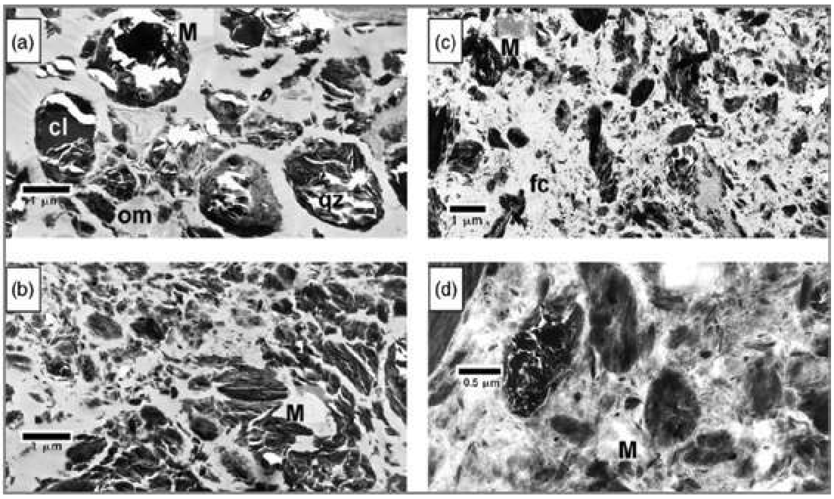


**Figure 4. Transmision electron microscopy of ultrathin sections of samples from within the top 5 cm at (a) virgin site V1, (b) adjacent farmed site A, (c) site under long-term con. cult. Tc, (d) site under no-till following long-term con. cult Tn. Scale bars are for 1 µm in all except (d), where it is for 0.5 µm. M = microaggregate; qz = quartz; cl = clay; om = org. matter; fc = fine clay**

Clearly, conventional cultivation over ~ 100 yr has led to a bulk loss of soils, together with a selective loss of fine (clay) material (Fig. 2). Neither is restored by no-till for 18 years. Lighter cultivation at Site A appeared to have avoided significant erosion. However, cultivation at all sites led to a loss of organic matter (Fig. 3). Photos of ultrathin sections of A horizon samples examined by transmission electron microscopy are shown in Fig. 4.

Virtually all of the < 2 µm material occurred within microaggregates of coarse clay size (1-2 µm) in soil at the virgin site (Fig. 4a) and most of this clay-size material occurred within smaller microaggregates at the adjacent farmed site (Fig. 4b). However, microaggregates were not only smaller but also less common in the two cultivated sites (Figs. 4c,d), where there was much dispersed fine clay that blocked pores which form between microaggregates. Pores were largely empty in the virgin site (Fig. 4a).

**Discussion**

The breakdown of microaggregates to fine clay represents an irreversible loss of structure in these soils (Tisdall and Oades, 1982; Churchman et al., 2010). The fine clay thus released is available for erosion, which occurs predominantly by wind in these soils within a flat landscape, hence to loss of both bulk soil and the winnowing out of the fine clay material (Fig. 2). Pores important for the transport of water, nutrients and air are also lost. These effects of long-term cultivation represent a most serious loss of soil quality.

Among the various soil quality indices, one proposed by Dexter (2004) that is based on the water characteristic curve, gives a measure which is related to the microstructure of a soil that is sensitive to degradation by e.g. extensive tillage (Dexter and Czy, 2007). It would be suitable to indicate the loss of quality seen in the South Australian soil under study. Organic matter content, proposed by many to indicate soil quality (e.g. Bastida et al., 2008) is not particularly sensitive to important changes to soil quality in the situation studied. Thus the adjacent farmed soil, which showed little deterioration at the fine particle size scale (Fig. 4b) nonetheless had similarly low organic C contents throughout its A horizons to the long-term cultivated soils (Fig. 3) that were clearly more degraded at the fine size scale (Fig. 4c,d).

Much recent evidence (e.g. Lehmann et al., 2007; Schmidt et al., 2011) indicates that microaggregation largely explains the extent to which a soil may sequester carbon. Carbon is sequestered when it persists in the soil for a long time, at least 100, and maybe thousands, of years. It was once thought that it persists because it is transformed from its plant and microbial forms into a recalcitrant form, a different type of molecule. However, recalcitrant charcoal-like forms of C in soils are a special case and usually a minor component, even if widespread.

Microstructure is made up of both micropores and microaggregates and the nature and extent of microporosity reflects the extent of microaggregation, as shown in Fig. 4. Using a more recent representation of the water retention equation, Dexter et al. (2008) have been able to separately distinguish the (micro)pore spaces due to structure and texture. Regardless of the method of its interpretation, the water retention curve offers the prospect of defining the extent and nature of microaggregation of soils. Therefore it may also be able to indicate the extent of sequestration of organic C. This is because stable microaggregates, better characterized as small heterogeneous associations (SHAs), provide the most secure environments within soils for the protection of organic C from predators and oxidation. As an example, Andisols, with high contents of allophane, provide good examples of SHAs for maintaining high soil quality and for sequestering considerable amounts of organic carbon.

## Conclusions

### *Philosophical Implications*

1. The slope of the moisture characteristic shows that soil physical quality is related to extent of microaggregation
2. Microaggregates protect organic matter for long-term storage
3. Microaggregates are better described as small heterogeneous association (SHAs)
4. Soil quality and C sequestration potential are enhanced by SHAs
5. SHAs – not textural particles - are the fundamental stable entities of good quality soils
6. As far as possible, soils should be studied as a whole

### *Practical Implications*

A. As SHAs, microaggregates take time to form and their formation result from biological processes, mainly the life and death of plants; hence carbon sequestration cannot be rushed

B. The availability of clays and oxides helps, and C storage may be assisted by their addition

C. Good soil management, e.g.optimising plant growth and clay content, and minimum tillage, should encourage enhanced microaggregation (or SHAs), better soil quality, and carbon sequestration.

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## Acknowledgements

I am grateful to Tony Dexter in Pulawy, Poland for useful discussions by email.

# Monday 3 December

## SOIL FERTILITY

*Presented Posters*

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## The vertical distribution of Cs-137 in Bavarian forest soils

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Persistently high activity concentrations of radioactive Cs-137 (T1/2 = 30.17 a) in various animals and fruits originating from Bavarian forests suggest that the contamination of soils in these ecosystems is still critical even decades after the severe inputs following the Chernobyl nuclear accident. Aware of the fact, that such inputs are a global threat that can re-emerge at any time, a new monitoring network was established in cooperation with the Bavarian State Ministry of the Environment and Public Health, to enhance the value of long-term radioprotection strategies in forests. Based on the investigation of 48 forest sites throughout the entire state territory, the project delivers a total of 889 gamma spectrometric records and demonstrates the current Cs-137 contamination situation of Bavarian forest soils, providing a valuable update on the residual contamination levels and thus a comprehensive inventory for any future radioprotection management.

Results are presented hereby. The total Cs-137 areal activity densities in Bavarian forest soils currently vary between 640 and 61,166 Bq m-2, with the peak areal activity density of each profile being located in the uppermost, humus rich mineral A-horizon in 68 % of all cases. Moreover, the results detect a positive correlation of humus thickness and relative areal Cs-137 activity density in humus horizons (R2 = 0.50), validating previous findings on that topic by means of a very comprehensive data set across 2.56 Mio ha forest stands by showing that humus bodies <7.5 cm still contain at least 50 % of the total areal topsoil activity density.

# Biosolids reuse in New Zealand – assessing the impacts of chemical cocktails on the soil ecosystem

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Of the 200,000 dry tonnes of biosolids produced annually in New Zealand over 90% is disposed of in landfills. There is a strong scientific case that land application is the most sustainable alternative, because biosolids contain high concentrations of valuable nutrients that can reduce dependence on artificial fertilisers. However, biosolids can also contain mixtures of micro-contaminants (metals and organic contaminants) the additive effects of which may result in a combined effect level that can be greater than the sum of the individual effects.

We aim to characterise the environmental risks arising from land application of biosolids containing a mixture of contaminants. Small lysimeters were established using field soils historically contaminated with copper and zinc at a range of concentrations (Cu: 50, 120, 300, 750, 2000 ppm; Zn: 70, 160, 400, 1000, 3000 ppm); with the addition of the organic contaminant triclosan (a commonly used antimicrobial in bodycare products) at 5 ppm and 50 ppm. A range of soil biological indices (e.g. soil enzymes, sensitive microbial biosensors, Rhizobium and molecular analysis of the soil microbial community) were used to determine impacts of the metal + organic mixtures. After 6 months, results indicated that the degradation and transformation of triclosan is reduced in the presence of high levels of co-contaminants (e.g. heavy metals), leading to an increased half life and persistence in soils. Some soil properties were sensitive to the levels of triclosan used in this study and for some properties, a synergistic effect of the presence of co-contaminants was observed.

# Compost effects on microbial activity and biomass and soil P pools as affected by particle size and soil pH

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Due to the low availability of phosphorus in many soils, P addition is required for adequate plant growth and yield. With diminishing world reserves of P deposits and rising fertiliser prices, it is important to find alternative sources of P for crops, such as compost. Composts consist of particles of different sizes which may differ in P release. Additionally decomposition and the size of various P pools differ in soils with different pH. Compost produced from garden waste was separated into fraction > 5 mm, 3-5 mm and < 3 mm and these fractions were mixed separately into three soils: acidic soil (pH 4.8), neutral soil (pH 6.1) and alkaline soil (pH 8.4) at a rate of 0.05 mg P $kg^{-1}$ soil. Unamended soils were used as controls. The soils were incubated at constant temperature and sampled on day 25 and 50. Cumulative respiration was measured from day 0 to day 25 and from day 26 to day 50. In both phases, cumulative respiration was highest in the acidic soil irrespective of compost addition due to its higher organic matter content. In all soils, cumulative respiration was greatest after addition of the largest fraction which had the highest C concentration. Microbial biomass carbon was also highest in acidic soil with addition of smaller particle sizes. The available P concentration was highest in the soils with the smallest particle size and decreased over time. The determination of the P pools is in progress.

# Visualisation of phosphorus diffusion from granular and fluid fertilizers in non-calcareous strongly phosphorus-sorbing soils

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Low phosphorus (P) availability is an important constraint for plant growth in strongly P-sorbing soils. In these soils, a large amount of P fertilizer is necessary to counteract low P efficiency, as P can be rapidly and irreversibly converted to forms unavailable for plant uptake. The increasing costs of P fertilizers have prompted research interest in improving fertilizer efficiency. However, improvements can only be achieved when the behaviour of fertilizers in soils is clearly understood. A laboratory incubation was conducted to investigate the diffusion of P from various granular (five sources) and fluid (two sources) fertilizers using a novel visualization method. All P fertilizers were applied at the same rate to the surface of slightly acidic high-P sorbing soils: an Andisol from New Zealand and two Oxisols from Australia. Additionally, a calcic Inceptisol and a non-P sorbing Alfisol from Australia were included for comparison. Significantly greater diffusion of P was observed for the fluid than for the granular fertilizers in the Oxisols, Andisol, and in the calcic Inceptisol, but not in the Alfisol. These results indicate that fluid P might be an effective fertilizer in certain strongly P-sorbing soils. The different behaviour of fluid and granular P is being further investigated using isotopic labeling technique with $^{32P.}$

# Development of the Diffusive Gradients in Thin-films (DGT) technique to measure plant-available potassium in soils

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Soil testing before seeding plays an irreplaceable role in optimizing crop yield by means of guiding a proper rate of fertilizer application. The current commercial methods for assessing available K in soil are mainly based on chemical extraction of the soil sample and relationships between extractable nutrient and crop growth response appear to be soil type dependent. The Diffusive Gradients in Thin films (DGT) technique has been successfully applied to measure available P in soil and more accurately predicts crop P requirement compared to traditional chemical methods. To adapt the DGT method to simultaneously measure available P and K in soil, a suitable binding medium is needed to create a diffusion gradient in the soil. We examined the properties of mixed binding layers created through mixing Amberlite cation exchange resin with ferrihydrite. The elution efficiency, capacity and diffusion coefficient of the new mixed Amberlite and ferrihydrite gel (MAF) for K were 80%, 800 µg and 1.007x10-5cm2 s-1 at 22□, respectively. A soil survey of 4 soil K test methods, solution K, NH4OAc K (Exchangeable K), Colwell K and DGT K, were carried out on 20 agricultural soils from southern Australia, with DGT K being assessed after a 24 hours deployment. The moderate to poor correlation between the DGT K and other K test methods suggests DGT is measuring a different pool of K in soil than the other tests. In addition, there was no significant difference (P□0.01) between DGT P measurements determined using the traditional P ferrihydrite gel and the MAF gel. Therefore, the MAF gel has potential to measure both P and K in one assay. In order to assess the ability of DGT to predict crop responses to K fertilizer, plant response experiments are necessary to compare relationships with extracted K.

# Effects of long-term grassland management on the chemical nature and availability of soil phosphorus.

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Relationships between the relative solubility of soil phosphorus (P) determined by sequential fractionation of inorganic P (Pi) and organic P (Po) and short-term plant P uptake were investigated using soils obtained from a long term field trial. The trial had been maintained under a contrasting mowing regimen (no mowing, mowing with clippings left, mowing with clippings removed) for 15 years. The P fertility gradient established between the three contrasting mowing regimes provided a unique template to investigate how soil P extractability-solubility relates to short-term plant P uptake. In a 5 month glasshouse pot experiment, P uptake by red clover and Italian ryegrass was found to be 40% lower for the clippings removed treatment compared with the no mowing treatment, which was consistent with the fact that concentrations of readily extracted inorganic P were 42% lower in the clippings removed treatment soil. However, P uptake was 51–54% higher for the clippings left treatment soil compared with no mowing, despite the fact that levels of readily extracted soil inorganic P were similar for the no mowing (247 mg P kg$^{-1}$) and clippings left (223 mg P kg$^{-1}$) treatments. This indicated that biological and biochemical processes associated with enhanced mineralisation of organic P and turnover of P through the microbial biomass made a greater contribution to increased plant P uptake in the clippings left soil compared with the other treatments. These findings highlight the importance of soil biological processes in determining the P nutrition and productivity of managed grasslands.

# What is BSES extractable P?

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Our understanding of the pools of plant available phosphorus (P) during the growing season remains incomplete, and is critical to predicting fertiliser response in Vertosols. The BSES soil P extraction method is being used in the northern grains region (NGR) in addition to standard soil tests to provide a measure of "slowly available" P in Vertisols. To understand what P minerals are extracted using the BSES method, P K-edge X-ray absorption near edge structure (XANES) and total element determination via PXRF and ICP-OES were conducted on soil before and after BSES extraction, and solution extracts. BSES solution extracts contained appreciable amounts of Ca and P which supported P K-edge XANES spectra demonstrating complete removal of the dominant Ca phosphate after BSES extraction. The BSES extractable residue did not contain any identifiable P mineral as measured by P K-edge XANES. BSES extractable P and total soil P was strongly correlated with BSES-P. In general, the inclusion of co-occurring P elements such as Al, Ca, Mg, Mn and Fe did not substantially improve the adjusted regression coefficient ($r^2$). However, there was a general trend that as BSES-P increased so did total soil Ca and Fe. This study confirms BSES extraction removes Ca phosphate and is related to total P in Vertisols located in northern NSW.

# Amelioration of slowly permeable hypersaline peaty-clayey sulfuric and sulfidic materials in acid sulfate soils by mixing with friable sandy loam soil

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## Abstract

Ecological and environmental impact assessment of acid sulfate soils (ASS) has become an important issue for the management of ASS in many countries. The principal management options aim to minimise disturbance of sulfidic and sulfuric materials in ASS while neutralizing actual or potential acidity with alkaline materials such as agricultural lime, in order to maintain good soil microbial activity and enhance the acid neutralizing capacity of the soil. There are however limitations in adopting some of the conventional management options because of the presence of ASS materials with hypersaline and slowly permeable properties, as occurs at the Gillman site in South Australia. It is therefore important to investigate and make available management options for concerned stakeholders that will be suitable for specific environments. This paper discusses the results of two laboratory-based experiments carried out to investigate the ameliorative potential of adding friable sandy loam (with neutral pH) to sulfuric and sulfidic materials that are extremely slowly permeable and hypersaline. The results show that mixing with sandy loam has the potential to buffer the negative impacts of acid materials in sulfuric soil or prevent sulfuric soil from acidifying when oxidized, and that the buffering capacity could be permanent when established. This is based on the observation that pH of both the sulfuric and sulfidic soils mixed with sandy loam was constant over time, in contrast to the changes observed in the sulfidic soil alone. There are several advantages of using sandy loam for ameliorating such soils, including the ease of mixing compared to lime, and the increased permeability, which will facilitate leaching of salt and the establishment of vegetation.

## Key Words

Amelioration, acid sulfate soil, friable sandy loam soil, soil permeability

## Introduction

Acid sulfate soils are naturally occurring soils, sediments or substrates formed under waterlogged, reducing conditions, with sulfide minerals that either contain sulfuric acid or have the potential to form it, in an amount that can have an impact on the soil. In general, ASS can occur in subaqueous, waterlogged and drained conditions in coastal, inland, mine spoil and wetland environments (Fitzpatrick *et al.* 2009). Acid sulfate soils contain oxidized minerals (pyrites), or their products, and have been described as the "nastiest soils on earth" because of their strong acidity, and their ability to move potentially harmful elements. Although ASS cover only a small area (17 million ha) of the world's problem soils (Poch *et al.* 2009)South Australia</title><secondary-title>Australia Journal of Soil Research</secondary-title></titles><pages>403-422</pages><volume>47</volume><number>1</number><dates><year>2009</year></dates><urls></urls></record></Cite></EndNote>, once the acid and the toxic elements are mobilized, they can have a variety of impacts on the environment (Ljung *et al.* 2009).

The assessment and management of ASS has become an important issue because of the wide range of threats ASS pose to different ecosystems (Vegas-Vilarrubia *et al.* 2008).The main impacts include reduced water quality, adverse effects to aquatic life, reduced soil fertility and crop productivity, and corrosion of concrete and steel structures on the built environment. Many studies have been conducted to try to assess the complex process of ASS formation and their chemistry (Ahern *et al.* 2004) as well as strategies that can be used to manage their impacts (Ljung *et al.* 2009). Some of the principal management strategies explored include minimising disturbance of the ASS, application of an alkaline material to neutralise the acidity (Ljung *et al.* 2009), incorporation of organic matter to improve the soil microbial activities (Oliveira and Pampulha 2006) and use of vegetation to extract chemical pollutants from the ASS (Haling *et al.* 2010).

There are however no studies to date that have investigated the potential of sandy loam soil to ameliorate the acidity of actual (sulfuric) or potential (sulfidic) ASS, which when used can facilitate permeability, leach salt and make the ASS more conducive to the colonisation by vegetation. This paper discusses the results of two experiments conducted to investigate the ameliorative potential of sandy loam on extremely slowly permeable and hypersaline sulfuric and sulfidic soil materials.

## Methods

### *Soil collection and preparation*

The soils (both sulfuric and sulfidic) used in the experiments were collected from an already identified acid sulfate soil site in Gilman, South Australia. The collected soils were carefully identified whilst in the field and kept separately in several buckets, labelled as either sulfuric or sulfidic soils. Sandy loam originating from Langhorne Creek, South Australia was obtained from a garden supplier and will be referred to as LC sandy loam.

Prior to setting up the experiments, both the sulfuric and sulfidic soils were washed using a cement mixer. After mixing, excess water was carefully drained. This process of washing was repeated several times until the conductivity of the soil suspension was below 5 dS/m. The purpose of reducing the conductivity was to make the soil substrate mixture conducive for establishing vegetation in later experiments. These suspensions were then filtered through coarse cloth and rinsed several times. The final rinsed soil was air dried and filtered again using a 2 mm sieving tube prior to use.

### *Measurement of pH and data analysis*

pH was measured in a 1:5 soil: water suspension following 5 min shaking and 30 min settling. For water-based pH measurements (pHw), 10 g of wet soil was used. To assess the potential acidity of the soil types, soil samples were oxidised with hydrogen peroxide before pH measurements (pHox). For this, 10 g of target soil was oven dried for 30 minutes at 60˚C then ground using a pestle and mortar. Two grams of the ground soil was used for pH measurements as per Ahern *et al*. (2004). All pH measurements (pHw and pHox) were made using an Orion pH meter (model SA 720). The data were analysed using Statistix 9.1 (Statistical Software), Tallahassee, USA. One-way ANOVA was also performed to test the significant differences between the treatments means at $p<0.05$.

### *Amelioration of sulfuric soil by mixing with sandy loam*

This experiment was carried out to investigate the ameliorative potential of LC sandy loam when mixed with sulfuric soil. A 1:1 mixture of these soils had a pHw of 5.78, pHox of 2.90 and conductivity of 3.23 dS/m. For comparison, sulfidic soil (pHw 6.72, pHox 2.10, conductivity 0.60 dS/m) and sulfuric soil (pHw 2.36, pHox 1.99 and conductivity of 18.70 dS/m) were incubated under the same conditions as the sulfuric/LC sandy loam mixtures. Additionally, a sulfuric soil was included for comparison.

Samples of each of the soil types were brought to field capacity and 20 g subsamples were spread onto 9 cm Petri dishes with 4 compartments. The overall thickness of the soil layer was approximately 2 mm as per Sullivan *et al*. (2009) as a standard thickness however, the incubation time was altered as per the needs of the experiment. The treatments were replicated four times and incubated at 24$^{o}$C. One compartment of each treatment was sampled for pH measurement every 3 d for 12 d (Figure 1).

### *Ameliorative effects of sandy loam on sulfidic soil*

To investigate the ameliorative effects of LC sandy loam on sulfidic soil, two incubations were set up. In the first incubation, sulfidic soil (pHw 6.02, pHox 2.06, conductivity 1.75 dS/m, 32.8 percent water holding capacity) was mixed with LC sandy loam (pHw 9.37, pHox 7.22, conductivity 0.08 dS/m, water holding capacity 23.6 percent) at a ratio of 1:1 (w/w). The resulting soil mixture had pHw 7.75; pHox 1.93 and conductivity 0.12 dS/m respectively. The water holding capacity of the final mixture was 17.9 percent. In the second incubation, sulfidic soil alone was used. Both soil substrates were set to 100 percent field capacity prior to the experiments.

A total of 20 g of both soil types was taken and placed in two Petri dishes and spread out evenly to form an approximately 2 mm thick layer and replicated three times. The Petri dishes were sealed with insulation tape to maintain 100 percent field capacity. Six dishes of each treatment were incubated at 24°C and three of each treatment at 4°C. The latter treatment was included to assess the performance of the test soil substrates under cooler winter conditions (Figure 2).

**Results**

*Amelioration of sulfuric soil by mixing with sandy loam*

When mixed with sandy loam, the pH of the sulfuric soil increased from 2.36 to 5.78 and then remained constant or increased slightly over the 12 d of incubation. In comparison, the pH of the sulfidic soil decreased from 6.72 to around 3.5 (Figure 1).

The results indicate that the neutralising effect of the sandy loam is sustained and that any residual sulphides in this soil are effectively buffered. The overall tendency of the pH to increase was unexpected and the underlying chemistry needs further investigation. One-way ANOVA showed that all treatments means were significantly different with p-values of sulfuric+SL 0.01 and sulfidic soil <0.001. Analysis of variance for the sulfuric soil was not done as the pH measurements were constant (Figure 1), making the sum of squares too small to perform the analysis. The observed variations had probability values of less than 1 percent.

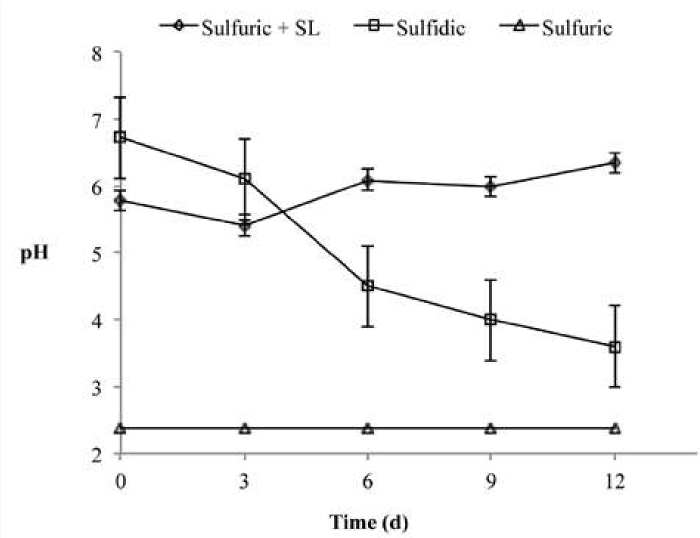


**Figure 1. Comparison of pH changes in sulfuric and sulfidic soil, and sulfuric soil mixed with LC sandy loam. Each point is the mean ± s.e. of 4 replicates.**

*Ameliorative effects of sandy loam on sulfidic soil*

When mixed with sulfidic soil, LC sandy loam was effective in first raising the pH, and then maintaining a stable pH for at least 12 d (Figure 2). To some extent this was unexpected, because the increase in permeability of the sulfidic soil due to the sandy loam would facilitate access of oxygen to the sulfidic components of the soil.

In the absence of sandy loam, the sulfidic soil was rapidly oxidised at 24°C (Figure 2), but the rate of oxidation was relatively slow at 4°C. These effects of temperature may be relevant to how quickly these soils are able to acidify between seasons. All the treatment means were significantly different with p-values of the SL+sulfidic, 24°C, sulfidic, 24°C, and sulfidic 4°C soils all <0.001. The probability values of the observed variations were also less than 1 percent, meaning the variations observed were as expected.

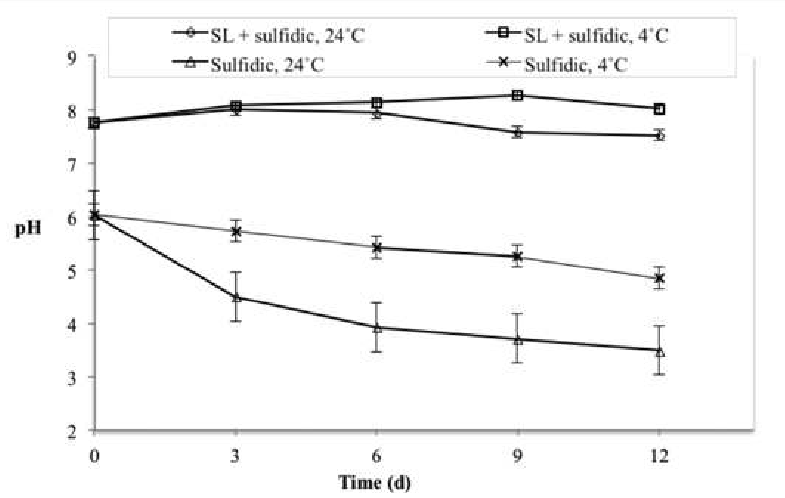


**Figure 2. Ameliorative effects of sandy loam on sulfidic soil. Values are the mean ± s.e. of duplicate (t = 0), 6 (◊, ∆) and 3 (ÿ, X) pHw measurements, respectively.**

**Conclusion**

The results of these two studies for the first time demonstrate that sandy loam has the potential to buffer the acidity of an already sulfuric soil or prevent a sulfidic soil from acidifying. The results further demonstrate that the buffering capacity, when established, can be sustained. The sandy loam used here was relatively alkaline and was able to effectively counteract the acidification caused by oxidation of the soil sulphides, most probably due to the presence of significant amounts of calcium carbonate. Sandy loam is readily available at low cost compared to lime, but perhaps equally importantly, has the capacity to improve the structure of the soil, making it more suitable for growth of plants. The soil organic carbon generated by plant turnover will have the added benefit of increasing the capacity of the soil to buffer acid production from oxidation of sulphides, as well as providing a carbon source for sulphur-reducing bacteria. These findings will have important implications for the management of both sulfuric and sulfidic soils.

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# Dissolved reactive P plays a minor role in P mobility on contrasting soils from the cropping region of south-west Australia

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Phosphorus (P) transfer from agricultural land via runoff, throughflow and leaching is of increasing concern for land managers worldwide. Previous studies suggested that organically bound P comprises a large proportion of soil P reserves in south west Western Australia (WA), but the implications of these findings for the processes and rates of P transport in soils are not known. Two contrasting soil profiles (sand and clay) from cropping land of the upper Fitzgerald River catchment in the south coast region of WA were studied in packed boxes to examine the P forms and fluxes in runoff, throughflow, leachate and soil solution after three rates of P application (equivalent to 0, 20 and 40 kg P/ha). Soil solution was collected at 5, 10 and 15 cm depths with inert soil solution samplers, and leachate was collected at the bottom of the 30 cm of packed boxes. Solutions were analysed for particulate P (PP), dissolved reactive P (DRP) and total dissolved P (TDP) while the dissolved uneactive P (DURP) was calculated by difference (TDP-DRP). Phosphorus transport increased with P rate. In the sand, DRP comprised < 35 % of TP in runoff whereas DURP and PP are about 90% of TP in runoff and leachate. In clay soil, 90 % of P losses in DURP and PP form via thoughflow and leaching while DRP constituted < 33 % of total P lost. The result suggested that major portion of mobilized P appeared to be associated with DURP and PP in runoff and leachate in association with dispersed inorganic colloidal compounds < 0.2 μm.

# Effects of clay addition to compost on nutrient holding capacity of a sandy soil

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Background and aims:
Compost and clay can increase soil nutrient and water holding capacity, but it is not known if a combination of compost and clay is more effective than either amendment by itself. A laboratory incubation experiment was conducted to assess the effect of garden waste compost and a natural clay soil on respiration and nutrient availability of a sandy soil. The sandy soil was amended with compost at a rate equivalent to 50 tons compost ha-1, the compost was combined with 5, 10, 20 and 30% (w/w) clay soil. Additional treatments included a non-amended soil, soil+ compost alone and clay alone with similar rates as those added with the compost. The compost or clay was mixed into the soil, and the soil was packed to 1.22 g cm-1. Soil respiration was measured continuously over 23 days. On days 0, 5 and 23, the soils were leached with 50 ml reverse osmosis water and the following parameters were measured in leachates and the soil: water-soluble organic carbon, available N and P. Clay addition will decrease respiration rate and nutrient leaching compared to compost alone.

# Monday 3 December

## SOIL CARBON

*Presented Posters*

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# Biochar increases soil pH, total C and total N after one season cereal cropping

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Soil organic carbon may be lost, soil health compromised and yields reduced through continuous cropping of sandy soils in rain fed production systems. Biochar and inorganic fertiliser were applied and incorporated in a sandy texture contrast soil in Tasmania, Australia, in May 2011 and planted with *Triticum aestivum* (Revenue) at 92 kg/ha. All treatments (including control) received 120 kg/ha DAP at planting with a separate treatment receiving a further 120 kg/ha DAP prior to sowing (total of 240 kg/ha DAP). Harvesting and soil sampling occurred in January and February 2012 respectively. Biochar applied at 20 t/ha significantly increased soil pH by 0.8 units, total C by 1 % and total N by 0.06 % from the control. Biochar applied at 10 t/ha significantly increased soil pH by 0.5 units and total N by 0.02 % from the control. Grain yield for 20 t/ha and 10 t/ha biochar was 6.80 t/ha and 6.84 t/ha respectively with both significantly higher than control (5.62 t/ha). These results suggest that after only one season of cereal cropping, biochar applied at 20t/ha can significantly increase soil pH, total C and total N and improve yield.

# Rice husk biochar as a soil amendment

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Rice husk is a major organic waste of rice milling. The husk comprises 20-25% of the harvested rice biomass and the global production is more than 130 MT/yr. In Asia, rice husk can be used as a heating fuel using small and large scale carbonisation technologies. These technologies produce a carbonised rice husk residue which can be classed as a biochar according to recent IBI guidelines.  Rice husk biochar (RHB) has been used as a soil amendment for many years in parts of SE Asia. The use of rice husk carbonisation technologies is expanding so that the amount of RHB available for soil amendment is increasing. These waste-energy-biochar systems can also reduce green house gas emissions and sequester atmospheric carbon. This paper reviews the properties of RHB and discuss these in relation to its' capacity to amend soil properties which limit crop production.

RHB is highly granular as the original rice husk structure is largely preserved during the carbonisation process. It has a low bulk density (0.16 kg/L) and high water holding capacity at saturation (2-4 kg/kg) and has the potential to substantially modify soil physical properties when applied at high rates. Available chemical data indicate that it can have a total carbon content of 30-40% if the carbonisation process is controlled to prevent high temperatures (<700 C) and stop after burning. This distinguishes it from rice husk ash which has a carbon content of less than 10%. Whilst alkaline, low temperature RHB usually has a lime equivalence of less than 10%. Field experiments on rice in Indonesia, peanuts in Vietnam have shown yield improvements of the order of 10-15% when inorganic fertilisers are also applied. These yield responses have been associated with increased uptake of P and K from applied fertilisers, using application rates of RHB of 10-20 t/ha. Whilst these rates are high, they are similar to rates used for other organic amendments such as manures which appear have shorter duration of effects. Field results to date suggest there is a need to better understand the effects RHB may have on the interaction between soil water availability and nutrient uptake.

# Effect of application rate of commercial lignite-derived amendments on early-stage growth of *Medicago sativa* and soil health, in acidic soil conditions

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Commercially available lignite-derived amendments, mainly sold as humate preparations, have been promoted as plant growth stimulants leading to higher crop yields. These products are also claimed to improve soil properties such as soil cation exchange capacity, and increase fertiliser efficiency. This study investigated the effect of application rate of lignite-derived amendments on the early-stage growth of a pasture legume, lucerne (*Medicago sativa*), and their effect on soil health in a soil type common to south eastern Australia, in a glasshouse setting. Measurements of root and shoot biomass, microbial biomass C and soil pH were taken after six weeks of growth. Differences in all measured parameters were observed between the amendment application rates, particularly in soil amended with a soluble humate product containing potassium. It was notable that application rates in the order of 9.5 kg/ha C produced an observable effect. An assessment of the effectiveness of lignite-derived amendments on plant growth, as well as their potential to improve the health of an acidic soil will assist farmers in making decisions regarding the use of these products.

# Residue decomposition in salt-affected soils: Effect of residue properties

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It has been shown that salinity decreases microbial activity, the size of the microbial biomass and may reduce microbial diversity community. Fungi have been found to be more sensitive to salinity than bacteria which may have important implication for decomposition of poorly degradable plant residues. If this is true, the relative decomposition rate of lignin-rich residues should decrease more strongly with increasing salinity than that of easily decomposable residues. However, most of previous studies only used one type of plant residue (e.g. cereal or legume shoots) or did not specify the added plant material. In this study, various residue types (legume, wheat, canola, salt bush, kikuyu grass, native grass, eucalyptus leaves, pine needles and sawdust) that differ in lignin content and different C/N ratio will be applied to naturally saline soils with EC1:5 ranging from 0.1, 1, 2, 3, 4, 5 dS m-1. Carbon dioxide emission will be measured continuously, microbial biomass carbon and water soluble carbon will be determined at four time points. The results will provide novel information about residue decomposition rates and therefore carbon sequestration potential of salt-affected soils.

# Decomposition of roots and shoots of perennial grasses and annual barley individually or as mixes

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Little is known about the decomposition rates of shoot and root residues of perennial grasses. Knowledge of the decomposition rates is important to estimate the carbon sequestration potential of the grasses. An incubation experiment was carried out in a sandy clay loam with shoot and root residues of three native perennial grasses (Wallaby grass, Stipa sp. and Kangaroo grass) and the annual grass barley either separately or in mixtures of two residues. Respiration rate was measured over 18 days, and microbial C and available N were measured on days 0 and 18. Decomposition was lower for roots than for shoots and lower for residues of perennial grasses than for barley. Cumulative respiration was positively correlated with water-soluble C in the residues but not with residue C/N. In the mixtures, the measured cumulative respiration was higher than the expected value in five of the nine mixes and was usually found were the differences in cumulative respiration between the individual residues were relatively small. Lower than expected cumulative respiration were found in two of the mixtures where the differences were large. From day 0 to day 18, the microbial biomass C concentration decreased in the soils with shoot residues but increased in the soils with root residues. It is concluded that the lower decomposition rate of residues perennial grasses should favour C sequestration, but that mixing residues of similar decomposition rate may accelerate their decomposition.

# Transport of glyphosate in water-repellent Andisols from Japan and New Zealand

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Soil water repellency (SWR) is an emerging problem associated with climate change that is threatening soil functioning. SWR compromises the soil's ecosystem service to infiltrate rainwater, prevent runoff, and to store plant-available water. The impact of SWR on water infiltration has been extensively researched. We focused on the generally neglected consequences of SWR for the fate of agrichemicals. We hypothesized that the soil's inability to store and buffer water will also degrade the filtering function of soils.

In transport experiments under controlled conditions, we characterized the filtering efficiency of Andisols from Japan and New Zealand. We focused on Andisols because they are unique soils and an important but limited natural resource for both countries. Glyphosate, the most widely used herbicide in the world, was selected as model agrichemical. Tritium was chosen as water tracer. Undisturbed topsoil cores were collected in the middle of summer when the soils were water repellent. We leached a pulse of glyphosate and tritium through the cores under unsaturated conditions without prior equilibrating the soils. To quantify the impact of SWR on the filtering of the soils for glyphosate, we conducted half of the experiments with 0.01 M $CaCl_2$-solution and the other half with ethanol, a fully wetting liquid, as background solution. Comparing fluxes and glyphosate breakthrough curves from the two experiments revealed the impact of SWR on the soils' filtering function.

## Soil incubations: How well are we measuring microbial respiration?

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The two most commonly used approaches for monitoring microbial respiration in soil incubation experiments are instantaneous measurements of $CO_2$ flux and the measurement of accumulated $CO_2$ over a fixed time interval. However, there has been little research testing whether or not the results are indeed similar. We designed a series of experiments to test whether or not these two methodologies produce comparable results. In the first and second experiments, mature compost (250 mg C g-1) was used as a labile substrate to study the dynamics of $CO_2$ accumulation in a sealed incubation vessel. In a third experiment, three soils from contrasting climatic regions in Australia were incubated for 23 days using both measurement methodologies. The initial experiments found: 1) $CO_2$ build up was highly nonlinear inside the incubation vessels with initially high respiration rates (mmol $CO_2$ hr-1) tailing off to significantly lower levels after approximately 12 hours; and 2) cumulative respiration (mmol $CO_2$) was 25% greater when the vessels were refreshed five times during an 100 hour period as compared to when the vessels were sealed for the entire duration. In the third experiment, it was shown that the instantaneous method resulted in greater cumulative respiration and that the respiration rates were sustained at similar levels throughout the incubation period while the accumulation method resulted in decreasing respiration rates. While it is difficult to ascribe which method is more accurate, these findings suggest that the choice of methodology shouldn't be taken lightly and care may be needed in comparing results across studies with different methodologies.

# Assessment of autotrophic and heterotrophic soil respiration in ‘paired’ pine-pasture land uses in Australia

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### Abstract

There is a lack of information on the influence pasture-to-pine land use change (LUC) has on autotrophic (root) and heterotrophic sources of soil respiration, which may have implications for potential soil carbon (C) storage under pine plantations. We employed tree-girdling and shading approaches at two ‘paired’ pasture-pine plantation sites, to permanently and intermittently terminate the recent photosynthate supply to pine and pasture roots, respectively. Soil carbon dioxide ($CO_2$) efflux was measured in control and girdled/shaded plots that represented total and heterotrophic soil respiration, respectively, over 1 year using a static chamber methodology. Root respiration was calculated by subtracting the heterotrophic component from the total soil respiration. The rates of autotrophic and heterotrophic soil respiration varied with the seasonality of soil temperature, moisture and plant C allocation patterns. On a cumulative basis (over 1 year), the total soil respiration was 53–60% lower in the pine plantations than in the adjacent pastures. The rates of autotrophic (from 8 to 109 versus 6 to 373 mg $CO_2$-C $m^{-2}$ $h^{-1}$) and heterotrophic (43 to 207 versus 79 to 597 mg $CO_2$-C $m^{-2}$ $h^{-1}$) soil respiration were 2 to 3 times lower in the pine plantations than the adjacent pastures. These results suggest lower belowground C allocation/ productivity and lower turnover of organic C in soil under pine than pasture. Our findings have relevance for models simulating the pasture-to-pine LUC impacts on soil C dynamics.

### Key Words

Girdling, soil respiration, root respiration, land use change, pasture, pine, heterotrophic respiration.

### Introduction

Afforestation of pastoral lands is a major LUC predicted to continue in Australia as demand for timber and wood products increases, and to achieve Australia’s emission targets through C sequestration in forest biomass (ABARE, 2011). While the above ground C sequestration benefits of afforestation are well recognised, there is little information on the change in soil organic C after afforestation of native pastures with pine plantation. A meta-analysis by Guo and Gifford (2002) suggested a 12–15% decline in soil organic C stock following the establishment of pine plantations on pastoral land. This is possibly due to changes in the quantity and quality of litter input and their turnover rates, the extent of belowground C allocation and the soil microenvironment under pine versus pasture (Guo *et al*. 2008).

Respiration is a major source of $CO_2$ emissions from soil and contributes 98 ±12 Pg $CO_2$-C $year^{-1}$ into the atmosphere (Bond-Lamberty and Thomson 2010) through the combined activity of “autotrophic” (i.e. roots and associated mycorrhizas) and “heterotrophic” organisms in soil (Högberg *et al*. 2001; Singh *et al*. 2011). The extent of heterotrophic respiration is likely to relate to the turnover rate of litter and native soil C. Although the autotrophic respiration may not directly impact soil C turnover, the plant derived C allocated belowground may stimulate (“positive priming”) heterotrophic respiration (Kuzyakov 2002). A better understanding is required of how LUC affects soil respiration and how soil respiratory components respond to the seasonality of environmental and plant C allocation patterns in pine and pasture land uses. This information may help in predicting the impacts of LUC on soil C dynamics and net $CO_2$ emissions.

Various techniques that have been used to separate sources of soil respiration include tree-girdling (ring- barking), shading and clipping of grasses, isotopic techniques and root exclusion and trenching (Craine *et al*. 1999; Hanson *et al*. 2000; Högberg *et al*. 2001). The advantages and disadvantages of these techniques are discussed elsewhere (Hanson *et al*. 2000). In the present study, we partitioned autotrophic and heterotrophic sources of soil respiration in two ‘paired’ pasture-pine plantation sites using tree-girdling in pine (Högberg *et al*. 2001) and intermittent shading in pastures (Craine *et al.* 1999). These techniques terminate the flow of current photosynthates from canopy to roots without impeding the translocation of water from roots to canopy. The objective of this study was to measure the impact of afforestation of pastoral lands with pine (*Pinus radiata*) on soil respiration and its sources, thereby providing insights into the impact of LUC on soil C storage.

## Methods

For the present study, two 'paired' pasture-pine sites (one in Glenwood State Forest on the north eastern side of Mount Canobolas and the second in Macquarie Woods State Forest in Vittoria) were selected in the central tablelands of New South Wales, Australia. Both sites are characterised by contrasting rainfall regimes (averaged over 40 years): ~1000 mm/annum at Glenwood and ~700 mm/annum at Macquarie Woods. The mean annual air temperatures for the Glenwood and Macquarie Woods sites over the duration of the study were 17.9 and 17.5 °C, respectively. The pine plantation at the Glenwood (GPS (55H UTM) E 679942 N 6311561) site was established in 1987, with an average diameter at breast height of 35.3 cm and a density of 521 stems per hectare 23 years after establishment (i.e. in 2010). The pine plantation at the Macquarie Woods (GPS (55H UTM) E 714022, N 6300045) site was established in 1988, with an average diameter at breast height of 31.7 cm and a density of 572 stems per hectare 22 years after establishment. The adjacent pastures at both sites were established in *circa* 1950s and 1970s, comprising pasture species such as *Trifolium subterraneum*, *Dactylis glomerata* and *Phalaris aquatica.* There was no grazing in either plantation site due to the absence of understorey vegetation. The pasture at the Glenwood site was grazed by cattle and at the Macquarie Woods site, the pasture was grazed by sheep.

At both sites, three paired circular plots (7 m diameter) containing 16-18 trees in the plantations, and three paired 2 $m^2$ (1 m wide × 2 m long) plots in the adjacent pastures, were established. Each plot was trenched to a depth of 1 m in the forests and 0.5 m in the pastures with the trencher then lined with a double layer of heavy duty plastic (2 mm thick) to prevent root penetration from external plants into the experimental plots. In the pine plantations, one plot from each pair was randomly selected and all trees within this plot were girdled at 1.3 m above ground, by complete removal of a 30 cm wide section of bark from around the trunk back to but not including the cambium (Högberg *et al*. 2001; Bhupinderpal-Singh *et al.* 2003). These plots are hereafter referred to as girdled plots. The three plots of non-girdled trees at each site were used as controls. Likewise, in the adjacent pastures, three randomly selected plots were intermittently enclosed (shaded) for 2 to 3 days before sampling using dark insulated plastic tubs. The girdling and shading treatments permanently and intermittently terminate the supply of recent photosynthate from the aboveground biomass to pine and pasture roots, respectively (Craine *et al*. 1999; Högberg *et al*. 2001). The enclosures from the shaded plots were removed between sampling times to expose the plots to normal grazing management.

*In situ* soil surface $CO_2$ efflux was measured on a regular basis between October 2010 and September 2011 using Vaisala GMP343 diffusion $CO_2$ probes mounted in static chambers (~6 L; r = 11.6 cm). The soil surface $CO_2$ efflux rates were calculated using linear rate of change of $[CO_2]$ in the chambers between 2 and 10 minutes after closure of static chambers. The $[CO_2]$ measurements during the first 2 minutes were discarded to allow $[CO_2]$ to stabilise. Concurrently, soil temperature at 5 cm depth and gravimetric soil moisture content (0–10 cm) were also measured in all the treatments at each sampling time point. The soil temperatures in the pine plantations and adjacent pastures at 5 cm depth across both sites ranged between 5–17 °C and 6–24 °C, respectively, at different times during the experimental period (October 2010-October 2011). Girdling was found to have no influence on soil temperature and moisture content whereas shading decreased soil temperature by an average of 2–3°C relative to the unshaded pasture plots in summer months. The cumulative $CO_2$ efflux was calculated by assuming linear efflux rates between sequential sampling dates and subsequent stepwise addition of total $CO_2$ efflux over 1 year.

Topsoil (0-10 cm of the mineral soil profile) samples were collected from both paired sites by a stratified random sampling scheme along a matching transect in both land uses and analysed for total and the relatively labile particulate (>53 µm) organic C by thermal conductivity detection after dry combustion using an Elementar CNS analyser. The total and particulate organic C contents of the soil from the Glenwood site were 53 and 18 g $kg^{-1}$ soil, respectively, in the pine plantation versus 70 and 21 g $kg^{-1}$ soil, in the adjacent pasture. At the Macquarie Woods site, the total and particulate organic C contents were 18 and 7 g $kg^{-1}$ soil, respectively, in the pine plantation versus 21 and 6 g $kg^{-1}$ soil, in the adjacent pasture.

## Results and Discussion

*Land use change impact on total soil respiration*

At the Glenwood site the rate of soil respiration in the pine plantation ranged from 57 to 277 mg $CO_2$-C $m^{-2}$ $h^{-1}$ , while in the adjacent pasture the soil respiration rate ranged from 111 to 688 mg $CO_2$-C $m^{-2}$ $h^{-1}$. At the Macquarie Woods site the soil respiration rate in the pine plantation ranged from 79 to 290 mg $CO_2$-C $m^{-2}$ $h^{-1}$, compared to the adjacent pasture in which the respiration rate ranged from 115 to 855 mg $CO_2$-C $m^{-2}$ $h^{-1}$ (data not shown). The maximum rates of soil respiration were observed in the summer months with minima observed in the winter months at both the low and high rainfall locations. The cumulative soil respiration under the pine plantation at each site was ~60% less than the adjacent pastures (Figure 1). Our results are consistent with the findings of Kellman *et al.* (2007) who observed a soil respiration rate 31-59% lower in a coniferous forest as compared to an adjacent pasture.

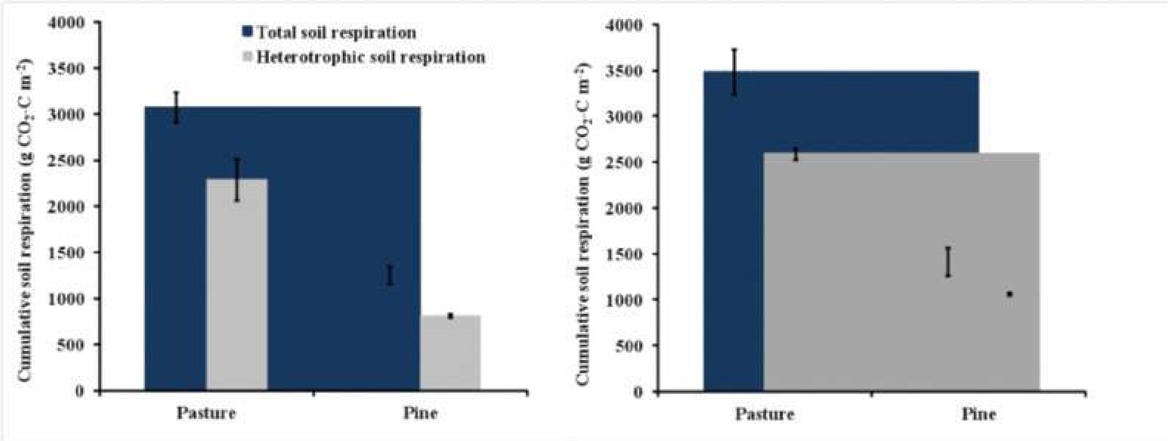


**Figure 1. Cumulative soil respiration (total and heterotrophic) over 1 year (October 2010 to October 2011) in paired pine–pasture land uses in Glenwood State Forest (left panel) and Macquarie Woods State Forest (right panel) in the central tablelands of New South Wales, Australia. Error bars are 2 standard errors of the mean (n =3).**

*Girdling and shading impacts on autotrophic and heterotrophic soil respiration in pine versus pasture*

Two weeks after girdling, soil respiration rates from the girdled plots were 30% lower than the non-girdled plots at both sites (data not shown). These results show a direct relationship between autotrophic respiration and the supply of current photosynthates from the forest canopy to root biomass (Högberg *et al*. 2001; Bhupinderpal-Singh *et al*. 2003). Soil respiration rates decreased further and became 38–47% lower in the girdled plots (c.f. non-girdled plots) 4 months after girdling. Intermittent shading of pasture plots effectively decreased soil respiration by a maximum of 55% at Glenwood and 42% at Macquarie Woods relative to unshaded plots. The seasonal patterns of autotrophic and heterotrophic soil respiration were similar to total soil respiration at both sites. The rates of both autotrophic (from 8 to 109 versus 6 to 373 mg $CO_2$-C $m^{-2}$ $h^{-1}$) and heterotrophic (43 to 207 versus 79 to 597 mg $CO_2$-C $m^{-2}$ $h^{-1}$) soil respiration were 2 to 3 times lower in the pine plantations than the adjacent pastures. Lower autotrophic and/or heterotrophic soil respiration in the pine plantation (*c.f.* adjacent pastures) may be attributed to: (a) lower soil temperature due to denser canopy and thicker litter layer; (b) lower amounts of total and/or particulate organic C, and possibly lower turnover rates of organic C in the soil; and (c) lower belowground C allocation and/or belowground biomass productivity (Guo & Gifford 2002; Högberg *et al*. 2001).

## Conclusions

Autotrophic and heterotrophic sources of soil respiration were lower in pine plantations than adjacent pastures. Our results suggest that a reduction in belowground C allocation and/or productivity and slower turnover rates of native organic C are potential causes for lower soil C contents under pine forests as compared to adjacent pastures.

**Acknowledgement**

We are grateful to Meagan Jenkins, Yunying Fang, Vivien de R. de Courcelles, Daojaru Katrot, and Waqaar Ahmad (The University of Sydney) and Kamaljit Kaur (NSW Department of Primary Industries) for their help in establishing the experiment. David Giles and Simranjeet Singh (NSW Department of Primary Industries) provided assistance with the laboratory and field work. We also thank William Shearman, Dael O'Neill (Forests NSW) and Clair Kerwick of the pasture lands for providing site information and access to the sites to conduct the experiment. This project was supported by funding from the Australian Government Department of Agriculture, Fisheries and Forestry under its Forest Industries Climate Change Research Fund program.

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# Changes in soil carbon of pastures after afforestation with mixed species

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Afforestation of agricultural land is increasingly recognised as a viable means of mitigating against climate change. However, the effects of afforestation on soil carbon stocks, and the time scales over which they change, remain unclear. Here we present results from a number of projects investigating the impact of afforestation of pastures on soil carbon. First, we conducted a meta–analysis of published data on the effects of afforestation of pastures on soil carbon and nitrogen stocks in a Mediterranean climate. Results indicate that more than 30 years will be needed for soil carbon stocks to reach those of remnant forests. Second, we surveyed soil carbon and nitrogen at 40 sites spanning a chronosequence (5-45 years old) of existing plantings, and their adjacent pastures to account for differences in soil type and land-use history among sites. Whereas changes in total soil carbon were not yet evident, a shift in the C:N ratio of the soils, and an increase in the density of plant litter, was observed. Third, we undertook a detailed analysis of soil carbon stocks at a sub-set of our field sites to determine the optimum number of soil samples needed to provide an accurate estimate of soil carbon in mixed species plantings. The results presented here will be discussed in the context of the role of revegetation activities to sequester carbon in the soil.

# Soil carbon and landscape function: The effect of grazing on soil carbon stabilisation mechanisms

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The majority of Australian grazing lands have been degraded since European settlement, resulting in loss of soil carbon and reduced capacity to provide ecosystem services, from primary productivity and climate moderation to social and aesthetic values. Land degradation has also reduced resilience, making grazing lands more vulnerable, and less able to adapt, to the impacts of climate change. With an already variable climate, much of the country is projected to become hotter and drier, with less rainfall, higher temperatures and higher evapotranspiration likely to shift some currently arable land into rangelands.

Restoring degraded land and building soil carbon levels offers multiple co-benefits – increasing primary productivity and water use efficiency while potentially providing a climate change mitigation option and improving adaptive capacity.

While inappropriate grazing practices have contributed to historical loss of soil carbon, there is strong anecdotal evidence of innovative grazing practices regenerating degraded land and building soil carbon levels. Grazing practices are highly variable and operate across a wide range of landscapes and climates, making it important to understand the effect of grazing on soil processes and properties as mechanisms for soil carbon stabilisation.

This paper presents the results of a grazing experiment in south-eastern Australia. A paired site is used to compare the effects of continuous grazing and rotational grazing using high stocking density, short stocking periods and long rest periods. A range of methods have been used to assess vegetation, surface and soil properties and processes and the implications for soil carbon sequestration.

# Use of compound specific stable isotope techniques to study soil organic matter turnover

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Soils store a greater pool of carbon than the atmosphere and the terrestrial biosphere combined. Therefore, understanding the turnover and stabilization of soil organic matter (SOM) is of great importance in understanding the global C cycle. Compound specific stable isotope analysis of microbial and plant biomarkers offer a little-explored opportunity to link microbial activity to biogeochemical processes in soils, thus providing a better understanding of the role of microbial processes in soil organic matter turnover and stabilization. These techniques will be applied to select Australian soils to determine the rate of assimilation of SOM pools of varying chemical stability into microbial biomass and the effect of temperature upon SOM assimilation and mineralization. Previous work along a chronosequence of C3 woody plant encroachment into C4-dominanted grasslands in southern Texas, USA, has highlighted the utility of compound-specific stable isotope analyses of microbial phospholipids. Results from this study have shown that the assimilation of newer, C3-derived carbon in woody areas is predominately dictated by fungi and gram-negative bacteria. This may be due to changes in the chemical stability of C3 inputs and/or due to the development of the rhizosphere. This study will apply this technique for the first time to Australian ecosystems and as such will be able to identify the members of the microbial community responsible for the cycling of particular components of SOM under different soil types, land uses, and temperature regimes.

# Monday 3 December

## SOIL FERTILITY

*Presented Posters*

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# Biochar and clay amendment of sands: Effects on P release, leaching and availability

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Managing phosphorus (P) supply on sands remains a challenge because of the propensity of P to leach while reducing P rates may lead to P deficiency in crops. Biochar and clay are two amendments with potential to improve the retention and availability of P to crops. This study investigated the effect of biochar (produced from wheat straw (WS) or chicken manure (CM)) and clay (kaolin or clay-rich subsoil) on P in soil solution and leachate and on P uptake by wheat on a grey sand from the south coast of West Australia. We hypothesized that P leaching will decrease with the addition of biochar and clay together to sands. More P was released during desorption by WS biochar than by the CM biochar. After 5 successive desorption steps, solution P concentration was 10 mg/L with CM biochar but declined to only 2 mg/L with WS biochar. This suggested that P in WS & CM biochar is readily soluble with more rapid release from WS biochar. Adding either kaolin or subsoil clay (50 t of clay/ha) with WS biochar or CM biochar (10t/ha) decreased the amount of P leached with or without P fertiliser added, however the decline in P leached and in soil solution P concentration was most pronounced with subsoil clay addition. In summary, subsoil clay had a greater effect than kaolin in reducing soluble P in soil solution and in leachate particularly with CM biochar but the soil solution P concentrations still remained above 0.2 mg P/L, and hence was maintained at concentrations adequate for plant growth.

# Growth of aerobic rice in the presence of biochar as soil amendment: Short-term effects in a clayey Rhodic Ferralsol in the Brazilian savannah (Cerrado)

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### Abstract

Increasing yields in aerobic rice systems (ARS) is a challenge in the Brazilian savannah (BS), where rice is grown under unfavourable conditions characterised by well drained and low fertility soils. Management options that could increase soil water availability and nitrogen (N) use efficiency would probably lead to higher grain yields in ARS. One promising option under consideration is the use of 'biochar', a by-product of charcoal made of hardwood, as a soil amendment. Biochar is high in resistant (pyrogenic) carbon (70 to 80% of the material), which influences some processes in soil, depending on the amount applied and its interaction with soil. Yet there are no conclusive field studies that quantify the effect of hardwood biochar application on grain yield of ARS in the BS. Here, we report single season effects of biochar application coupled with N fertilisation on aerobic rice growth and grain yield in a clayey Rhodic Ferralsol in the BS. At 72 days after sowing, leaf area index and total shoot dry matter of aerobic rice was negatively related to biochar rates above 16 Mg/ha. This effect might be related to changes in soil properties due to biochar application, such as increased soil nitrate availability. We found that biochar applications did not influence grain yield. The effect of N fertilisation on yield followed a quadratic pattern, with an optimal N rate of around 46 kg/ha to achieve a grain yield above 3 Mg/ha, regardless of biochar application. The trends will guide future research.

**Key Words**

*Oryza sativa*, yield, pyrogenic carbon, nitrogen, Oxisol.

### Introduction

Aerobic rice-based cropping systems (ARS) are typically rain fed, where rice is grown on well drained soils. Less demand for labor, mechanization and water are important advantages of this system. Disadvantages of ARS include high weed pressure, water limitation due to rainfall variability, and low soil nutrient availability (Fageria 2001). A suggested strategy to improve soil fertility is the application of 'biochar'as soil amendment. Pieces of charcoal smaller than 8 mm are a by-product of charcoal production and are readily available in the Brazilian savannah (BS). An otherwise worthless by-product, which we refer to as 'biochar', can be recycled as a soil amendment. Biochar is high in resistant (pyrogenic) carbon (70 to 80% of the material). Adding large amounts of biochar might increase soil mineralization or immobilization of nitrogen (N) immediately after biochar application in soil. Given the potential for co-limitations, the interactions between N fertilisation and biochar application are particularly relevant. Some of the effects of hardwood biochar on soil chemical properties in a Ferralsol have been reported (Lehmann *et al*. 2003, Steiner *et al*. 2007). However, there are still no conclusive field studies that quantify the effect of hardwood biochar application on grain yield of ARS in the BS. The objective of this paper was to report on the short-term effects of biochar application and its interaction with N fertilisation on aerobic rice growth and grain yield in a clayey Rhodic Ferralsol in the BS.

### Methods

*Study location and experimental design*

An experimental field was established on June 9, 2009, in a clayey Rhodic Ferralsol at the National Rice and Beans Research Centre ('Embrapa Arroz e Feijão'), in Santo Antonio de Goiás, Goiás, Brazil (16°29'17 ''S and 49°17'57 ''W). Experimental plots were arranged in four replications, with nitrogen

(0, 30, 60 and 90 kg/ha) and biochar (0, 8, 16, 32 Mg/ha) each applied in four rates. Each plot had an area of 28 $m^2$. Plots were located under a centre pivot irrigator; a total of 78 mm of water via irrigation was applied throughout the growing season (from November 2009 to February 2010). Total rainfall during the growing season was 855 mm, and average temperature was 24°C. Biochar was incorporated into soil at 0-20 cm layer using a harrow, six months prior to sowing rice (June 9, 2009). Following a crop of common beans (*Phaseolus vulgaris*), aerobic rice, cultivar 'Primavera', was sown on November 3, 2009, in seven 10 m-rows in each plot with row spacing of 40 cm, at a plant density of 100 seeds/m. The crop was harvested on February 22, 2010. Rates of 120 kg/ha of P and 60 kg/ha of K were applied to all plots and incorporated in soil together with the rice seeds. Rates of nitrogen (urea) were divided in two applications: 50% at sowing incorporated in soil together with rice seeds, and 50% at 33 days after sowing, as topdressing.

*Biochar properties*

The source of the biochar was charcoal produced from plantation timber (*Eucalyptus* sp.) by slow pyrolysis at around 400-550 °C. It was milled to pass a 2 mm sieve before soil application. Chemical analysis showed that 1 Mg of biochar contained around 770 kg of carbon, 3 kg of nitrogen, 0.005 kg of phosphorus, 0.06 kg of potassium, 0.6 kg of calcium, and 0.3 kg of magnesium. This accounts for 81% of the biochar mass.

*Plant and soil sampling and measurements*

Yield (weight of grains dried to 13% moisture) was determined from an area of 2.4 $m^2$ (two rows of three meters) at 110 days after sowing (DAS). Harvest index was calculated as the ratio between weight of dried grains and weight of total shoot dry matter (including grains) at 110 DAS. Spikelet fertility was calculated as the ratio between the number of full grains and total number of grains per spikelet. At 72 DAS, the leaf area index (LAI) and total shoot dry matter (TDM) were determined using plants collected from a 50 cm row (area of 0.2 $m^2$) in each plot. Leaf area of ten tillers was measured using Li-Cor (Inc. Lincoln, NE, EUA). The total number of tillers in 50 cm of row was counted and used to calculate LAI. All plants in 50 cm of row were dried in an oven at 75°C for 48 hours and weighed to determine TDM. Soil moisture, ammonium and nitrate availability were measured frequently throughout the growing season in the 0-10 cm soil layer. Three sub-samples were collected to get a 30 g soil sample in each plot. Around 10 g of soil was weighed before and after drying in an oven for 24 hours at 45ºC. Soil moisture ($cm^3/cm^3$) was calculated by considering the soil bulk density ($g/cm^3$) determined in each plot. Nitrate and ammonium (mg/kg) were extracted from soil samples by shaking 20 g of soil with 60 mL of 1M KCl for 60 minutes (Mulvaney 1996).

*Data analysis*

We adopted generalized linear mixed models that account for spatial autocorrelation among plots, by including rows and columns (coordinates of plots) as random effects. Linear and quadratic functions were fitted to investigate effects of nitrogen (N), biochar (char) and its interactions (N*char), here treated as fixed effects. Analyses were performed using the SAS/STAT Mixed procedure (SAS Institute Inc. 2010).

## Results

*Effects on plant growth and soil properties*

At 72 days after sowing (DAS) rice, rates of biochar (char) above 16 Mg/ha tended to negatively affect leaf area index (LAI) and total shoot dry matter (TDM) (Figure 1). The influence of biochar on LAI and TDM showed a quadratic trend. Predicted LAI ($m^2/m^2$) varied from 4.4 (16 Mg char) to 3.6 (32 Mg char), whereas predicted TDM (Mg/ha) varied from 6 (16 Mg char; 90 kg N) to 3 (32 Mg char; 0 kg N).

The fitted function for the average soil nitrate availability ($Y = 0.5002$ char $+ 0.3357$ N $+ 43.85$) showed that soil nitrate availability (mg/kg) in the 0-10 cm soil layer increased linearly ($p \leq 0.0001$) with biochar application. On the other hand, there was no effect of biochar on the average soil ammonium availability, which was linearly increased ($p \leq 0.05$) with N application ($Y = 0.06349$ N $+ 16.83$). Throughout the growing season of aerobic rice, soil moisture was up to 0.03 $cm^3/cm^3$ higher in treatments with 16 and 32 Mg/ha biochar than in treatments without biochar application (Figure 2).

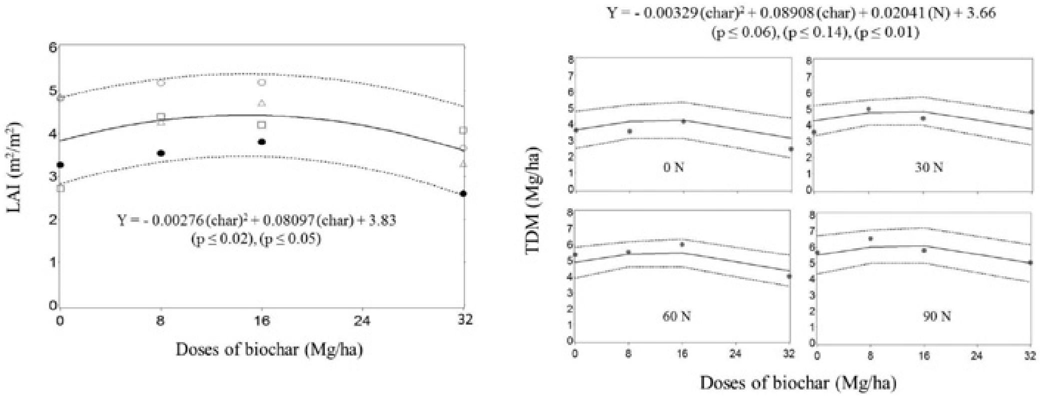


**Figure 1. (*Right*) influence of biochar rates (0, 8, 16, 32 Mg/ha) on leaf area index (LAI) of aerobic rice cultivated under different nitrogen rates (●0, □30, △60, ○90 kg/ha); and (*left*) influence of biochar (char) and nitrogen (N) rates on total shoot dry matter (TDM) of aerobic rice in a clayey Rhodic Ferralsol in the Brazilian savannah, at 72 days after sowing rice, growing season 2009/2010. Black lines indicate the fitted function. Dotted lines correspond to respective 95% confidence bands. Values between brackets are nominal significance values corresponding to hypothesis tests on linear or quadratic effects.**

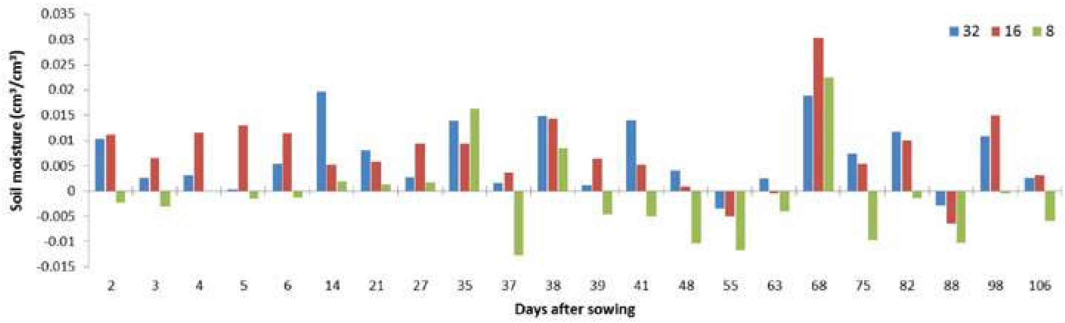


**Figure 2. Difference in soil moisture (cm$^3$/cm$^3$) between treatments with biochar (8, 16, 32 Mg/ha) and control (without biochar) in a clayey Rhodic Ferralsol in the Brazilian savannah, throughout a growing season of aerobic rice, summer 2009/2010.**

*Effects on grain yield and yield components*

There was no effect of biochar on grain yield of aerobic rice. Rates of nitrogen (N) above 60 kg/ha tended to negatively affect grain yield (Figure 3-*right*), even though the higher the rate of N, the higher the total shoot dry matter (TDM) at 72 days after sowing (Figure 1-*left*). Based on the fitted function for grain yield, the optimal rate of N would be around 46 kg/ha to achieve a grain yield above 3 Mg/ha, regardless of biochar application. Predicted grain yield (Mg/ha) varied from 3 (60 kg N) to 2 (0 kg N).

The influence of N on spikelet fertility (SF) also followed a quadratic trend ($Y = -0.00003\ N^2 + 0.00201\ N + 0.72$). Rates of N above 60 kg/ha tended ($P \leq 0.10$) to negatively affect SF. Predicted SF varied from 0.75 (30 kg N) to 0.65 (90 kg N).

There was, however, an interaction between biochar and N on harvest index (HI), meaning that the effect of biochar was dependent on the rate of N applied. The higher the rate of biochar applied, the more N was required to achieve the same HI (Figure 3-*left*). Predicted HI (Mg/Mg) varied from 0.44 (0 Mg char; 0 kg N) to 0.37 (32 Mg char; 0 kg N).

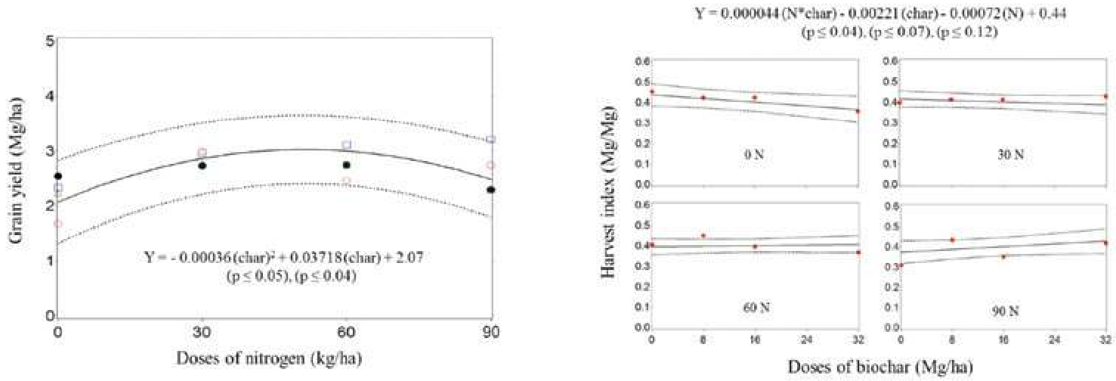


**Figure 3. (*Right*) influence nitrogen rates (0, 30, 60, 90 kg/ha) on grain yield of aerobic rice cultivated under different biochar rates (●0, □8, △16, ○32 Mg/ha); and (*left*) influence biochar (char) and nitrogen (N) rates and its interaction (N*char) on harvest index of aerobic rice in a clayey Rhodic Ferralsol in the Brazilian savannah, growing season 2009/2010. Black lines indicate the fitted function. Dotted lines correspond to respective 95% confidence bands. Values between brackets are nominal significance values corresponding to hypothesis tests on linear or quadratic effects**.

## Conclusions

At 72 days after sowing aerobic rice, leaf area index (LAI) and total shoot dry matter (TDM) were negatively affected by biochar rates above16 Mg/ha. The negative effect of biochar on LAI and TDM might be related to changes in soil properties due biochar application, such as increased soil nitrate availability. Since nitrate can be easily lost via $N_2O$ emissions under anaerobic conditions, and because biochar rates above 16 Mg/ha led to high soil moisture throughout the growing season of rice, the negative effect of biochar on LAI and TDM might be related to decreased soil N availability. There was no effect of biochar application on grain yield. The effect of N fertilisation on yield followed a quadratic pattern. The optimal rate of N was around 46 kg/ha to achieve a grain yield above 3 Mg/ha. The negative effect of N rates above 60 kg/ha on grain yield might be related to the lower carbohydrate remobilisation capacity of modern aerobic rice cultivars, such as ‘Primavera’, than traditional upland varieties (Pinheiro *et al*. 2006). Increases in TDM beyond a certain threshold could result in yield penalties. The consistently lower grain yield, harvest index, TDM and LAI of the high biochar treatment (32 Mg/ha), without N fertilisation, require further investigation.We will study effects of biochar and N and its interactions in following growing seasons of aerobic rice in the same experimental field and in areas under less favorable conditions (sandy soil, only rain fed) in the Brazilian savannah.


## Acknowledgement

The study was funded by the Brazilian Agriculture Research Corporation - Embrapa (Project No. 03100109).


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# Effect of repeated biosolids application on soil chemical properties, tree nutrition and growth of radiata pine

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**Abstract**

Biosolids, rich in organic carbon and nutrients, are commonly used as supplemental fertilisers and soil amendments on crop land, and *preferably* on forest land in New Zealand. Since the mid-1990s, a *Pinus radiata* plantation growing on a sandy, low fertility soil at Rabbit Island in Nelson, has received aerobically digested liquid biosolids. A research trial was established on the site in 1997 to investigate the long-term effects of biosolids application on soil quality, tree nutrition and growth. Biosolids have been applied to the trial site every three years since 1997, at three application rates: 0 (Control), 300 (Standard) and 600 kg N/ha (High). Tree nutrition status and growth are monitored annually. Soil samples were taken in May 2010 to assess the effects of the repeated biosolids application on soil chemical properties. Biosolids application significantly increased foliar N concentration and tree stem volume growth. Both the Standard and High treatments significantly increased soil (0-0.25 m) total C, N, and P, Olsen P and CEC but reduced soil pH. The High biosolids treatment also increased concentrations of soil total Cd, Cr, Cu and Pb at 0.25-0.50 m, but they were considered very low for soil. Our results indicate that repeated application of biosolids to a plantation forest on a poor site could significantly improve soil fertility, tree nutrition and site productivity. Biosolids-derived heavy metals were strongly retained in the litter surface *soil*. The long-term fate of biosolids-derived heavy metals needs to be further monitored.

**Key Words**
Biosolids, soil quality, radiata pine, tree nutrition and growth

## Introduction

Beneficial use of biosolids as a supplemental fertiliser and soil amendment is one of the most common options for biosolids management (Magesan and Wang 2003). In New Zealand, application of biosolids on forest land is preferred than on agricultural land because it can reduce the risk of contaminants entering the human food chain and it can also increase tree growth and subsequent economic returns (Kimberley *et al.* 2004; Wang *et al*. 2006). Treated biosolids from the Nelson Regional Sewage Treatment Plant have been applied to a 1000-ha *Pinus radiata* D. Don forest plantation at Rabbit Island near Nelson City since 1996. A long-term research trial was established on the site in 1997 to monitor the environmental effects of the repeated application of biosolids on the plantation, and to determine sustainable application rates. Since then tree nutrition, growth and wood properties have been assessed along with a number of environmental variables, such as groundwater quality (Wang *et al.* 2004; 2006). The objective of this study was to investigate the effects of repeated applications of biosolids on tree nutrition and growth over time, and soil chemical properties at 13 years after establishment of the trial.

## Materials and Methods

The research trial was established in a 6-year-old *P. radiata* stand in the Rabbit Island plantation in 1997. The soil at the trial site is classified as a sandy raw soil (Hewitt 1998), which is deficient in nitrogen (N) but provides free rooting access to shallow groundwater 2.0 to 4.2 m below the surface (Wang *et al*. 2004). The trial was a split-plot, randomised block design with four replicates. Three biosolids treatments, applied in main-plots, were (1) Control (no biosolids), (2) Standard treatment (300 kg N $ha^{-1}$), and (3) High treatment (600 kg N $ha^{-1}$). Each main-plot contained three tree stocking rates (subplots) at 300, 450 and 600 stems/ha. There were 36 subplots in total. Each subplot measured 25 m × 25 m, plus 5 m buffer zones. Treated biosolids from the Nelson regional wastewater treatment plant were applied in 1997, 2000, 2003, 2006 and 2009 at the same rates to the same plots. The biosolids contained high N concentrations (approximately 10% N with about 40% in ammonium form). Details about properties of the biosolids applied and method of application were given by Wang *et al.* (2004).

Tree height and diameter at breast height were measured annually for all plots from ages 7 to 19 (years). Stocking, mean top height, basal area and stem volume were estimated for each subplot as described previously (Wang *et al.* 2006). To assess tree nutrition status changes due to biosolids application, current-year foliage samples were collected annually from the top third of crowns of selected trees in each plot in summer since 1998 (7 years old). The impact of biosolids applications on soil chemical properties was assessed from samples taken from the forest floor litter layer, topsoil (0–0.25 m) and subsoil (0.25–0.5 m) in May 2010. Samples were taken from all subplots within each biosolids treatment main-plot and bulked, resulting in four replicate samples per biosolids treatment. The foliage and litter were oven-dried (70°C) and ground for chemical analysis. Soil samples were air-dried and ground to pass a 2-mm sieve. Soil pH was measured at a soil:water ratio of 1:2. Total C, N and S in soil, tree foliage and litter samples were determined by dry combustion using a LECO CNS 2000 Analyzer. Concentrations of soil exchangeable Ca, Mg, K, and Na were measured using the ammonium acetate method (Blakemore *et al.* 1987). Extractable soil P was determined using the Olsen P method. Acid digestion was used to extract heavy metals in biosolids and soil samples (ASTM International 1999). Flame atomic absorption spectrometry was used to determine concentrations of As, Cd, Cr, Cu, Pb, Ni, and Zn in the acid digestion samples. Mercury was analysed using cold vapour atomic absorption spectrometry (APHA 1995). Foliage and litter samples were digested with concentrated $HNO_3/H_2O_2$, and concentration of nutrients and heavy metals in the digest were determined using inductively coupled plasma optical emission spectrometry.

Analysis of variance (ANOVA) and least significant difference (LSD) tests were conducted to determine the statistical significance of the biosolids treatment effects on tree nutrition and growth over time and soil properties at 13 years after the first application of biosolids using the SAS/STAT Version 9 GLM procedure.

## Results and Discussion

*Effect of biosolids application on forest floor litter and soil chemical properties*

Both the standard and high biosolids application significantly ($P < 0.05$) increased N, P and S concentrations but reduced Mn concentration and C/N ratios in forest floor litter (Table 1). The Standard treatment also significantly ($P < 0.05$) increased B concentration in the litter when compared to the control. These changes in litter elemental concentration could be attributed to the initial litter chemical composition and retention of biosolids-derived elements. For example, significantly lower Mn concentrations in foliage samples were observed (Wang *et al.* 2004). The results indicate that the biosolids application improved litter quality as a result of improved tree nutrition status (see Figure 1).

**Table 1: Effect of biosolids application on litter chemical properties (sampled in May 2010)***

| Treatment | C | N | P | K | Ca | Mg | S | Na | C/N | B | Mn | Fe | Al |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | % | | | | | | mg kg$^{-1}$ | | |
| Control | 37 a | 0.91 b | 0.05 b | 0.11 a | 0.46 a | 0.23 a | 0.09 b | 0.034 a | 40 a | 12 b | 544 a | 4039 a | 3810 a |
| Standard | 38 a | 1.20 a | 0.07 a | 0.10 a | 0.49 a | 0.23 a | 0.14 a | 0.043 a | 32 b | 14 a | 294 b | 4200 a | 4069 a |
| High | 39 a | 1.19 a | 0.07 a | 0.10 a | 0.49 a | 0.22 a | 0.15 a | 0.037 a | 33 b | 13 ab | 335 b | 3910 a | 3809 a |

*Values within a column followed by different letters differ significantly at $P = 0.05$ (LSD test)

In the top soil layer (0–0.25 m), significantly ($P < 0.05$) higher total C, N and P, Olsen P and CEC but lower pH were found for both the Standard and High treatments (Table 2). In the subsoil layer (0.25–0.5 m), the High treatment significantly increased total C, N and P and Olsen P and reduced soil pH while the Standard treatment significantly increased total C and Olsen-P and reduced soil pH when compared to the control (Table 2). However, biosolids application had no significant effect on concentrations of exchangeable Ca, Mg, K, and Na in either soil layer. The results indicate that biosolids application, especially the High treatment, not only resulted in accumulation of C, N and P in the topsoil, but also caused some movement of these nutrients down the soil profile. This agrees with findings of Lu and O'Connor (2001) who reported that biosolids-derived P may be susceptible to leaching through sandy soils, due to the low soil P-sorbing capacity. Biosolids application significantly ($P < 0.05$) reduced soil C/N ratios in the 0–0.25 m layer but increased the ratios in the 0.25-0.5 m layer. This could be explained by the relatively greater downward movement of biosolids-derived C than N. The lower pH in both the Standard and High treatments could result from the greater nitrification of biosolids-derived N.

Biosolids applications significantly ($P < 0.05$) increased total concentrations of Cd, Cr, Cu and Zn in the litter (Table 3).This implies that a large proportion of the biosolids-derived metals were strongly retained in the litter layer. High metal retention capacity by forest litter was also reported by McLaren *et al.* (2007), who found that the concentration of heavy metals in the litter layer was greatly elevated even a few years after application of biosolids.

**Table 2. Effect of biosolids application on soil chemical properties (sampled in May 2010)***

| Depth | Treatment | pH | Total C | Total N | Total P | C/N | Olsen P | K | Ca | Mg | Na | CEC |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | % | | | | mg $kg^{-1}$ | cmol $kg^{-1}$ | | | | |
| | Control | 5.4 a | 0.63 b | 0.03 b | 0.024 c | 22 a | 21 c | 0.15 a | 1.05 a | 1.08 a | 0.08 a | 4.80 b |
| 0-0.25m | Standard | 5.0 b | 0.93 a | 0.05 a | 0.029 b | 18 b | 39 b | 0.13 a | 1.23 a | 1.09 a | 0.11 a | 5.46 a |
| | High | 4.8 b | 1.03 a | 0.06 a | 0.032 a | 17 b | 54 a | 0.15 a | 1.26 a | 1.08 a | 0.09 a | 5.41 a |
| | Control | 5.7 a | 0.25 c | 0.02 b | 0.026 b | 14 b | 15 c | 0.12 a | 0.77 a | 1.02 a | 0.07 a | 4.14 a |
| 0.25-0.5m | Standard | 5.3 b | 0.43 b | 0.03 b | 0.026 b | 16 ab | 24 b | 0.11 a | 0.96 a | 1.15 a | 0.08 a | 4.97 a |
| | High | 5.0 c | 0.65 a | 0.04 a | 0.030 a | 18 a | 33 a | 0.12 a | 1.11 a | 1.16 a | 0.09 a | 5.00 a |

*For each depth, values within a column followed by different letters differ significantly at $P = 0.05$

**Table 3. Effect of biosolids application on total concentrations of heavy metals in litter and soil (sampled in May 2010)***

| Depth | Treatment | As | Cd | Cr | Cu | Pb | Hg | Ni | Zn |
|---|---|---|---|---|---|---|---|---|---|
| | | mg $kg^{-1}$ | | | | | | | |
| | Control | 1.8 a | 0.07 b | 11 b | 10 b | 12 a | n.a | 8.3 a | 32 b |
| Littler | Standard | 2.3 a | 0.23 a | 17 a | 31 a | 12 a | n.a | 10 a | 68 a |
| | High | 2.5 a | 0.26 a | 18 a | 35 a | 12 a | n.a | 9.8 a | 75 a |
| | Control | 3.1 a | 0.17 a | 29 a | 3.6 b | 4.8 a | <0.01 a | 24 a | 30 a |
| 0-0.25m | Standard | 3.2 a | 0.18 a | 28 a | 4.6 a | 4.8 a | <0.01 a | 26 a | 29 a |
| | High | 3.2 a | 0.18 a | 29 a | 5.0 a | 4.9 a | <0.01 a | 25 a | 29 a |
| | Control | 3.2 a | 0.18 b | 28 b | 3.7 c | 4.4 b | <0.01 a | 37 a | 26 a |
| 0.25-0.5m | Standard | 3.2 a | 0.18 b | 27 b | 4.1 b | 4.4 b | <0.01 a | 33 a | 27 a |
| | High | 3.3 a | 0.19 a | 35 a | 4.5 a | 4.8 a | <0.01 a | 35 a | 29 a |

*For each depth, values within a column followed by different letters differ significantly at $P = 0.05$

Both the Standard and High treatments significantly ($P < 0.05$) increased Cu concentrations in both soil layers. The High treatment also significantly increased concentrations of Cd, Cr and Pd in the 0.25–0.5 m layer (Table 3). The results indicate slow soil accumulation and downward movement of these biosolids-derived heavy metals. The increased Cu concentration in soils receiving biosolids applications (Table 3) did not result in increased foliar Cu concentration of standing trees (data not shown), indicating a low bioavailability of the biosolids-derived Cu. General, heavy metal concentrations in soils with or without biosolids application (Table 3) were low and well below the soil contaminant limits defined by the guidelines for the safe application of biosolids to land in New Zealand (NZWWA 2003). Overall, repeated application of biosolids improved soil fertility, and increases in heavy metal concentrations were within soil quality guidelines.

*Effect of biosolids application on tree nutrition and growth*

Annual foliage analyses showed that foliar concentrations of all nutrients except N had remained in the "satisfactory" range for tree nutrition, indicating that none of these nutrients were limiting tree growth (Will 1985). However, the supply of natural soil N in the Rabbit Island *P. radiata* forest was low, with the foliar N concentration of the Control treatment averaging 1.2% N since monitoring began in 1998 (Figure 1, left). This indicates the *P. radiata* stand suffered N deficiency (Will 1985). Biosolids application significantly ($P < 0.05$) increased foliar N concentration of the Low treatment to marginal level (averaging 1.4% N) and of the High treatment to *sufficiency level (*averaging 1.5% N) (Figure 1, left). Compared to the Control treatment, biosolids application significantly reduced foliar concentrations of Ca by 20-28% and Mn by 42-51%. The Standard and High treatments increased tree stem volume by 25% and 34 % respectively when compared to the Control treatment (Figure 1, right).

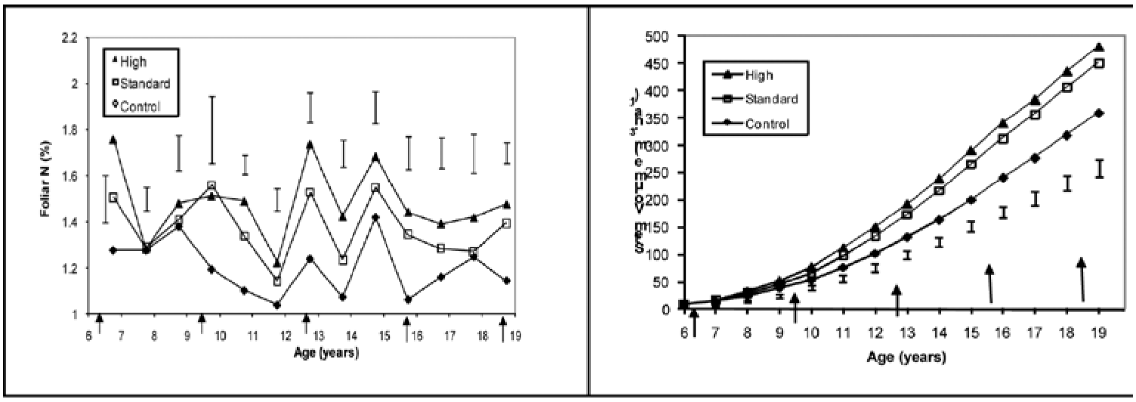


**Figure 1. Effect of biosolids application on foliar N concentration (left) and tree stem volume (right). Arrows indicate time of biosolids application. Error bars show least significant differences (*P* = 0.05) for comparisons among the treatments.**

## Conclusions

Repeated application of biosolids to a *P. radiata* plantation on a low fertility sandy soil significantly increased total soil C, N and P and their availabilities in soil. Improved tree growth with biosolids application was probably mainly a result of enhanced soil N supply. Biosolids application, especially the High treatment, resulted in a reduced soil pH and slow accumulation of total Cu, Cd, Cr and Pb. Overall concentrations of these heavy metals were considered very low for soil. However, the biosolids-derived heavy metals were strongly retained in the litter layer. *Further studies* are *warranted* to assess the long-term fate of biosolids-derived heavy metals.


## Acknowledgements

The authors wish to thank PF Olsen Limited, Tasman District Council, the Nelson Regional Sewerage Business Unit, NZ Ministry of Science and Innovation for funding or in-kind support, J.D. Graham for foliage and soil sampling, and all those who have contributed to the project over the years.


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# The effect of tree size on soil properties in a scattered tree environment

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Scattered trees are keystone structures because of the potential ecological functions they carry out. To quantify soil function under scattered trees the effect of different sized Yellow Box (*Eucalyptus melliodora*) trees on the spatial variation of soil properties was addressed. Mineral soil samples to a depth of 5cm and litter samples were taken under ten Yellow Box trees and in the surrounding grassland in an Australian temperate woodland. Trees ranged in size from 15.5cm to 132cm diameter at breast height (dbh). Soil properties sampled under the canopy improved compared to the surrounding grassland. For example total nitrogen (N), soil organic carbon (SOC) extractable phosphorus (P) soil pH, litter load and litter nutrient concentrations (total N and P) increase with bulk density decreasing under the canopy. The effect of scattered trees on soil function was significantly correlated to tree size with Dbh having a highly significant ($P$ <0.0001) effect on SOC, total N and bulk density. This was supported by multiple regression models of soil properties, where the effect of tree size was highly significant in all models. The position under the canopy was also significant in all models except for soil nitrogen. These models provide quantitative evidence of increased soil function under large scattered trees, and strong spatial heterogeneity in the patterning of these processes. These findings confirm the role of large trees as keystone structures. As a result the maintenance of large trees is critical for the maintenance of soil function within scattered tree environments.

# Arbuscular mycorrhizas modify tomato responses to soil zinc and phosphorus addition

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Arbuscular mycorrhizas (AM) play an important role in plant phosphorus (P) and zinc (Zn) uptake and nutrition; however, relatively few studies have directly investigated the interactive effects of these nutrients on AM. Therefore, we undertook a glasshouse experiment to study the effects of soil Zn and P on plant growth and nutrition, and AM formation and functioning. A mycorrhiza defective tomato mutant (*rmc*) and its mycorrhizal wildtype progenitor (76R) were used in this experiment. Plants were grown in soil that had been amended with five Zn concentrations, ranging from deficient to toxic, and two P concentrations. The benefits of forming AM were found to be greatest when plants were grown under low soil P and Zn. Furthermore, the effect of soil Zn supply on plant growth, nutrition and AM colonization was strongly influenced by the concentration of P in the soil. Consequently, a second glasshouse experiment was conducted in which the tomato genotypes were grown under five soil P addition treatments, and two soil Zn addition treatments. This experiment showed that mycorrhizal and non-mycorrhizal plants have different patterns of biomass allocation under high and low Zn conditions. Together these results indicate that AM responses to Zn supply are modulated by soil P supply. The results will be discussed in the context of improving our understanding of the interactive effects P and Zn on AM functioning, and plant nutrition.

# Time dependence of soil water repellency for volcanic ash soils in Japan and New Zealand

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Water repellency (WR) of soil can induce significant hydrological problems such as reduced water infiltration, enhanced surface runoff and erosion and the forming of preferential flow patterns in soils. Although WR has been observed in many countries including Japan and New Zealand, relatively few studies evaluated the time dependence of the degree of WR for aggregated volcanic ash soils. In this study, the effects of water content on the WR of Japanese and New Zealand volcanic ash soils at different depths were investigated. The time dependence of the degree of WR at different water contents and its relation to persistence were evaluated using the coefficient of temporal change in WR, A value (Subedi et al., 2012). The degree of WR of samples adjusted to different moisture contents was assessed with the sessile drop method and the molarity of ethanol droplet test as contact angle of a droplet to the soil surface, and the persistence of WR was quantified with the water drop penetration time test. The degree of WR varied considerably with region and soil depth while the clay saturation index (Dexter et al., 2008) with n=7 separated WR soils from non-WR soils. The initial contact angle increased sharply with increasing water content from air-dry, and then decreased gradually to 0$^{o}$. The contact angle sharply decreased with soil-water contact time at higher water contents (higher A value). The relationship between measured initial contact angle and A value gave a significant exponential relation.

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# Soil fertility changes following conversion of grassland to oil palm

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Changes in soil fertility following land use change in the tropics have implications for sustainability. We examined changes in soil fertility (0-0.05 and 0.10-0.15 m depth) following conversion of grassland to oil palm at 15 sites on volcanic ash soils in Papua New Guinea. The oil palm sites had been planted 1-25 years previously and adjacent grassland was assumed to represent the pre-oil palm condition. Under oil palm, samples were taken separately from the avenue (average 77.5% of area), frond pile (10.5%) and weeded circle (12.0%) management zones. The range of parameter values for all samples was: bulk density 457-1203 kg/m$^3$, pH 5.0-7.0, total C content 13-108 g/kg, Colwell P content 3-99 mg/kg, and effective cation exchange capacity (ECEC) 18-314 mmol+/kg. The difference between oil palm and grassland soil was not consistently related to age of the oil palm stands, indicating an effect of site characteristics on the land use-induced change in soil properties. Compared to grassland, and taking the mean of the two sampling depths at all sites, oil palm soil (weighted for zone area) had lower bulk density (difference of -9, -82, -16 kg/m$^3$ in the avenue, frond pile and weeded circle zones, respectively), lower pH (-0.3, 0.0, -0.4 units), lower C content (-2.2, +8.1, +0.1 g/kg), lower Colwell P content (-3, +6, -6 mg/kg) and lower ECEC (-14, +54, -14 mmol+/kg). Based on data from one site sampled to 1.5-m depth (25-year-old palms), the effects oil palm on soil fertility occurred mostly within the top 0.5 m.

# The Brigalow Catchment Study: Comparison of soil fertility, forage quality and beef production from buffel grass vs. leucaena-buffel grass pastures

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## Abstract

This paper assesses the fertility status of soils and forages in two grazed pasture systems in central Queensland: 1) buffel grass only, and 2) leucaena-buffel grass. Trends observed in our preliminary results indicate that the paddock with leucaena had similar or lower soil fertility (0 to 10 cm) than the grass only paddock based on concentrations of available phosphorus, organic carbon, nitrate and ammonia; however, the leucaena paddock had similar or better pasture quality based on crude protein and total phosphorus. This can be partly explained by the deeper root system of leucaena which can utilize nutrients further down the soil profile than grass pastures. Furthermore, greater live weight gains of cattle in the leucaena-grass paddock can be attributed to higher crude protein concentrations found in leucaena vs. grass leaves.

## Introduction

Leucaena hedgerows planted with companion pasture grasses are one of the most productive, profitable and sustainable grazing systems in tropical and subtropical Australia (Dalzell *et al*. 2006, Shelton and Dalzell 2007). In managed agricultural scenarios, leucaena has many reported benefits, including reduced soil erosion, improved runoff water quality, and enhanced soil fertility (Dalzell *et al*. 2006, Shelton and Dalzell 2007). Beef production on leguminous pastures in central Queensland is becoming more common, as there is an increase in the availability of data that demonstrates greater biomass and nutritive value of forage from leucaena-grass paddocks vs. grass only paddocks. This paper links data on soil and pasture fertility status with beef production per hectare between pastures with and without legumes.

## Methods

### *Study site*

This research was conducted on the Brigalow Catchment Study (BCS) site near Theodore in central Queensland. The BCS commenced in 1965 and is an ongoing long-term study on the impact of land development on hydrology, productivity and resource condition. This project compares two paddocks from the BCS site: 1) buffel grass (*Cenchrus ciliaris* cv. Biloela) pasture that was originally cleared in 1982 and planted in 1983, and 2) leucaena (*Leucaena leucocephala* cv. Cunningham) and buffel grass pasture that was originally cleared ~1968 and cropped for 10 to 15 years before converting into a grazed paddock. The leucaena was planted in 1998 on 8 m hedgerows. Both paddocks are predominantly grey and black Vertosols with an average slope of 2.5%, and neither paddock has a history of fertiliser application.

### *Soil fertility and forage quality*

Soil was sampled at approximately three month intervals from December 2008 until October 2009. Six samples 0 to 10 cm from the soil surface were collected from three permanent monitoring sites within both the grass only (N=18) and leucaena-grass paddock (N=18) for each sampling period. The samples were analysed for Colwell available phosphorus (P), Walkley and Black organic carbon (OC), nitrate ($NO_3$-N), and ammonia ($NH_4$-N). Pasture was sampled at approximately three month intervals from December 2008 until March 2010. Six samples 1x1 m were collected from the same three permanent monitoring sites within both the grass only (N=18) and leucaena-grass paddock (N=18) for each sampling period. The samples were analysed for total Kjeldahl nitrogen (TKN) and phosphorus (TKP). Crude protein (CP) was later calculated as 6.25 x TKN.

*Beef production*

Two drafts of weaner cattle were grazed on the grass only and leucaena-grass pastures; the first draft from May 2008 to May 2009 and the second draft from June 2009 to March 2011. Similar stocking rates were used for the first grazing period with 2.1 ha per head for the grass only paddock and 2.2 ha per head for the leucaena-grass paddock. In the second grazing period, the stocking rate was decreased in the grass only paddock to 3.4 ha per head and increased in the leucaena-grass paddock to 1.5 ha per head to match feed availability. Production was measured as cumulative weight gain of the cattle per ha. Values reported are based on results from Thornton and Buck (2011).

*Statistical analyses*

Means and standard errors were calculated in GenStat (v.14) for soil and pasture fertility. Furthermore, one-way ANOVA's with protected Fisher's LSD for pasture type were performed ($P$<0.05).

**Results**

*Soil fertility and forage quality*

Overall, there was a trend of lower organic carbon and available phosphorus in soils of the leucaena-grass pastures (OC mean 1.22% ± S.E. 0.03%; P mean 9.25 mg/kg ± S.E. 0.29 mg/kg) than in the grass only pastures (OC mean 1.77% ± S.E. 0.04%; P mean 11.89 mg/kg ± S.E. 0.35 mg/kg) (Fig.1). Both of these nutrients exhibited little variation, with an overall detectable difference between paddocks (OC $F_{1, 178}$=85.96, $P$<0.001; P $F_{1, 178}$=29.13, $P$<0.001).

Nitrate concentrations were higher in the grass only (mean 3.64 mg/kg ± S.E. 0.25 mg/kg) than leucaena-grass paddock (mean 2.99 mg/kg ± S.E. 0.26 mg/kg) (Fig.1). Nitrate in the grass only paddock was lower in the wet season from November to March than in the dry season; however, the late commencement of sampling from the leucaena-grass paddock does not currently allow seasonal variations to be determined. Overall, large variability was observed in the results for the last three sampling periods and no difference could be detected between the two pasture systems ($F_{1, 178}$=3.01, $P$=0.084).

Ammonia was similar between the two pasture systems in the last three sampling periods; however, concentrations were overall higher in the leucaena-grass (mean 5.50 mg/kg ± S.E. 0.27 mg/kg) than grass only paddock (mean 4.81 mg/kg ± S.E. 0.22 mg/kg) (Fig.1). Little variation was observed in the data, but an overall difference between the paddocks was detected ($F_{1, 178}$=3.97, $P$=0.048).

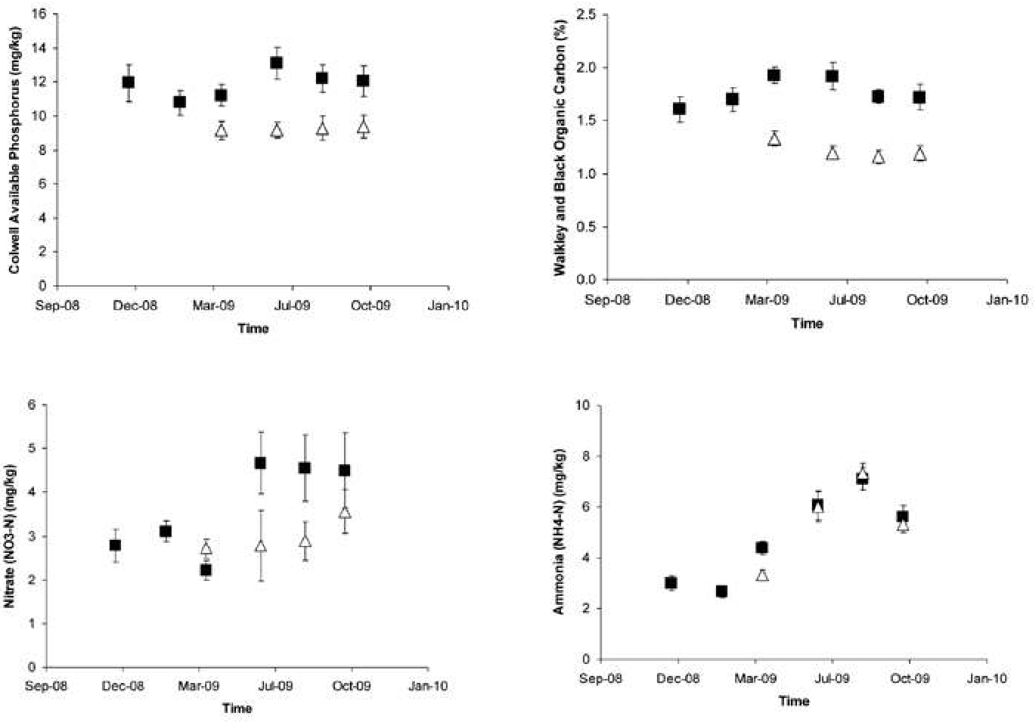


**Figure 1. Fertility of soil (0 to 10 cm) based on available phosphorus, organic carbon, nitrate and ammonia concentrations in the grass only (black square) and leucaena-grass (white triangle) pastures; mean ± standard error bars.**

Forage yield was greater in the grass only (mean 1617.0 kg/ha ± S.E. 134.5 kg/ha) than in the leucaena-grass paddock (mean 906.5 kg/ha ± S.E. 56.3 kg/ha), whilst yield from leucaena leaves was much lower (mean 385.9 kg/ha ± S.E. 51.3 kg/ha) (Fig.2). Although variability within sampling periods was larger in the grass only paddock, overall, differences were detected between all three pasture types ($F_{2, 336}$=30.59, $P$<0.001).

Crude protein of pasture grasses from the grass only (mean 4.44% ± S.E. 0.15%) and the leucaena-grass paddock (mean 5.88% ± S.E. 0.27%) were similar over time (Fig.3). In contrast, crude protein of leucaena leaves (mean 16.70% ± S.E. 0.59%) from the latter paddock was much higher than from the companion pasture grasses in the same paddock. Variability within each treatments sampling period was small, and overall, a difference between all three pasture types was detected ($F_{2, 318}$=322.54, $P$<0.001).

Total phosphorus concentrations were similar between pasture grasses from the grass only (mean 0.10% ± S.E. 0.005%) and the leucaena-grass paddock (mean 0.12% ± S.E. 0.004%), and also leucaena leaves (mean 0.12% ± S.E. 0.004%) from the latter paddock (Fig.3). Although data from all three pasture types exhibited variability in concentrations overtime, overall the grass only paddock had a detectably lower concentration that the other two pasture types ($F_{2, 318}$=10.20, $P$<0.001).

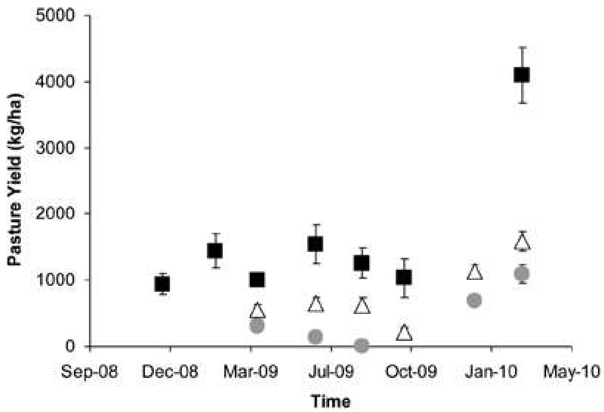


**Figure 2. Total yield of pasture from the grass only (black square) and leucaena-grass (leucaena = grey circle, and grass = white triangle) paddocks; mean ± standard error bars.**

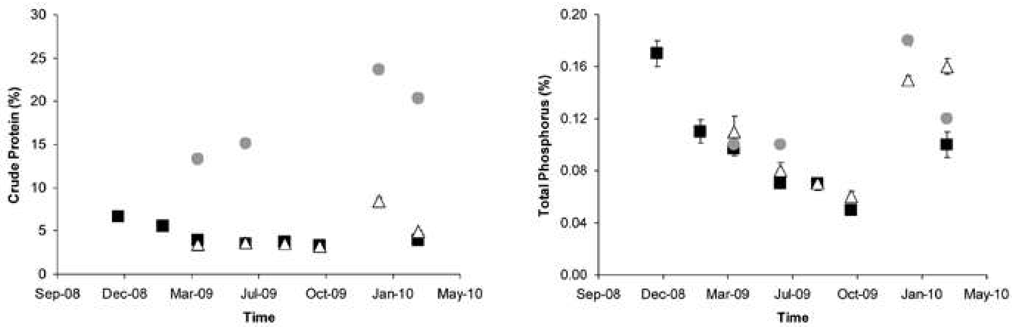


**Figure 3. Nutritive value of pastures based on crude protein and total phosphorus in the grass only (black square) and leucaena-grass (leucaena = grey circle, and grass = white triangle) paddocks; mean ± standard error bars.**

*Beef production*

During the first grazing period when the two pasture systems had similar stocking rates, the cumulative live weight gain of cattle per ha was comparable, though slightly higher in the leucaena-grass (94 kg/ha) than in the grass only paddock (81 kg/ha) (Fig.4). During the second grazing period when stocking rates were adjusted to match feed availability, beef production per hectare was much higher in the leucaena-grass paddock (117 kg/ha) whereas it remained consistent in the grass only paddock (57 kg/ha) (Fig.4).

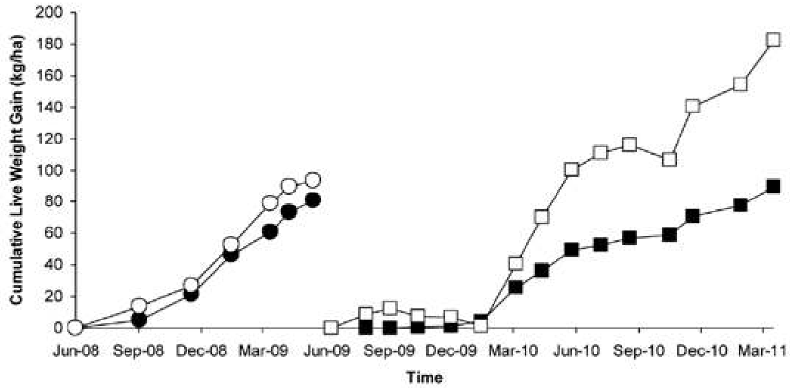


**Figure 4. Cumulative live weight gain of cattle grazed on grass only (black shapes) and leucaena-grass (white shapes) paddocks at similar (circle) and feed on offer (square) stocking rates. The two stocking rate treatments used a different draft of cattle. Source: Thornton et al. (2010), Thornton and Buck (2011).**

## Discussion

In this study, soil fertility was similar or lower in the leucaena-grass paddock than the grass only paddock. However, the nutritive value of grass forages was similar or better for the leucaena-grass paddock. The contrasting results can be partly explained by soil sampling depth (0 to 10 cm) and the physiology of the studied plants. That is, leucaena has a deep root system that can access subsoil moisture and nutrients that are typically beyond the reach of grass roots. The deep root system of leucaena enables the plant to remain productive during the dry season; thus, enabling continued cattle production (Radrizzani *et al.* 2010). However, the results in this paper are in contrast to other literature which discuss the importance of grass in leguminous pastures to utilise mineral nitrogen (Fillery 2001) and improvements to soil fertility through nitrogen fixation (Shelton and Dalzell 2007). Thus, it is surprising that the nutritive value of grass from both the grass only and leucaena-grass paddock are similar over time, as it indicates that the transfer of nitrogen from leucaena to the grass is inefficient or not occurring at all. Based on feed availability, the leucaena-grass paddock was able to stock more cattle resulting in greater beef production per hectare than the grass only paddock. Due to the lower concentrations of organic carbon, phosphorus and nitrate in soils from the leucaena-grass paddock, the greater cattle stocking rates and live weight gains observed can be attributed to the higher concentrations of crude protein found in the leucaena leaves (Shelton and Dalzell 2007).


## Acknowledgments

This work was jointly funded by the Department of Natural Resources and Mines and the Reef Rescue Research and Development Program (Project RRRD009).


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# Phosphorus nutrition of vetch in cotton-based rotations sown in a Vertosol from northern NSW

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## Abstract

Introducing vetch (*Vicia* spp.) into rotations in cotton-wheat systems of northern NSW has a number of benefits. This study investigated the phosphorus (P) nutrition and response of 3 vetch species grown in soil from long-term rotations at the Australian Cotton Research Institute, Narrabri, NSW. Labile P increased by 36% in the surface soil of long term rotations where vetch was included in the rotation. This resulted in up to 17% more dry matter production of vetch grown in cotton-wheat-vetch rotation soils in a system where legumes remained responsive to P application. Part of this response can be attributed to improved nodulation and N uptake associated with higher available P where vetch is included in the rotation. Of the three vetch varieties examined, Woolypod vetch (*Vicia villosa* ssp. *dasycarpa* cv. *Namoi*) indicated the greatest P efficiency. The study demonstrates that legume inclusion in rotation improves P status and production as well as providing labile N for following crops.

## Introduction

Legumes increase soil mineral N content, improve soil structure, increase infiltration, enhance water stable aggregates, aid decomposition of high C: N material and increase biological activity (Rochester and Peoples 2005). As a result vetch (*Vicia* spp.) has been used in cotton rotations to enhance these factors, as well as break disease cycles in cotton production and cycle nutrients important for plant growth and yield. Vetch is a vigorous frost-, drought- and disease-tolerant, forage legume that is useful as a break-crop due to its limited regrowth post manuring (Evans *et al.* 2003; Rochester and Peoples 2005). Vetch can contribute up to 130 kg mineral N/ha to a rotation system when green manured (Evans *et al.* 2003). A number of varieties are available, but the most commonly used is the Namoi Woolypod in northern NSW. New varieties are being developed by South Australian researchers, and this study compares the phosphorus (P) responsiveness of Namoi Woolypod with Popany (*Vicia villosa* subsp. *banghalensis*) and Rasina (*Vicia sativa* cv. Rasina) vetches. The P status in northern Vertosols is increasingly constraining production of cereals and cotton crops due to depletion associated with long term cropping without replacement (Wang, Lester *et al.* 2007). Increasingly researchers are investigating the P benefits of legume break crops, with evidence suggesting that P availability to crops following legume breaks is higher (Bunemann *et al.* 2006)**.** We investigated the long term benefits of including vetch in a rotation on the availability of P to a variety of vetch species.

## Materials and Methods

Soil was collected from the Australian Cotton Research Institute (ACRI), Myall Vale, Narrabri, NSW, (30º18’23.13”S 149º46’20.89”E), from three treatments within the long term rotation trial described by Hulugalle *et al.* (2012a) Approximately 40 kg of surface soil (0-10cm) was collected from 1) cotton – winter fallow – cotton plot (C-C) ; 2) summer and winter fallow – cotton – wheat (stubble incorporated) (C-W), and; 3) summer and winter fallow – cotton – wheat (stubble standing) – summer fallow – vetch (C-W-V) plots from each of the replicated plots. Soil analysis of the plots was undertaken on separate samples taken from replicates within the field trial, whilst the pot trial used a bulk mixture of soil from each rotation. Soil was collected in March and C-C plots had cotton standing, C-W plots were fallow, and C-W-V plots were taken with standing wheat stubble present but no growing plants.

A full factorial pot trial was established with rotation (n=3), phosphorus status (n=2) and vetch variety (n=3) as the factors, replicated 4 times. Basal nutrients were added to all pots and moisture was maintained at field capacity throughout the trial. Phosphorus was applied at 150 mg/kg to positive P control pots. The vetch species used in the study were Rasina (*Vicia sativa* cv. *Rasina)*, Popany (*Vicia villosa* subsp. *banghalensis)* and Namoi Woolypod (*Vicia villosa* ssp. *dasycarpa* cv. Namoi*)*. Five pre-germinated Woolypod, Popany or Rasina vetch seedlings were planted in each pot and growth conditions were maintained at a temperate 22°C maximum temperature for 8 weeks. Plants were harvested, dried ground and digested for P according to the method of Anderson and Henderson (1986) and N status by combustion on a LECO. Roots were carefully washed from the soil following soaking of the soil and nodules on the root system were counted.

Sequential P fractionation was undertaken on the soil from each rotation according to the method of (Guppy *et al.* 2000) and P status was measured using malachite green. Labile P samples were considered to be resin and bicarbonate extractable pools. All data were analysed using a 3-way ANOVA and significant differences ($P$<0.05) were determined using LSD's calculated where significant F-tests were observed.

**Results**

Labile P (the sum of resin and bicarbonate extractable P) in the vetch rotation was 36% greater than the continuous cotton rotation (Table 1). Total P was12% higher in the surface soil of the vetch rotation relative to continuous cotton and cotton-wheat rotations (Table 1).

**Table 1: Labile and total P (mg/kg) of a Vertosol (0-10 cm) after 7 years of a rotation trial including various combinations of cotton (*Gossypium hirsutum*), wheat (*Triticum aestivum*) or vetch (*Vicia villosa* ssp. dasycarpa cv. Namoi).**

| Rotation | Labile P (mg/kg) | Total P concentration (mg/kg) |
|---|---|---|
| Cotton-cotton | 47 | 590 |
| Cotton - wheat | 43 | 590 |
| Cotton-wheat-vetch | 67* | 670* |

*indicates significant difference within a column ($P$<0.05).

Woolypod vetch produced 21 and 11 % greater dry matter than Popany and Rasina respectively (Table 2). The application of P increased vetch dry weight by 37% (P<0.001) (Table 2). The cotton-wheat-vetch rotation produced 15 and 17 % more dry matter than the continuous cotton and cotton-wheat rotations respectively ($P$=0.003) (Table 2).

**Table 2: Vetch (*Vicia sativa* cv. Rasina, *Vicia villosa* subsp. *banghalensis*, and *Vicia villosa* ssp. *dasycarpa* cv. Namoi) dry matter production (g/pot) in a Vertosol soil collected from rotations with continuous cotton, cotton-wheat or cotton-wheat-vetch following eight weeks growth in a glasshouse with and without P addition.**

| | Continuous cotton | | Cotton -wheat | | Cotton-wheat-vetch | |
|---|---|---|---|---|---|---|
| | -P | +P | -P | +P | -P | +P |
| Popany | 0.56* | 1.13* | 0.63* | 1.29 | 0.92 | 1.33 |
| Woolypod | 0.90 | 1.53 | 0.83 | 1.49* | 1.23* | 1.39 |
| Rasina | 0.85 | 1.41 | 0.76 | 1.22 | 1.09 | 1.55* |

*indicates significant difference within a column ($P$<0.05).

In the majority of treatments the addition of superphosphate resulted in a threefold increase in P uptake (Table 3). The exception is the C-W-V rotation where P uptake doubled, except for Rasina which had a P uptake four times greater when P was added. Phosphorus uptake was up to 22% higher from soil from the vetch rotation (Table 3).

**Table 3: Vetch (*Vicia sativa* cv. Rasina, *Vicia villosa* subsp. *banghalensis*, and *Vicia villosa* ssp. *dasycarpa* cv. Namoi) phosphorus uptake (µg/pot) in a Vertosol soil from rotations with continuous cotton, cotton-wheat or cotton-wheat-vetch following eight weeks growth in a glasshouse with and without P addition.**

| | Continuous cotton | | Cotton -wheat | | Cotton-wheat-vetch | |
|---|---|---|---|---|---|---|
| | -P | +P | -P | +P | -P | +P |
| Popany | 140* | 480* | 180* | 580 | 320* | 610* |
| Woolypod | 180 | 600 | 190 | 660 | 380* | 700* |
| Rasina | 170 | 570 | 190 | 500* | 270* | 810* |

*indicates significant difference within a column ($P$<0.05).

Within each species, N uptake was similar where P was applied to soil from each rotation (Table 4). However, in the absence of P, N uptake was up to 36% higher from soil collected from the vetch rotation (Table 4).

**Table 4: Vetch (*Vicia sativa* cv. *Rasina, Vicia villosa* subsp. *banghalensis*, and *Vicia villosa* ssp. *dasycarpa* cv. Namoi) nitrogen uptake (mg/pot) from Vertosol soil from rotations with continuous cotton, cotton-wheat or cotton-wheat-vetch following eight weeks growth in a glasshouse with and without P addition.**

| | Continuous cotton | | Cotton -wheat | | Cotton-wheat-vetch | |
|---|---|---|---|---|---|---|
| | -P | +P | -P | +P | -P | +P |
| Popany | 2.2* | 4.4* | 2.4* | 4.6 | 4.0 | 5.5 |
| Woolypod | 3.2* | 6.1* | 3.2 | 6.3* | 5.0* | 6.5* |
| Rasina | 2.7* | 5.5* | 2.9 | 4.2 | 3.6 | 4.9 |

*indicates significant difference within a column ($P$<0.05).

Addition of P increased nodulation by 43%. Nodulation was higher in the C-W-V rotation in the absence of P (Table 5).

**Table 5: Nodule numbers of vetch (*Vicia sativa* cv. Rasina, *Vicia villosa* subsp. *banghalensis*, and *Vicia villosa* ssp. *dasycarpa* cv. Namoi) grown in a Vertosol soil from rotations with continuous cotton, cotton-wheat or cotton-wheat-vetch following eight weeks growth in a glasshouse with and without P addition.**

| | Continuous cotton | | Cotton -wheat | | Cotton-wheat-vetch | |
|---|---|---|---|---|---|---|
| | -P | +P | -P | +P | -P | +P |
| Popany | 4 | 31 | 11 | 24 | 22 | 32 |
| Woolypod | 17 | 27 | 16 | 15 | 30 | 30 |
| Rasina | 11 | 31 | 21 | 27 | 39 | 33 |

*indicates significant difference within a column ($P$<0.05).

**Discussion**

The long-term presence of vetch in the rotation significantly increased labile P with flow on consequences for N nutrition and growth. All three varieties of vetch trialled were more productive when grown in soil where vetch was part of the rotation. These results are related to two factors, increased labile P and the increased nodulation (and hence N availability). The increase in nodulation in vetch rotation soil may be related to a higher rhizobial population associated with a history of vetch growth in the rotation. However, the ability of soils supplemented with P to achieve similar nodulation where vetch was not part of the rotation suggests that it was in fact the higher P nutrition of the vetch soil that resulted in higher nodulation and N uptake. Furthermore increased N uptake is most likely related to better P nutrition and root growth, rather than significant contributions from increased nodulation of vetch over such a short time-scale. Vetch in the rotation does contribute N to the soil (Hulugalle *et al.* 2012b), but as all pots were dried and passed through a sieve prior to use, mineralisation of labile C and N would have been high regardless. This does not preclude increased lability of soil with vetch residues, but suggests better root growth from P availability drove increased N uptake.

Producers in northern NSW have incorporated Namoi Woolypod vetch into rotations due to its drought and frost tolerance seeking disease break benefits and fixed N. Another advantage may be better P use efficiency. Woolypod vetch achieved the highest growth as a percentage of P unlimited growth in all rotations, implying a much greater efficiency of P utilisation under constrained conditions. Further research is necessary to elucidate both the increased efficiency of Woolypod vetch and mechanisms underlying increased labile P where vetch is included in the rotation. The study demonstrates that legume inclusion in rotation improves P status and production as well as providing labile N for following crops.

## Acknowledgements

The authors would like to acknowledge the assistance of Michael Faint and Leanne Lisle in glasshouse and technical analysis and the Cotton Catchment Communities CRC for Summer Scholarship funding of this project.

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# An electrostatic model predicting metal toxicity to root growth and microbial processes in soils

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Assessing environmental risks of metal contamination in soils is a complex task because the biologically effective concentrations of metals in soils vary widely with soil properties. Toxicity data for plants and microorganisms in soils have been described with ion competition models, however these models disregard electrostatic and osmotic effects known to affect ion sorption and toxicity. Using European soils with diverse characteristics, the factors that influence the toxicity of soil Cu or Ni to root elongation rate (RER), potential nitrification rate (PNR), and glucose-induced respiration (GIR) were evaluated based on the electrical potential ($\psi_0$) and ion activities of metal ($\{M^{2+}\}_0$) at the outer surfaces of root- or bacterial-cell membranes (CMs). The toxicity data (RER, PNR, and GIR) were statistically related to (i) $\{M^{2+}\}_0$, (ii) the $\psi_0$-influenced electrical driving force for cation uptake across CMs, and (iii) osmotic effects. For root growth, Ca deficiency was also identified as an important factor. Electrostatic models were developed to relate RER ($R^2 > 0.751$), PNR ($R^2 > 0.818$) and GIR ($R^2 > 0.868$) to these factors. These predictions were generally better than those by previous models, demonstrating the importance of electrostatic effects on the toxicity of metals. The suggestion that metal toxicity in spiked soils is partly related to a spike-induced osmotic increase is corroborated by fitting the model to spiked soils that were or were not leached to reduce the osmotic increase. This study extends current theory to evaluate the bioavailability and toxicity of metals, indicating potential utility in risk assessment in soils.

# Effects of soil fertility and compaction on root standing mass and production

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The dynamics of root growth in pasture systems are likely to come under increasing scrutiny, given the role of roots as a major source of soil carbon (C) inputs. Efforts to increase soil C stocks via pasture management will rely on increased knowledge about the effects of management (e.g. soil fertility, grazing intensity) on root mass, depth and turnover.

We conducted two studies of root standing mass and production under an irrigated dairy pasture in the Manawatu region of New Zealand. The first examined the effects of nitrogen (N) inputs and soil phosphorus (P) status over one year (2008-9) and the second examined the interacting effects of soil P status and artificial soil compaction over one year (2011-12). Root measurements were made to 150 mm depth and production was measured by the in-growth core technique.

Over most of the two periods studied there were no significant differences in root standing mass or production between treatments. However, higher N+P fertility resulted in lower root standing mass (20% less) but higher root production (35% greater) during spring 2008. In 2011, higher P fertility resulted in lower root mass in autumn only and no effect on growth. Compaction reduced root production by ~20% year-round. Fertility effects on root growth seem more strongly influenced by N than P in this system. If soil C inputs are proportional to root mass then these results indicate the potential for high soil fertility to contribute to reported declines of soil C in intensive systems.

# Monday 3 December

## SOIL FERTILITY AND SOIL CONTAMINANTS (NITROGEN)

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# Nitrate leaching from high production forage crop sequences in New Zealand

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### Abstract

New Zealand pastoral industries have set a goal of producing 45 t dry matter (DM)/ha annually from supplementary feed crops. Achieving this will require high inputs of water and nutrients that may increase the risk of nitrate leaching. This research describes nitrate leaching losses from the first year of intensively managed forage crop sequences to identify mitigation options. For cut-and-carry crops, the highest annual nitrate leaching losses were from crop sequences starting with maize compared to kale or barley. While maize-based sequences had the highest production (30-32 t DM/ha) they also had the highest N leaching losses per t DM produced (2.4 kg N/t DM). The best performing sequence was barley/oats/Italian ryegrass which produced 28 t DM/ha but leached <1 kg N/t DM produced. Based on plots treated with synthetic urine and subjected to treading, we calculated leaching losses of 70-124 kg N/ha from autumn grazed crops (forage rape or oats). In general, nitrate leaching losses were not closely linked to estimates of cumulative drainage or fertiliser N inputs but were strongly influenced by crop type, sowing date and additions of livestock urine.

### Introduction

New Zealand pastoral industries have set a goal of producing 45 t DM/ha annually from supplementary feed crops as well as reducing N leaching losses by 50% relative to baseline levels. Meeting the industry's feed production targets will require high inputs of water and nutrients that may increase the risk of N losses. This is a concern given that elevated concentrations of nitrate ($NO_3^-$) have been measured in shallow ground water in the Canterbury Plains and elsewhere in New Zealand (Francis *et al.* 1999). Our aim was to quantify $NO_3^-$ leaching losses from intensively managed, high input, high production forage crop sequences established in Canterbury, NZ to maximise production. We describe results from two years of research as the first step in identifying management practices that mitigate N losses from high production forage crop systems.

### Materials & Methods

The Forage Crop Sequence Trial was established on a silt loam soil (sand at 30-50 cm) at Lincoln, New Zealand. It comprised successive summer and winter crops grown in rotation over 2 years (Oct 2007 to Oct 2009) to maximise DM production. The first crop in each sequence (est. Spring 2007) was either maize (*Zea mays*), kale (*Brassica oleracea*), or barley (*Hordeum vulgare*) (Table 1). After harvest, the treatments were split to give six in the second phase (autumn-winter 2008) and 12 in the third (spring-autumn 2008/09) and fourth (autumn/winter 2009) phase of each sequence. Sequences based on maize or kale were managed as cut-and-carry crops. Those based on barley were suitable for grazing (rape, oats, triticale) or whole crop silage (barley). Untreated control, fertiliser and synthetic urine treated subplots (2x2 m) were also established in the ex-barley plots.

All crops were managed with optimum rates of N and irrigation to produce maximum crop yields. Irrigation was applied to maintain non-limiting soil water deficits (0-80 mm, 1.5 m). Soil moisture (20 cm intervals to 1.50 m), air and soil temperatures and total incident radiation were recorded. Mineral N (2M KCl) was measured in soils (0-20, 20-40, 40-60, 60-90, 90-120 and 120-150 cm) collected shortly before sowing and following harvest of summer and autumn/winter crops. Nitrate leaching was measured using soil solution samplers (3.5 cm d. PVC tubes with porous ceramic tips) placed at 0.60 and 1.50 m depth in each plot. Solution samples were collected whenever rainfall triggered a drainage event > 15 mm. Drainage was calculated with a water balance model. Nitrate ($NO_3$-N) and ammonium ($NH_4$-N) concentrations were analysed on a Foss FIAstar 5000 analyser.

**Results & Discussion**

The average total biomass production of summer crops was 23.2, 21.3 and 16.4 t DM/ha for maize, kale and barley, respectively. Other crop performance information has been reported by de Ruiter *et al.* (2009). The total crop water use by maize, kale and barley was 659, 650 and 398 mm, respectively. However, owing to differences in soil water deficits imposed by the crops over time in association with differences in crop growth rates and duration (de Ruiter *et al*. 2009), drainage of water from the top 0.6 m of soil under maize (93 mm) and kale (72 mm) was considerably higher than that of the earlier harvested barley crop (24 mm).

Nitrate leaching losses from the summer crops at 0.6 m depth were low but significantly different ($P<0.05$), ranging from near zero losses under barley to 10 and 25 kg N/ha under kale and maize, respectively (data not shown). The $NO_3^-$ leached under maize and kale crops at this depth was primarily associated with two drainage events where heavy rainfall followed a recent irrigation application. The difference in the $NO_3^-$ leached from all three crops was more consistent with differences in drainage than fertiliser N inputs, though a full interpretation of these results requires a detailed N budget over time (i.e. during crop development) that is beyond the scope of this paper. Very low levels of $NO_3^-$ leaching (0.5 to 6 kg N/ha) were also recorded under the summer crops at 1.5 m depth though these losses are most likely associated with mineral N that was in the soil profile at the time the first crops were sown.

Despite low levels of nitrate leaching from the first crops, there were differences in the concentration of $NO_3^-$ remaining in the soil profile following their harvest that were consistent with the differences in $NO_3^-$ leached from the first crops. Nitrate concentrations following barley and kale ranged from about 20 µg N/g in the top 20 cm to about 12 µg N/g at the 1.2-1.5 m depth. On average, these concentrations were only about 10 µg N/g higher than the levels measured before the establishment of the crops in October (baseline). In contrast, nitrate concentrations following the maize crops ranged from about 75 µg N/g in the top 20 cm to as low as 30 µg N/g at 1.2-1.5 m depth. There was a significant $NO_3^-$ bulge at the 0.9-1.2 m depth where concentrations exceeded 50 µg N/g. As discussed below, this residual mineral N posed a risk to $NO_3^-$ leaching during the establishment of the second crops (wheat and triticale/faba beans) that followed maize.

Drainage and nitrate leaching losses under the autumn/winter crops were much higher than those measured under the summer crops and the leaching losses were significantly affected by the previous summer crop ($P<0.01$). For this reason the results for the summer and autumn/winter crops were combined to investigate the effects of each forage crop sequence on drainage and $NO_3^-$ leaching losses (Table 1). In general, there were no significant effects of the crop sequences on the total drainage of water over the first year of cropping but there was a significant effect of measurement depth. On average, the drainage from 0.6 m depth under these crop sequences was 322 mm compared to about 262 mm at 1.5 m depth over the annual cycle.

**Table 1: Total fertiliser N applied, drainage and $NO_3^-$ leached from each forage crop sequences grown.**

| | | ----------- 0.6 m ---------- | | ----------- 1.5 m --------- | |
|---|---|---|---|---|---|
| Crop Sequence (Abbrev) | N Fertiliser (kg N/ha) | Drainage (mm) | $NO_3^-$ leached (kg N/ha) | Drainage (mm) | $NO_3^-$ leached (kg N/ha) |
| Barley/Rape/Oats (Bro) | 342 | 287 | 43 | 255 | 15 |
| Barley/Oats/Ryegrass (Boi) | 252 | 330 | 26 | 258 | 31 |
| Kale/Wheat (Kw) | 321 | 325 | 79 | 266 | 34 |
| Kale/Triticale+bean (Ktb) | 321 | 306 | 40 | 256 | 28 |
| Maize/Wheat (Mw) | 250 | 340 | 73 | 260 | 53 |
| Maize/Triticale+bean (Mtb) | 250 | 345 | 77 | 277 | 53 |
| LSD (0.05) | | 52 | 31 | 57 | 20 |

Nitrate leaching losses from the forage crop sequences were significantly affected by the first and second crops grown, the depth of measurement and their interaction. Over the full annual cycle, the $NO_3^-$ leached from the maize-based sequences (Mw & Mtb) at 0.6 m depth (73-77 kg N/ha) was higher than all of the other crop sequences except where wheat followed kale (Kw) (Table 1). Overall, the barley-based sequences had the lowest total $NO_3^-$ leaching losses from 0.6 m depth; the losses from Bro (43 kg N/ha) being slightly higher than the losses from Boi (26 kg N/ha). The relatively low leaching losses from the barley-based sequences at this depth are most likely the result of higher water and N use owing to earlier establishment and higher DM production by the forage rape/oats (Bro, 7.5 t DM/ha) and oats/ryegrass (Boi, 11.6 t DM/ha) crops in these sequences. For the kale-based sequences, the leaching losses from wheat (Kw, 79 kg N/ha) were much higher than the losses from triticale+faba beans (Ktb, 40 kg N/ha). In this case the triticale+faba beans established more rapidly than the wheat but their total DM production at harvest did not differ greatly (3.1 vs 4.7 t DM/ha, respectively). The amount of $NO_3^-$ leached from 0.6 m under these crops did not correspond with differences in total drainage or fertiliser N inputs.

Despite similar amounts of drainage at 1.5 m depth, the average $NO_3^-$ leached from the maize-based sequences (53 kg N/ha) was about 1.7 times more than that of the kale-based sequences (31 kg N/ha) and 2.3 times that of the barley-based sequences (23 kg N/ha). These losses are comparable to the estimated annual losses of 30-40 kg N/ha (Francis *et al.* 1999; de Klein & Ledgard 2001) from typical dairy pastures (2.5-2.8 cows/ha; 50-60 kg/ha fertiliser N) and measured annual losses of 20-75 kg N/ha in the Waikato (Ledgard *et al.* 1999) and Southland (Monaghan *et al.* 2005). Leaching losses can vary substantially from year to year in relation to winter rainfall and distributions (Francis 1995). However, this variation may be reduced, and the total $NO_3^-$ leached increased, where irrigation lowers soil water deficits.

Among the six crop sequences, the maize-based sequences of Mw and Mtb had the highest total DM production, Bro and Ktb sequences were intermediate and Kw and Bro the lowest. Although the Mw and Mtb had the highest DM production, they also leached more nitrate-N per t DM produced (2.4 kg N/t DM) from the 0.6 m depth than all of the other crop sequences except where wheat followed kale (Kw, 3.2 kg N/t DM). Overall, the best performing sequence was Boi which produced a total of 28 t DM/ha but lost <1 kg N/t DM produced. Although the losses from 1.5 m were lower per t DM produced, similar differences were observed between the crop sequences.

The highest nitrate leaching losses were measured at 0.6 m depth under autumn/winter crops treated with synthetic urine and simulated treading (Table 2). Nitrate leaching losses from the 0.6 m depth were nearly 2-fold higher from the urine-treated plots under ryegrass (Boi, 368 kg N/ha) than under oats (Bro, 134 kg N/ha). Leaching losses from the untreated control plots were only 5-32 % of the losses from the grazed treatments and did not differ significantly from the fertilised treatments. Based on these measured losses, if we assume that approximately 30% of the soil surface area is exposed to urine when a forage crop is grazed, then the $NO_3^-$ leached from these autumn grazed crops would be expected to range between 70 and 124 kg N/ha. The $NO_3^-$ leached from 1.5 m depth was much lower than from the 0.6 m and was comparable to the losses measured at this depth under the other fertilised but ungrazed autumn/winter crops (e.g. Mw & Mtb).

The $NO_3^-$ leached from the urine-treated ryegrass plots (Boi, 59 kg N/ha) was nearly double that of the control and fertilised treatments but there were no significant effects of urine additions to the oat plots where the losses were much lower (Bro, 14-21 kg N/ha). The higher leaching loss from the urine-treated ryegrass (Boi) plots at 0.6 m was consistent with the high concentrations of $NO_3^-$ measured in the subsoil (0.6-1.5 m) following harvest of the crop (Fig. 1). In contrast, the $NO_3^-$ bulge recorded at 0.2-0.6 m depth in the urine-treated oat plots at crop harvest was consistent with the lower leaching loss measured from 0.6 m depth in this treatment. The high levels of residual N in the soil profile of urine-treated plots following crop harvest may pose a risk for $NO_3^-$ leaching during the establishment of subsequent crops.

**Table 2. Effects of fertiliser and cow urine on drainage and $NO_3^-$-N leached from ex-barley forage crop sequences**.

| | | | ----------- 0.6 m ------ | | ------------- 1.5 m ----------- | |
|---|---|---|---|---|---|---|
| Crop sequence | Treatments | N Applied (kg/N ha) | Drainage (mm)[1] | N Leached (kg N/ha) | Drainage (mm) | N Leached (kg N/ha) |
| Rape/Oats (Bro) | Control | 0 | 220 | 43 | 190 | 14 |
| | Fertilised | 219 | 263 | 45 | 217 | 19 |
| | Urine | 800 | 220 | 134 | 190 | 21 |
| Oats/Ryegrass (Boi) | Control | 0 | 238 | 15 | 200 | 30 |
| | Fertilised | 129 | 251 | 26 | 202 | 31 |
| | Urine | 800 | 238 | 368 | 200 | 59 |
| LSD (0.05) | | | 42 | 44 | 36 | 19 |

Autumn/winter grazed forage crops are a potentially large source of nitrate leaching losses from intensively managed crop sequences on well drained soils of Canterbury. The differences in nitrate leaching losses measured in this study were not closely linked to cumulative drainage or fertiliser N inputs and therefore imply that crop demand for N and the timing of sowing and grazing (i.e. urine applications) are important determinants of the nitrate leaching losses. On-going work seeks to identify crops and management practices that mitigate nitrate leaching losses from high production forage crop sequences.

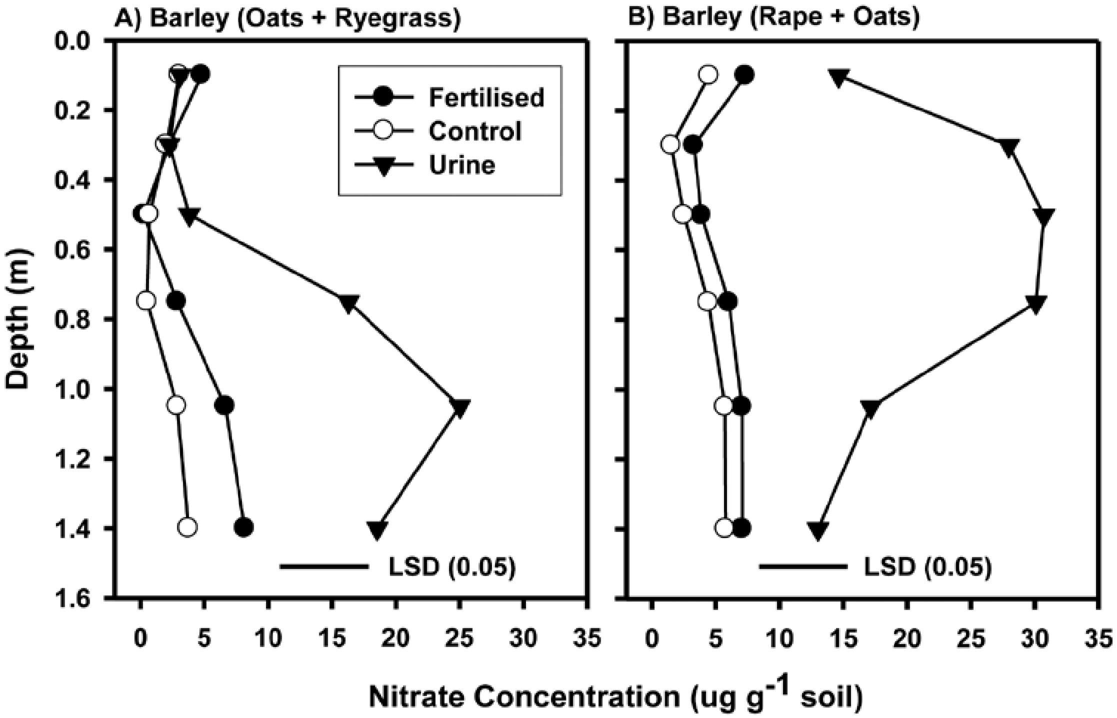


**Figure 1: Nitrate-N concentrations in the soil profile to a depth of 1.5 m following harvest of the A) Oats/Italian ryegrass and B) Forage rape/Oats grown following summer barley. The treatments include plots receiving fertiliser N, urine and unfertilised controls. Bars are LSD (0.05) for comparing treatment means within depth.**


## Acknowledgements

We are grateful to Andrew Fletcher, Richard Sim, Mike George and Peg Gosden for their scientific and technical contributions. Funding was provided by two MSI contracts: Delivering Tools for Improved Environmental Performance (C10X0603) and Lifting Farm Profit & Production while Reducing Environmental Footprint (C0X1117) with additional support from the Plant & Food Research's core funded *Land Use Change and Intensification* programme.


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# Nitrate leaching losses in site-specific nitrogen management corn crop (*Zea mays L*) in Inland Pampas, Argentina

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An average 10–30% of total N inputs in cropping systems are lost due to nitrate leaching. This has led to environmental contamination and concerns regarding use of N fertilizers. Development of alternative management strategies will be vital to sustaining cereal production systems. NLEAP model was developed for a rapid and accurate estimation of potential nitrate leaching lossess moving below the root zone, associated with agricultural practices. The objective of this study was to assess nitrate leaching losses in N fertilized corn crop with uniform and site-specific N technologies. N leaching losses were simulated with NLEAP model under different climatic scenarios (wet, dry and average). Soil nitrate-nitrogen and water content was determined in two moments: prior to N application and after harvest in field experiments (0-1,5 m depth) with uniform (UM) and site-specific N management (SSM). Several soil parameters were also determined under field conditions to run NLEAP. The residual nitrate content and the potential leaching losses were simulated with NLEAP and were compared with experimental data. Simulated data showed a high correlation with the observed values suggesting that NLEAP was capable to predict soil nitrate leaching under the studied conditions. In maize low-productivity zone, resource conservation was improved with SSM by reducing residual soil nitrate content that would be leached during a rainy season. SSM resulted in lower N leaching losses than UM under all climatic scenarios. We strongly encourage soil and water conservation scientists to continue looking for alternatives for managing future impacts to soil and water quality.

# Field scale nitrogen leaching and uptake from pasture irrigated with treated municipal effluent, Taupö, New Zealand

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Field scale nitrogen leaching and pasture uptake are measured at the Taupö land treatment site in the North Island of New Zealand. Municipal wastewater is irrigated onto land and pasture harvested in a cut and carry scheme. To increase nitrogen loading rates allowed under resource consent, from 550 to 650 kg N/ha/yr, Taupö District Council have established a five year trial where sectors of land are irrigated with varying effluent loads.

Forty eight intact barrel lysimeters (30 cm dia. x 43 cm depth) were installed flush within 30 ha of ryegrass pasture. Centre pivot travelling irrigators were programmed to slow down (more effluent per unit area) and speed up (less effluent per unit area) to apply the varying rates of effluent at each sector; nominally 450, 550 and 650 kg N/ha/yr. The amount of effluent nitrogen applied to land was calculated using irrigation volumes and weekly effluent nitrogen concentrations. Variation in effluent applied between each treatment was measured with a plastic rain gauge. Lysimeter leachate was pumped periodically and analysed with a discrete analyser for TN, $NH_4$-N and $NO_x$-N. To determine pasture uptake of nitrogen, grass growing within each lysimeter was collected before paddock harvest, dry matter determined and analysed for nitrogen content in a LECO furnace.

A second experiment was established to explore dry matter production and nitrogen uptake by pasture under different harvesting frequencies. Initial results indicate that changing pasture harvesting frequency, from once every 10 to every 5 weeks, increased pasture dry matter production and nitrogen removal.

# The effect of pasture species composition on nitrate leaching losses

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Nitrate-N ($NO_3$--N) leaching losses from New Zealand grazed pasture systems can be detrimental to water quality. It has been suggested that deep rooting pasture species could be used to reduce nitrate leaching losses from grazed pasture systems. The objective of this research project was to determine the effect of pasture species compositions on $NO_3$--N leaching losses. Fresh cow urine was applied to 32 undisturbed Templeton fine sandy loam soil monolith lysimeters to quantify $NO_3$--N leaching losses beneath four pasture species at the Lincoln University Research Dairy Farm. Pasture treatments were: (i) Perennial ryegrass (P. ryegrass) White clover (WC); (ii) Tall fescue (T. fescue) WC; (iii) Italian ryegrass (It. ryegrass) WC; and (iv) 'diverse' pasture – It. ryegrass P. ryegrass WC Red clover Chicory Plantain. The trial was run over a 2-year period. Nitrate-N leaching losses were lowest under the It. ryegrass WC pasture (24-25% less than under the P. ryegrass WC pasture) and highest under the T. fescue WC pasture (up to 39% greater than the P. ryegrass WC). Total dry matter production was highest in the It. ryegrass WC treatments and lowest in the T. fescue WC treatments. In conclusion, the deep-rooted T. fescue WC pasture is less effective at capturing soil N during the winter period than other shallow-rooted species because of its inactive growth during the cooler months. In contrast, the rapid winter growth pattern of It. ryegrass WC uses greater amounts of soil N during times of high drainage despite its' shallower root system, resulting in lower NO3--N leaching losses.

# Quantifying the effective area of a urine patch: The effect of concurrent 15N amended urine and fertiliser application on pasture response, and soil N dynamics

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A urine patch can be considered as comprising a 'wetted area' (where urine is directly voided) and an 'effective area' including the pasture outside the voided area that can access urine. The effective area can be more than twice the wetted area of a urine patch due to urine-N diffusion and plant root extension. Few studies have compared the wetted area of a urine patch to the effective area, and many leaching studies using lysimeters and field plots have not quantitatively accounted for pasture uptake and soil N dynamics in the effective area. We present data from a field trial where 15N amended urine and urea fertiliser treatments were applied concurrently. Pasture response and soil N was monitored from both the wetted area, and the effective area. There was a considerable pasture response (kg DM) within 0.25 m from the edge of a urine patch, which decreased with distance. The edge effect on soil N (ammonium) was less pronounced and reached 650 kg NH4-N ha-1 under a urine patch, while 0.25 m outside the urine patch it reached only 60 kg NH4-N ha-1. There was no effect observed >0.25 m from the wetted edge. The concurrent urine plus fertiliser treatment resulted in only slightly larger pasture yield and soil N than the urine only treatment. Nitrogen uptake from the effective area may have implications for N leaching loss calculations and N leaching models that are based on data where only the wetted area of a urine patch is accounted for.

# Monday 3 December

## SOIL CARBON

*Presented Posters*

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# Greenhouse gas emissions in saline, waterlogged soils of Western Australia

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## Abstract

Patterns of greenhouse gas emissions from saline soils are difficult to predict since low biological productivity combined with variable waterlogging regimes may cause contrasting effects on the emissions of $CO_2$, $N_2O$, $CH_4$ and $C_2H_2$. Moreover, there is a need to understand the effect of revegetation in recovering salinised/sodic soils, sequestration of soil C and thereby increase in biological activity and its effect on patterns of GHG emissions. An incubation study was conducted to examine the effect of organic material addition (salt bush leaf incorporated at 0 or 7.5 t $ha^{-1}$) to a saline soil (EC 1:5 58.7 mS $cm^{-1}$) while maintaining water levels at 0 (complete saturation), 10 cm or 15 cm below the soil surface. Low rates of $CO_2$ emission were found without organic matter addition although initially the highest rates were found when water level was 10 cm below the soil surface. With OM addition there was an immediate and sustained increase in $CO_2$ emission especially when the water level was at 10 or 15 cm depth. Acetylene ($C_2H_2$) emissions were 1-2 orders of magnitude lower than those of $CO_2$, but generally followed the same pattern of response. Rates of $C_2H_2$ emission were very low in unamended soil but six- to eightfold higher with added OM. Methane emissions were lower, being 1-2 orders of magnitude lower than that of acetylene. Addition of organic matter only intermittently stimulated emission of $CH_4$ with no significant effect of water level. In unamended soil, low rates of $N_2O$ emission were recorded. Addition of organic material increased the emission of $N_2O$ during the first 2-3 days after addition with the cumulative loss being highest with water level at 15 cm. Decreases in extractable $NH_4$-N and $NO_3$-N levels were recorded in soils amended with organic material and also in unamended soils after 15 days of incubation. Addition of organic material increased soil microbial biomass carbon, and organic carbon status increased after 15 days of incubation at varying water levels. Lack of C substrate was the dominant limitation for greenhouse gas emissions on saline soil, overriding the effects of soil water regime. The greenhouse gas emissions from the saline soils were dominated by $CO_2$ loss and were dependent on the input of decomposable C. These results suggest that revegetation by increasing available C substrate on saline soils may increase their greenhouse gas emissions.

# Contribution of dissolved organic carbon leaching to the carbon budget of a grazed pastoral system

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Soils are the largest terrestrial store of carbon (C) and changes in this store of C can impact on soil quality and atmospheric CO2 concentrations. Research on C budgets at paddock to national scales has focused most attention on the processes of respiration and photosynthesis in determining the net loss or gain of carbon from agricultural soils. However, leaching of dissolved organic carbon (DOC) is a potentially important component of the carbon budget that is rarely measured when developing carbon budgets, and as a consequence, is often estimated or ignored. The objective of this study is to quantify DOC leaching and to determine the fate of DOC under an intensively grazed pasture in New Zealand. DOC is being sampled in five paddocks using 100 suction cup lysimeters. These were installed within the footprints of two eddy covariance towers on a dairy farm in the Waikato (NZ). Water extracted from the suction cups is analysed for DOC and nitrate. These measurements are coupled to a water balance model created from water exchange data, including measured evaporation and rainfall, to determine the flux of C leached. This flux will be integrated into a total carbon budget determined by eddy covariance, and farm imports and exports. In order to understand the suite of processes that influence carbon dynamics in the soil, internal cycling process including mineralisation and sorption of DOC will be investigated in the laboratory and will help determine the fate of the leached material in the vadose zone.

# Increasing soil carbon storage in extensive grazing systems in temperate regions

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Degraded soils in extensive grazing systems have the potential to sequester significant amounts of carbon, and in doing so improve productivity and long-term sustainability. In this study we are investigating the effect on soil carbon of a range of commercially practicable grazing and pasture management strategies that aim to increase the persistence of perennial pasture. The strategies that have been implemented on farms across northern and southern Tasmania include: different periods of rest via rotation or exclusion of livestock; sowing of perennial pasture species; rehabilitation with native pasture and tree species; and application of organic amendments. For each trial site total and labile carbon is measured and a cross-fence comparison done with normal farm management.

This Carbon Farming Initiative, Action on the Ground project, is undertaken in collaboration with two NRM regions, Greening Australia and with co-operation from a major private agricultural service provider and the State Government. The results of the on-farm demonstration trials will be applicable to producers utilising dryland pastures for meat and wool production.

We will present initial findings from field work and existing datasets.

# Changes in soil organic carbon stocks following conversion of grassland to oil palm plantations

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Change to soil organic carbon (SOC) stocks under oil palm cultivation are poorly known, considering the rate of expansion in this crop and its potential as a biofuel. Changes in SOC stocks following conversion of grassland to oil palm are interesting because this conversion is argued to create net carbon storage. Additionally, stable carbon isotope ratios can differentiate the $C_3$ oil palm-derived SOC from $C_4$ grassland-derived SOC. Knowledge of soil carbon dynamics in oil palm plantations will help to manage soil quality and also enable more accurate assessments of greenhouse gas emissions associated with plantations. In this study, in Papua New Guinea, we measured SOC stocks in 16 plantations aged between 25 and 6 years and in the adjacent, original grassland. The rate of loss of grassland-derived SOC and accumulation of oil palm-derived SOC were calculated by isotope mass balance. The amount of recalcitrant 'black carbon' in the soil was determined using hydrogen pyrolysis. Soil respiration was measured and spatial and temporal variability in respiration was investigated. Initial results indicate that SOC stocks decline slightly over a 25-year period and SOC derived from the original vegetation remains dominant 25 years after conversion to oil palm. 'Black carbon' derived from regular grassland burning constituted approximately 5% of the SOC at 0-0.05m depth and up to 15% at 0.5-1.0m depth. Soil respiration is highly variable in the plantations in accordance with the oil palm root distribution and organic matter inputs. Soil respiration averaged 7.4 $\mu mol\ m^{-2}\ s^{-1}$ in studied plantations and 5.7 $\mu mol\ m^{-2}\ s^{-1}$ in grassland.

# Poultry litter biochar enhances and maintains nutrient content of a degraded red vertosol amended with cow manure and maize stubble on the north west slopes of NSW, Australia

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Research to date shows that biochar can influence nutrient dynamics in the soil. However, long term data to support these claims are lacking for Australian farming systems. Biochar made from poultry litter (PLB) is rich in phosphorus (P), nitrogen (N), carbon (C) and has been found to improve P availability in acidic soil and increase crop and pasture yields. This paper presents nutrient and soil respiration dynamics over 12 months from six mixtures containing combination of soil (red vertosol), cow manure (CM), maize stubble (MS), and PLB: (1) control (soil only); (2) soil+PLB; (3) soil+ CM; (4) soil+CM+PLB; (5) soil+MS; and (6) soil+MS+PLB. Each mixture was placed in a fine mesh bag and buried in the field within 10 cm depth. Bags were harvested at 6 and 12 months and soil samples were analysed for total C, N, available P, soil pH. Soil respiration rate was determined by incubating soil in a closed jar where $CO_2$ was trapped in NaOH solution, then titrated with HCl. Both PLB and CM had the same N content (1.6%) and a similar C content (36% and 39%, respectively). The available P in CM was highest (1.4 times that in PLB, and 5.4 times that in MS). At the start of the experiment, adding CM, MS and PLB to soil significantly increased total N, C, the available P and soil pH. After 6 months, total N and C, available P, and soil pH in mixtures without PLB approached those under the control treatment. In contrast, these nutrients remained significantly higher in mixtures containing PLB at 12 months. Amongst the mixtures containing PLB, combination with other organic amendments (CM or MS) enhanced total C, N and soil pH beyond that with PLB only treatment. The liming effect of PLB was more persistent than the organic amendments. Available soil P was increased by PLB and this persisted for 12 months. There were short term increases in available P in the CM and no change in the MS treatments. Soil respiration rates increased with the addition of all types of amendment, and it was highest in MS mixtures. The effect of PLB on soil respiration rate only became significant at 12 months. Soil amended with PLB alone had more constant soil respiration rate and carbon content during the monitoring period, whereas soil respiration and carbon content decreased with time for treatments which included organic amendments. This study is continuing and subsequent samples will be collected annually to evaluate the longevity of biochar effect in carbon and nutrient cycling from other organic matter on the North West Slopes of NSW.

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# An observational study of clay delving and its impact on the A2 horizon in sand over clay soils

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Sandy surfaced texture contrast soils in South Australia exhibit a range of limitations regarding their potential for agricultural production. The process of "delving" clay to the surface began in the south east of the state in order to address the limitation of water repellence. Clay delving was seen to mimic the water repellence ameliorating effects of clay spreading, only without the expense of importing the clay. Subsequently it has become clear that the delving process also addresses a number of other limitations associated with sandy soils. This paper discusses some of the improvements observed in the A2 horizons of 12 sand over clay soils. All of the sites exhibited bleached A2 horizons prior to the clay delving. Post clay delving, the region disturbed by the delving tine underwent varying degrees of mixing. This profoundly altered the chemistry and subsequent productive potential of the soil.

**Key words:** bleached A2 horizon; clay delving; clay spreading; infertile; yield

## Introduction

The use of clay applied to the surface of water repellent sands has been practiced by South Australian farmers since the 1970s (Cann 2000). Most of the sands were spread with clay in order to treat a surface horizon that was water repellent due to the presence of hydrophobic organic material, as described in Ma'shum (1989). This water repellence was usually the principle soil limitation cited by farmers undertaking the clay spreading. The practice was widely taken up as producers saw immediate yield gains across a range of sandy surfaced soils. In most cases clay was sourced from pits within a few hundred meters of the spreading site. Once spread (average rate of around 250 tonnes of clay per ha), the clay was generally smeared onto the soil using heavy metal bars, and cultivated into the soil, usually to a depth of no more than 10 cm.

Clay delving began in the South East of South Australia as an attempt to access clay more cost effectively for treating water repellence. Where sandy surface horizons overlay clayey B horizons within around half a meter of the surface, clay could be accessed using modified ripping tines. This involves metal tines (usually about 1 to 1.5m long) set at an angle designed to peel the clay from the B horizon, to travel up the tine and spill onto the soil surface. Desbiolles (1997) provides a detailed description of delving operations and outlines a general design for the machinery used.

The first trials of the technique, including the first prototypes of machines to bring up the clay, were first undertaken in the early to mid 1990's (Groocock, personal communication). Since this time it has become evident that the improvements to soil condition derived from clay delving greatly exceed those from merely overcoming water repellence. One of the main impacts appears to be the alteration of the A2 horizon, which is often a bleached and highly infertile sand in its natural state. Following clay delving, the A2 is often blended with A1 and B2 materials, profoundly altering its clay content, hydraulic properties and nutrition status. This study aimed to assess these changes by measuring chemical soil properties and observational soil description and root growth over a range of situations in the field.

## Method

Comparative field descriptions and chemical analysis were made of 12 delved soil profiles (exposed in approximately 2 m wide x 1.5 m deep pits). The soils in their natural state represented a range of Brown Chromosols (2), Red Chromosols (1), Brown Sodosols (2) and Yellow Sodosols (7) (Isbell 2003), with sandy A horizons ranging from 15 cm to 60 cm thick (median A thickness was approximately 35 cm). All exhibited strongly bleached A2 horizons (A2 cation exchange capacities ranged from 0.7 to 3.2 cmol(+)/kg). Sharp texture contrasts were characteristic of all sites, with most B2 horizons being mottled and coarsely columnar. The horizon notation used in this paper is from McDonald *et al.* 1990.

All the sites had undergone clay delving, using a range of locally available delving machinery. The time since the delving had taken place prior to sampling varied from six months to seven years. The spacing between the delving tines varied from around 1 m to around 1.8 m (median of 1.5 m), so all sites exhibited strips where the soil had been disturbed in the subsurface and subsoil (along the delve lines), and strips where it had not (between the delve lines).

The A1p of all the sites had been relatively homogenised through the smearing and cultivating of the surface applied clay. Consequently, the main comparisons made in this paper comprise the A2 of the soil where the delving tine had not affected soil morphology compared to where the analogous layer that had been modified by the tine. The A2 material where delving had not induced morphological changes was assumed to be reasonably representative of the A2 in its native state, prior to delving taking place. Hand sampling of the A2 was based on segregating the horizon into three readily identifiable soil materials (refer to Table 1). These were the bleached sand of the A2 unaffected by the delve tine, the sandy material from the zone of the A2 mixed by the delving tine, and the larger lumps of clay within the sandy matrix of the delve affected A2.

## Results

*Observed soil changes*

The effect of delving, in terms of the amount of soil disturbance and inversion that was observed, was highly variable between sites. It is probable that differences in design of the delving implement, the soil type, and the moisture content of the soil all contributed to this variability (May 2006). However, of the 12 sites discussed here, all portrayed significant morphological changes to the A1, A2 and B2 horizons.

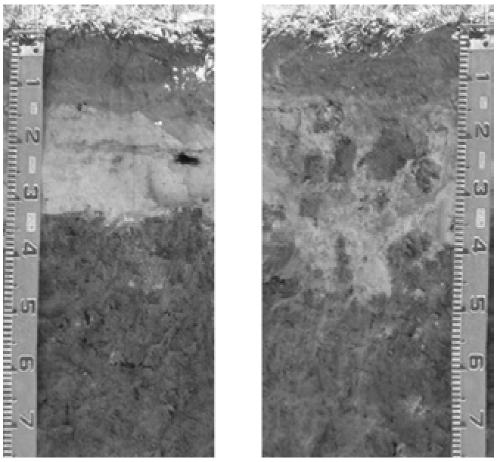

**Figure 1: Brown Sodosol- Left hand side shows the profile with the A2 and B2 unaffected by the mixing action of delving. The right hand side shows the A2 and top of B2 affected by delving. The two photos were taken immediately adjacent to each other from the same soil pit.**

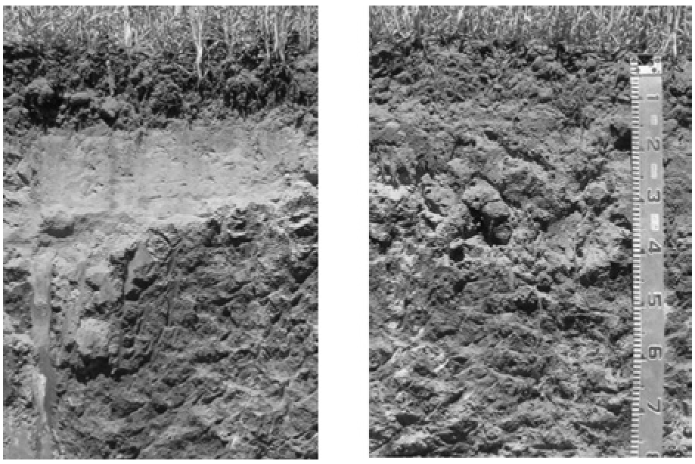

**Figure 2:** Red Chromosol- Left hand side shows the profile with the A2 and B2 unaffected by the mixing action of delving. The right hand side shows the A2 and top of B2 affected by delving. The two photos were taken immediately adjacent to each other from the same soil pit.

The results of the morphological changes are represented in Figures 1 and 2. All four photographs are of soils that have been delved. The A1 horizons of each of these soils has effectively been clay spread, with the clay brought to the surface via delving having been levelled out uniformly and cultivated into the A1p. The differences illustrated in the two sets of photographs relate to the zone of soil mixed by the delving tine, in contrast with the soil to each side of this mixed zone. Depending on the spacing between the delver tines, the soil with a mixed zone has been observed to comprise between around 10% to over 60% of the soil area within a delved paddock.

*Effects of delving on A2 chemistry*

The comparative measurements made for this paper were from the A2 horizons of soils to determine the changes relating to soil mixing compared to the relatively unaltered bleached A2 horizons adjacent to the delved zone.

**Table 1: Differences in a range of chemical analytes were measured between the identifiable soil materials in the A2 horizons of the 12 delved soils (standard deviations given in brackets).**

| Mean A2 soil material extractable chemistry | | | | |
|---|---|---|---|---|
| | Undelved | Delved (sandy material) | Delved (clay lump) | Frequency of sites where results from undelved sand were lower than delved sand (%) |
| P (Colwell) mg/kg | 7.3 (5.5) | 12.1 (10.0) | 3.9 (3.1) | 75 |
| K (Colwell) mg/kg | 39.5 (18.4) | 60.9 (29.8) | 348.3 (140.7) | 92 |
| S (KCl-40) mg/kg | 3.9 (4.7) | 7.3 (7.6) | 17.6 (18.7) | 92 |
| $NO_3^-$ (KCl) mg/kg | 2.9 (2.1) | 6.1 (4.5) | 10.1 (8.2) | 67 |
| Org Carbon (W/B) % | 0.3 (0.1) | 0.9 (0.4) | 0.7 (0.2) | 100 |
| Reactive Fe (Tamms) mg/kg | 230 (114) | 378 (182) | 768 (257) | 92 |
| CEC ($NH_4Cl$/ $BaCl_2$) cmol(+)/kg | 2.0 (0.9) | 3.7 (2.0) | 16.1 (4.0) | 92 |
| pHCa | 5.9 (1.0) | 5.6 (0.9) | 6.3 (0.5) | 27 |
| ETDA Cu mg/kg | 0.6 (0.7) | 2.8 (2.5) | 0.9 (0.8) | 75 |
| ETDA Zn mg/kg | 0.8 (1.4) | 1.5 (1.3) | 0.7 (0.4) | 83 |
| ETDA Mn mg/kg | 1.4 (1.1) | 3.8 (3.0) | 2.7 (1.6) | 100 |
| ETDA Fe mg/kg | 70.3 (34.3) | 136.5 (72.3) | 76.4 (51.6) | 92 |
| Boron ($CaCl_2$) mg/kg | 0.4 (0.1) | 0.6 (0.1) | 2.2 (0.6) | 92 |

For all the soil chemistry and nutrition measurements made on the soil materials from the A2, the sandy soil materials visually unaffected by delving were, when averaged, all lower than the sandy material mixed through delving (except for pH). Table 1 shows a summary of the results from soil tests of the unaffected sand, the sandy material mixed through delving, and the coarse clay lumps present in the delved A2. The exception to the trend was soil pH, with eight out of the 11[1] sites measured for pH showing a decrease in pH in the sandy material mixed through delving compared to the undisturbed A2 horizon sand. However, the coarse clay lumps brought into the A2 were generally of a higher pH than both the other identified materials.

## Discussion

Delving of bleached sand over clay soils has become a common practice in South Australia over the last 15 years. However, aside from a few papers such as those referenced here, there is very little published on this topic within the scientific literature. Publications relating to clay spreading and to deep ripping, while presenting some relationship to the topic of clay delving, generally do not cover the mixing of A1 and B materials with the A2 horizon (for example, Hall *et al.* 2010). This paper does not attempt to investigate the topics usually presented when discussing ripping or clay spreading, such as non-wetting surface soils, or

[1] One site was irrigated with alkaline ground water, and subsequent pH data was excluded from this analysis.

hard pans, but instead focuses upon the induced changes in soil morphology and chemistry within the A2 horizon.

Observations of the soil profile of delved paddocks show profound changes to the A1 horizon of the soil, analogous to clay spreading. In addition to this, the A2 and upper B horizons of the soil are also significantly altered along the path of travel of the delving tine. Comparisons made between the A2 that has been visually unaltered by the delving, with that of the adjacent soil profile mixed by delving, show stark changes in the A2 properties. In most cases, extractable macro and trace elements are extracted in higher quantities in the mixed A2 soil material. The same is true for the cation exchange capacity of the mixed soil. Soil pH is less clear, however, it appears that the soil mixing may reduce the pH of the sandy material in the mixed A2.

The limited measurements that have been made on plant yield also suggest the subsurface and subsoil mixing has a marked impact on productivity (Rebbeck 2007; Bailey unpublished data). Observations of root abundance have also found marked increases in roots in the mixed A2 (Bailey unpublished data).

## Conclusion

Clay delving can result in significant changes to many of the properties of a soils A2 horizon. These potentially include changes to a range of physical, biological and chemical attributes. This paper presents preliminary findings relating to the latter attributes, with indications being that the former two attribute groups are also worthy of further study in relation to affects of clay delving. The main finding of this study has been that, where the soil has been mixed by the delving tine, the A2 is generally enriched in its inherent fertility and its extractable nutrition. The source of these changes are likely to stem from both the mixing of clayey B horizon and the sandy A1 horizon with the low fertility A2 horizon material.

## Acknowledgments

We wish to thank Roger Groocock for his practical knowledge of clay delving and for organising many of the sites.

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# In search of new approaches to furrow irrigation

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Furrow irrigation is the most common method of irrigation used worldwide. However, as global food requirements increase, new ways need to be found to boost the productivity of furrow irrigated agriculture, and reduce associated environmental impacts. Unwanted impacts include rising water tables, degradation of soil ecological infrastructure, salinisation of soil and groundwater, and surface and groundwater pollution. There is also a growing societal expectation for improved water and nutrient use efficiency and reduced deep drainage and leaching. This paper discusses results of a simulation study carried out using the HYDRUS-2D model. This study analysed water flow and non-reactive solute transport through a cross-sectional furrow, which was subjected to different combinations of soil surface management strategies and nitrogen fertilizer placements. Soil management strategies included: leaving the soil in its normal state as the control; compacting the bottom of the furrow; and covering the bottom of the furrow with plastic. The greatest potential benefit, in terms of water and fertilizer use, was achieved by covering the bottom of the furrow with plastic to force the applied water directly into the ridge. This helped maximise the benefit of capillary forces in retaining water and nitrogen within the plant root zone, while minimising the impact of gravitational forces that draw water and nitrogen below the root zone. We discuss opportunities and constraints associated with applying these approaches in developed and developing countries, and conclude by highlighting areas where further research is needed to progress the transformation of furrow irrigation into more productive and environmentally sustainable irrigation systems.

# Impact of soil physical structure on shallow- and deep-rooted crops grown on a fluvial silt loam in Hawke's Bay, New Zealand

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Soil physical constraints can limit crop growth by reducing the movement of air and water through the soil matrix and impeding root exploration. We conducted two trials to quantify the impact of varying amounts of compaction on the above- and below-ground development of a deep rooted maize crop (*Zea mays*) and a shallow rooted onion crop (*Allium cepa*). Both experiments were established on Mangateretere silt loams (Typic Haplaquept) in Hawke's Bay, New Zealand.

In the maize trial, compaction was created before sowing with a tractor-planter unit (~ 8 t), mimicking the type of physical constraint often encountered in headland areas of a paddock. There were four paired plots comparing these compact areas to uncompacted controls. In the onion trial, compaction was also created with a tractor unit (~ 5 t). However, before sowing a power harrow was used to alleviate compacted areas to three depths (0.05, 0.10 and 0.15 m), mimicking the impact of harvesting crops in wet autumn conditions and then using restorative cultivation the following spring. An uncompacted control was included, and all four treatments were replicated four times. In both trials, maize and onion crops received standard agronomic management in line with commercial practice.

Regular measurements of crop biomass, leaf development and soil penetration resistance were recorded in both trials. At harvest soil cores were removed from key depths and used to calculate soil bulk density and macroporosity. In our talk we will present the links between these physical attributes and indicators of crop performance and discuss the implications for soil management.

# Changes in soil physical quality and land use within the Auckland region

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As part of State of Environment reporting, regional councils have the responsibility to undertake soil quality monitoring within their designated regions. Of concern within the Auckland Region is the trend of decreasing soil physical quality, particularly for soil macroporosity which continuingly falls below recommended national guidelines for grazed pasture. This has prompted a more detailed study undertaken in 2011 to resample the original 35 sites, first sampled in 1995/00 and repeated in 2009/10, to determine whether these national guidelines are realistic for soil types representative of the Auckland region. Comparisons were made between different land uses with soil samples taken underneath adjacent/undisturbed fence lines at each site. Macroporosity was significantly less for grazed (disturbed) than ungrazed (undisturbed) treatments. However, macroporosity in the ungrazed treatment, was only just above recommended guidelines for a prominent Soil Order that represents >40% of soils within the Auckland region.

As a result of land use change, what were originally described as two land use categories; dairy (n=19) and drystock (n=16), increased to four categories; dairy (n=6), dairy-drystock conversion (n=8), drystock (n=10), and lifestyle (n=11), the latter reflecting dairy and drystock conversions to lifestyle blocks. Although 83% of dairy sites occupied what would be considered prime land, a large proportion of prime land was occupied by lifestyle sites (64%), followed dairy-drystock (38%) and drystock (30%). Such information raises two concerns; 1) that a large proportion of prime soil in the region is not being used for primary production, 2) the difficulty in developing temporal soil quality trends under intensive land use as a result of the considerable land use change that is being experienced within the Auckland region.

# The effect of soil aeration and natural recovery on soil structure of a Pallic soil following winter grazing of sheep and cattle

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The North Otago Rolling Downlands (NORD) has undergone considerable land-use change and intensification following the greater use of irrigation and increased pasture production, fertiliser input and animal stocking rates of both cattle and sheep. A decline in soil physical quality associated with livestock grazing has been reported in a number of studies that have been based in this region. Once compacted, soil structure may be restored through a process of natural recovery facilitated by wetting and drying, freeze-thawing, pasture growth and earthworm activity. These processes however tend to be limited to the top 0-5 cm of soil where moisture fluctuations, pasture root growth and biological activity are often greater than at lower soil depths. Mechanical aeration has been used to ameliorate the adverse effects of soil compaction by loosening top-soils to greater depth within the soil profile. Here we report findings from a trial that was established in the NORD region to compare the rates of soil recovery (as determined by increases in soil macroporosity values) via natural or mechanical means. This assessment was made at a trial site where four years of winter forage cropping under cattle or sheep grazing had caused a significant amount of soil compaction.

Six months after mechanical aeration to 20 cm soil depth, soil macroporosity levels at depths of 0-5 and 5-10 cm were significantly greater in aerated soils relative to the non-aerated treatment. This was apparent under both cattle and sheep grazed pastures. However, no difference was evident eighteen months after aeration. Higher pasture growth rates were measured in the aerated treatment during the first year following aeration. Results indicate that mechanical aeration provides an immediate improvement in porosity of compacted soils, although this benefit appears to be short-lived.

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## To what extent has soil drying induced earlier grape ripening in wine regions of southern Australia?

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Using continental-scale modelling, Webb *et al*. (2012)* attributed the trend in early ripening of wine grapes in five viticultural regions of southern Australia to increasing growing season temperatures and soil drying linked to global warming, and crop management. However, the allocation of only one soil class to each 5 x 5 km cell, for which key soil properties such as hydraulic conductivity were derived from pedotransfer functions, was at too coarse a scale for the experimental vineyards of 0.2-16 ha. Further, if prolonged soil drying had occurred, this should be correlated with a decrease in annual rainfall, given that actual evapotranspiration should change little as the effect of increased carbon dioxide concentration in the atmosphere balances the effect of ambient temperature increase. Analysis of the longest, most complete Bureau of Meteorology records for each vineyard site showed a highly significant decrease in rainfall (27 mm/decade) at Margaret River, consistent with the Bureau's record for southwest Western Australia, but paradoxically no significant early ripening. Conversely, significant earlier ripening was observed for the vineyard on the Mornington Peninsula, even though there was a significant increase in rainfall in the long term (8.5 mm/decade). Clearly, while growing season temperatures and crop management to reduce yields may have contributed to earlier ripening at the experimental vineyards, the case for an effect of prolonged soil drying, based on continental scale modelling, was unconvincing.

*Webb LB, Whetton PH, Bhend J, Derbyshire R, Briggs PR, Barlow EWR (2012) Earlier wine-grape ripening driven by climatic warming and drying and management practices. *Nature Climate Change* **2**, 259-264.

# The intrinsic energy of soil aggregates

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Soil aggregates are a key indicator of sustainable soil systems but an absolute measure has been elusive. Because soil aggregates are not random, but they represent a unique organised state that is comprised of its mineral particles and colloids as building blocks bound together by clay minerals and organic matter. Therefore, soil aggregates exist because there is an *intrinsic energy* that binds the particles together and is the energy required to maintain soil aggregation resulting in an organised soil system. Modelling soil aggregate liberation and subsequent dispersion (ALDC) in response to the actual energy involved in dispersion provides us with an absolute measure of the relative aggregate stability and kinetics of aggregate breakdown. Using soil types that represent different mineral and colloid composition and quantities it is shown that the resulting aggregates have different amounts of intrinsic energy to maintain their existence, and therefore the integral of aggregates energy partially represents the amount of energy that has been accumulated at that point in time.

# Efficiency and efficacy of drip irrigation in a Tasmanian vineyard

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### Key Words

Water use efficiency, drip irrigation performance, dye tracer, surge irrigation, preferential flow

### Abstract

Irrigation performance was determined by dye tracer staining, soil hydraulic assessment and modelling. The dye tracer studies demonstrated that infiltration through the topsoil was almost entirely vertical with little if any lateral water spread below the saturated area of ponding directly beneath the dripper. However in the subsoil, irrigation spread laterally between drippers via macropores rather than as saturated flow on top of the clay subsoil. Irrigation application uniformity was very low as the majority of the root zone was not wetted by irrigation. Pulse irrigation increased the proportion of the topsoil wetted by irrigation, although pulse irrigation was also associated with greater preferential flow beneath the rootzone. This study demonstrated that use of dye tracers is an effective, simple means of evaluating drip irrigation performance.

### Introduction

In recent years drought and water scarcity have impacted on viticulture production in many wine growing regions of Australia. Viticulturalists are under increasing pressure to improve irrigation performance, however little is known about the efficiency and efficacy of drip irrigation for viticulture in Tasmanian soils. The lack of data on wetting patterns, effective root zones and soil hydrology has resulted in a dearth of information to assist irrigators establish evidence-based drip irrigation management practices. In this paper we present novel field and modelling approaches for determining the irrigation efficiency and efficacy of drip irrigation practices.

### Methods and Materials

Brilliant blue food dye tracer was applied to 10 separate drippers in an 8 year old block of Sauvignon Blanc at Frogmore Creek Vineyard, Tasmania. The site has a Mediterranean climate, mean annual rainfall of the Cambridge area is 502 mm with a mean daily evaporation of 3.6 mm (Bureau of Meteorology 2011). The soil profile consisted of a sandy clay loam (18 % clay, 68% sand) A1 horizon (0-20 cm depth) which overlayed a thin, sandy, discontinuous (0-5 cm thick) A2e horizon (7 % clay, 82% sand), which had a sharp boundary to the mottled clay B2 horizon (45-46 % clay, 40-42% sand).

Flow rate and application volume for each dripper was measured. Irrigation with slightly saline water (0.6-0.8 dS/m) was applied continuously to five drippers for two hours at 3.47 L/hr and as a pulse consisting of consecutive on/off periods over a 200 minute period at 4 L/hr. The irrigated area was excavated (approximately 1.5 m long x 0.8 m wide x 1.2 m deep) 2 days after irrigation to enable the dye staining flow paths to be visualised. Digital images of the infiltration pathways were taken, corrected for radial and keystone distortion and converted to binary images using Photoshop CS3 and Image J software (Hardie *et al.* 2011).

Soil hydraulic properties were determined by *in situ* infiltration by tension infiltrometers at 5 supply potentials. Saturated hydraulic conductivity was estimated by extrapolation of the $K(\psi)$ relationship to zero using RETC. The soil water retention function was solved for van Genuchten–Mualem model by the Rosetta pedotransfer function and by inverse solution of cumulative tension infiltration data (Simunek and van Genuchten 1997) with the soil water content at -1000 kPa determined by pressure chamber analysis. Drip irrigation events were simulated using HYDRUS 2D/3D to explore the effects of initial moisture content and dripper spacing on irrigation scheduling and performance.

**Results and Discussion**

In the A1 and A2 soil horizons irrigation infiltration was almost entirely vertical, with no lateral movement of irrigation into the surrounding soil. In the B1 horizon, irrigation infiltrated both laterally and vertically through a continuous network of macropores. Preferential flow via sand infills was also evident in a number of locations (Figure 1a).

Irrigation application uniformity (evenness of dripper output) was also very low averaging 63%. Continuous irrigation wet only about 10% of the target root zone (40 cm deep x 125 cm wide), while pulse irrigation resulted in approximately 14% of the target root zone being wetted (Figure 1b). Consequently between 85% and 90% of the target rootzone was not wetted by irrigation.

Pulse irrigation resulted in greater wetting of the topsoil (Figure 1b), and thus had greater efficacy than the continuous irrigation. However pulse irrigation was associated with greater preferential flow below the root zone which resulted in lower irrigation efficiency.

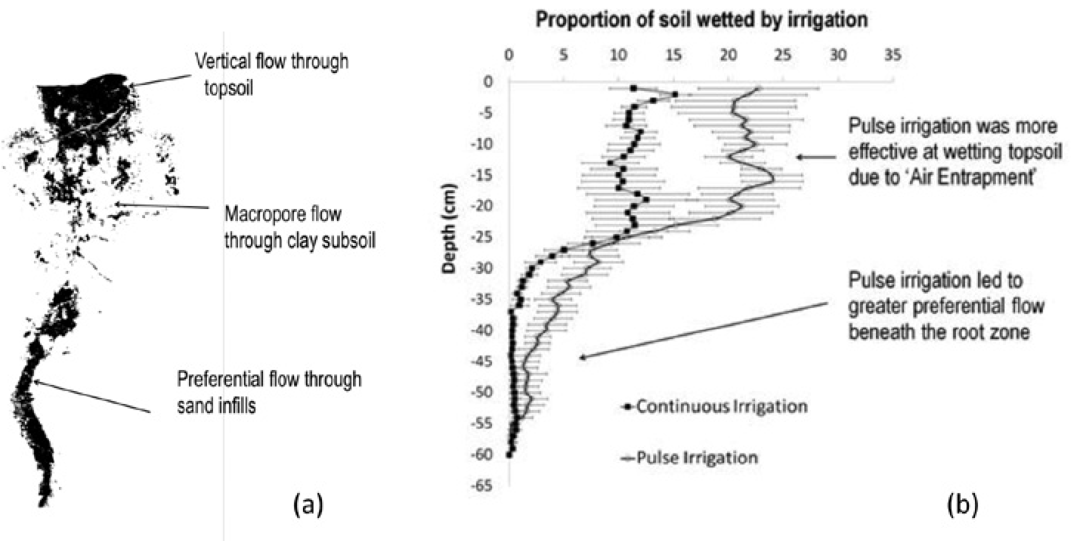


**Figure 1. (a) Example dye tracer stained infiltration pathways, 35 cm x 65 cm. (b) Effect of pulse and continuous irrigation on the proportion of soil wetted by irrigation.**

The presence of preferential flow in the subsoil resulted in discrepancies between the irrigation simulations and the dye tracer studies. The pedotransfer approach predicted higher saturated water content (Qs) and $\alpha$ than the inverse solution approach (Table 1) which indicated the soil had lower functional macroporosity in both the A1 and B21 horizons than would otherwise be expected.

Modelling demonstrated that the current practice of placing drippers 45 cm from the vine trunk required 2-3 hr irrigation for the wetting front to penetrate to 30 cm depth beneath the vine. Whereas when drippers were placed immediately beside the vine trunk the wetting front penetrated to 30 cm depth in approximately 20 minutes.

Table 1. van Genuchten parameters determined by inverse infiltration of tension infiltration data and the Rosetta pedotransfer model

| | Parameters estimated by inverse parametisation of tension infiltration data in HYDRUS 2D | | | | | Parameters estimated by the 'Rosetta' pedotransfer model | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Horizon | QR | Qs | $\alpha$ | n | Ksat (cm/min) | QR | Qs | $\alpha$ | n | Ksat (cm/min) |
| A | 0.100 | 0.380 | 0.019 | 1.230 | 0.180 | 0.066 | 0.524 | 0.024 | 1.406 | 0.104 |
| A2 | | | | | | 0.047 | 0.382 | 0.038 | 1.855 | 0.083 |
| B1 | 0.160 | 0.380 | 0.027 | 1.300 | 0.080 | 0.091 | 0.449 | 0.022 | 1.261 | 0.010 |
| B2 | 0.160 | 0.380 | 0.027 | 1.300 | 0.080 | 0.085 | 0.403 | 0.023 | 1.200 | 0.005 |

## Conclusion

The current irrigation practice was not able to adequately wet the root zone. Drippers need to be closer to effectively wet the topsoil. The benefits from pulse irrigation were minor. Consideration should be given to converting drippers to microspray irrigation in order to improve the uniformity of topsoil wetting.

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# Consequences of soil crust formation on soil hydraulic properties and irrigation performance

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**Abstract**

Rainfall and irrigation resulted in the formation of severe surface soil crusts. Crust formation resulted in significantly higher bulk density and significantly lower hydraulic conductivity including loss of nearly all macroporosity in the surface soil layers. Crust formation was accompanied by resettlement of the cultivated beds which also resulted in significantly higher bulk density and significantly lower unsaturated hydraulic conductivity. Crust formation and bed resettling prevented irrigation wetting up the crop seed bed, and increased the proportion of irrigation lost by runoff from 0.5 % to 35 %. Inverse simulation of cumulative tension infiltration data enabled the soil water characteristic of the surface crust to be determined and the effect of crust formation on infiltration modelled in HYDRUS 2D.

**Key Words**

Crusting, hydraulic conductivity, irrigation efficiency, surface seal

## Introduction

Soil crusts are dense compacted layers that form on the surface of bare soils as a result of rainfall or irrigation droplets impacting the soil surface (Lehrsch and Kincaid 2006). The formation of a soil crust impedes the germination and emergence of seedlings. Soil crusts have higher bulk density than the underlying soil, as they have been 'compacted' during formation (Moss 1991). Soil crusts reduce the soil infiltration rate resulting in higher runoff and erosion whilst decreasing water use efficiency and the opportunity to store rainfall and irrigation for crop growth.

In packet salad production systems, frequent cultivation, low soil organic matter content and lack of crop residues have led to development of severe crusts following rainfall or irrigation. Options to reduce crusting through the use of smaller droplet size irrigation are limited by frequent high winds in the region. Observations indicate that crust development reduces infiltration, such that prolonged irrigation is required to 'wet up' the root zone (0-5 cm). This results in greater runoff and erosion than would otherwise occur without the presence of crusts. This study sought to determine the effect of (i) irrigation on crust formation, (ii) crust formation on soil hydraulic properties, and (iii) crust development on irrigation performance.

## Methods and Materials

The trial was conducted near Cambridge, Tasmania (E537653 N5259099), which is described as having a Mediterranean type climate, with 498 mm mean annual rainfall, and mean diurnal January temperature range between 12 and 22$^0$C (Bureau of Meteorology 2009). The trial site was established on a Yellow Chromosol (Isbell 1996), consisting of a 0-12 cm deep 5YR 4/2 grey-brown clay loam topsoil, over a 2.5Y 5/4 olive yellow medium clay. Soils on the property are degraded and erosive resulting from loss of soil organic carbon as a consequence of frequent cultivation and the absence of green manure inputs or pasture leys.

The effect of crust formation on soil hydraulic properties was determined on the crust and near surface soil prior to the first irrigation (day 1) and periods early (day 11) and late (day 59) in the crop season. Bulk density was determined for the topsoil using the core method, as well as the water replacement method described by Cresswell and Hamilton (2002). Infiltration of the crust surface and near surface soil was determined by tension infiltrometers (disk permeameters) at five supply potentials. Saturated hydraulic conductivity was estimated by extrapolation of the K(ψ) relationship to zero using RETC. The soil water retention function was solved for the van Genuchten–Mualem equation by inverse solution of cumulative tension infiltration data in HYDRUS-2D (Simunek and van Genuchten 1997).

Paired plot treatments (15 m x 3 beds) were established immediately after bed formation and crop sowing to compare the effect of irrigation droplet size on crust formation and soil hydraulic properties. Two irrigation events (water EC 0.4-0.6 dS/m) were monitored for runoff, droplet size, sprinkler uniformity, and depth of wetting on days 2 and 67 after sowing. Irrigation events were simulated in HYDRUS-1D to determine the effect of soil crust development on irrigation performance and to optimise irrigations for the site.

**Results**

The bulk density of the soil crust increased significantly from 1.11 (± 0.13) $g/cm^3$ (day 1) to 1.75 (± 0.12) $g/cm^3$ (day 59) over the trial period. However the density of the near-surface soil (immediately below the soil crust) also increased significantly over the trial period from 1.22 (± 0.02) $g/cm^3$ to 1.32 (± 0.03) $g/cm^3$ due to settlement and consolidation of the beds (Figure 1).

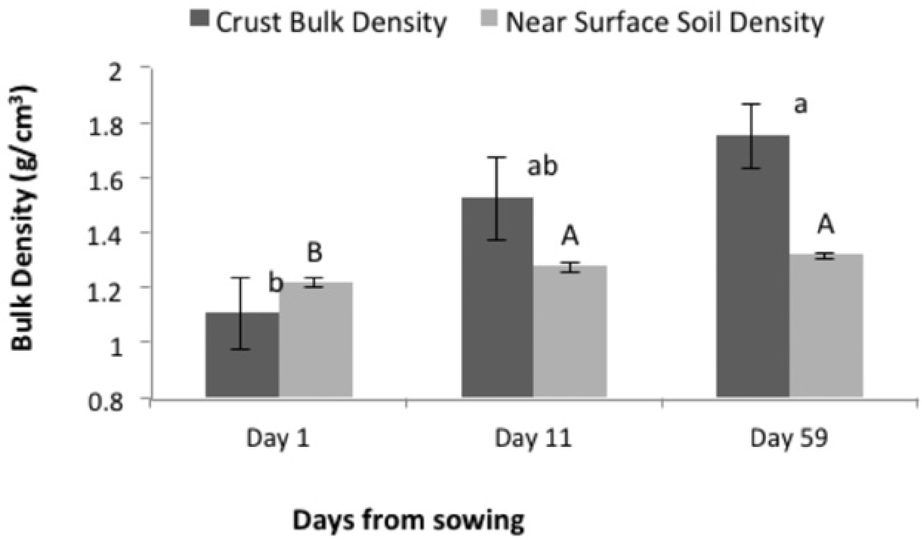


**Figure 1. Change in Bulk density over time. Different letters represent significant difference (P <0.05).**

Determination of hydraulic conductivity on day 1 at supply potentials between -0.50 and -0.10 kPa failed due to increased soil moisture and the weight of the apparatus causing the beds to collapse and consolidate during infiltration. At the ψ -0.60 kPa supply potential, infiltration decreased from 11.72 (± 2.76) mm $hr^{-1}$ at day 1 to 4.24 (± 1.13) mm $hr^{-1}$ at day 11 and 1.58 (± 0.68) mm $hr^{-1}$ by day 59. By day 11 the unsaturated hydraulic conductivity (Kunsat) of the surface crust was significantly lower than that of the near surface soil at all supply potentials (Table 1). By day 59 Kunsat of the soil crust had decreased significantly from 2.13 (± 0.12) mm $hr^{-1}$ (day 11) to 0.54 (± 0.16) mm $hr^{-1}$ at ψ -0.55 kPa, and from 11.53 (± 1.10) mm $hr^{-1}$ (day 11) to 2.66 (± 0.65) mm $hr^{-1}$ at ψ -0.15 kPa (Table 1). Differences in infiltration between day 1 and day 59 at ψ – 0.60 kPa and the minimal difference in Kunsat between ψ – 0.55 (0.54 mm $hr^{-1}$) and -0.15 kPa (2.66 mm $hr^{-1}$) at day 59 demonstrated that crust formation reduced the contribution to total flow from both large macropores (>500 µm equivalent to 0.06 kPa) and pores less than 500µm.

Reduction in infiltration and unsaturated hydraulic conductivity in the soil crust was also accompanied by a significant reduction in the infiltration and unsaturated hydraulic conductivity of the near-surface soil (just below the soil crust). The Kunsat of the near-surface soil at ψ -0.15 kPa decreased significantly from 34.64 mm $hr^{-1}$ on Day 11 to 8.91 mm $hr^{-1}$ on Day 59.

**Table 1. Effect of crust development and bed consolidation on unsaturated hydraulic conductivity (mm $hr^{-1}$) of the soil crusts and near surface soil**

| | | Surface crust | | Near surface soil | |
|---|---|---|---|---|---|
| Supply potential (ψ) | Day 1 | Day 11 | Day 59 | Day 11 | Day 59 |
| -0.55 kPa | 2.87 | 2.13a | 0.54b | 5.14 | 0.21ab |
| -0.40 kPa | na | 4.21a | 0.74b | 11.05 | 1.50b |
| -0.25 kPa | na | 7.01a | 1.70b | 20.47 | 3.45b |
| -.015 kPa | na | 11.54a | 2.66b | 34.64 | 8.91a |
| Estimated Ksat. 0 kPa | na | 41.82 | 6.36 | 97.62 | 82.14 |

Different postscript letters indicate significance at $P<0.05$

Irrigation nozzle type had no effect on droplet size, crust density or irrigation application uniformity. Time to first runoff and runoff volume differed significantly between day 2 (0.62 ± 0.34 L) and day 67 (40.82 ± 5.57 L). Total runoff from irrigation on day 2 prior to crust formation and bed settling resulted in an average of 1.87 L runoff from the 75 $m^2$ plots or 0.52 % of total irrigation as runoff. At day 67 crust formation and bed settlement resulted in 122.5 L runoff or 34.9 % of total irrigation as runoff (Table 2). Crust formation and loss of structure within the beds also meant that irrigation on day 59 was not able to wet-up the root zone to a depth of 5 cm. Simulation of the irrigation events indicated that increased runoff on day 63 was not related to differences in initial soil moisture content. However simulation also indicated that the majority of runoff (83 %) originated from the furrows not the crop beds.

**Table 2. Effect of crust development and bed consolidation on irrigation performance**

| | Day 2 – initial irrigation | | Day 67 – final irrigation | |
|---|---|---|---|---|
| Site | A | B | A | B |
| Drop size (mm) | 1.03 (0.14) | 0.99 (0.10) | 1.29 (0.27) | 1.34 (0.19) |
| Uniformity (%) | 72.85 | 66.53 | 85.96 | 90.31 |
| Initial moisture (θv) | 0.15 | 0.14 | 0.18 (0.02) | 0.18 |
| Runoff (L) total | 1.52 | 2.22 | 129.13 | 115.85 |
| Runoff (L) furrow average | 0.51 (0.06) | 0.74 (0.74) | 43.04 (9.39) | 38.62 (8.21) |
| Runoff as % of irrigation | 0.45 | 0.58 | 34.9 | 48.1 |
| Final moisture (θv) | 0.18 | 0.176 | 0.2 (0.03) | 0.19 (0.02) |
| Time to first runoff (mins) | 16.17 (1.04) | 26 | 9.05 (3.47) | 12.01 (3.56) |

Values in brackets indicate ± 1 standard deviation.

## Conclusion

Irrigation and rainfall significantly increased the bulk density of soil crusts which in turn significantly reduced unsaturated hydraulic conductivity at supply potentials between ψ -0.15 and –0.55 kPa. Crusts continued to develop over the cropping season, not just after the first rainfall or first irrigation event. Crust formation resulted in the almost complete loss of all macropores larger than 545 µm from the surface soil. Crust formation and bed consolidation reduced infiltration, which prevented adequate wetting of the crop root zone and increased runoff from 0.5 % of irrigation to 35 % of irrigation. Importantly this study also demonstrated that van Genuchten soil water parameters could be derived for soil crusts through inverse simulation of cumulative infiltration data from tension infiltrometers.

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# The effect of Infiltrax™ on soil structure, aggregate stability, pore number and connectivity

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Soil structural stability in agricultural soils is of a high importance for maximising crop productivity, enhancing carbon sequestration and reducing nitrous oxide emissions. This study investigates the effect of a soil ameliorant (Infiltrax™) on soil structure, particularly on aggregate stability and porosity. Infiltrax™ is a combination of electrolytes, wetters, permeants and specific functional groups to aid in the aggregation and stabilisation of soil aggregates. Infiltrax™ is a non-toxic biodegradable liquid product. Urrbrae soil (Red Chromosol) was packed in columns and treated with three wetting and drying cycles using Infiltrax™ as a treatment solution. Saturated hydraulic conductivity ($HC_s$) measurements were taken at the end of the experiment and compared with an untreated control column. Significant increase in $HC_s$ was observed in the Infiltrax™ treated soil. Emerson dispersion test, performed on the air tried treated soil aggregates, showed the significant improvement in aggregate stability comparing with the control. X-ray computed tomography (CT) scanning of the replicate columns provided a means of measuring changes of soil structure in two (2D) and three (3D) dimensions. 2D and 3D images reconstructed from CT scans allowed visualisation of the structural effects of Infiltrax™ and quantification of increased porosity and connectivity due to Infiltrax™ treatment. CT scan results confirmed that the soils treated with the Infiltrax™ solution had much higher porosity than that of the soil leached with water only. Further studies are in progress to evaluate Infiltrax™ for the major farmed soil types of Australia.

# Monday 3 December

## SOILS AND INFRASTRUCTURE DEVELOPMENTS

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# Maximising the rehabilitation success through soil restoration strategies under severe uncertainty of future rainfall regimes

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Properties of the soil profile fundamentally control water-limited ecosystems. Soil water availability affects ecosystem processes but is also affected by them. Understanding this eco-hydrological interplay and the role of physical soil restoration is crucial for successful rehabilitation and management of post-mining landscapes in semi-arid areas. However, uncertainties exist in both the mathematical description of plant communities within ecological (sub)models and the predictability of future rainfall events. This study examines how both uncertainties impact decision-making in the context of ecosystem rehabilitation. A range of management strategies are investigated with regard to soil thickness as a representative soil attribute to identify the optimal soil restoration strategy for successful rehabilitation of native plant communities in Central Queensland. The results emphasise that the optimal soil thickness can counterbalance uncertainty in mathematical description of plant communities by increasing the simulated survival probability of plant communities. However, the effect of soil thickness on the robustness of plant communities against changes in future rainfall amount is rather minor. By allocating an economic value for restoring a particular soil thickness, these results can be used to evaluate the costs of ecosystem re-establishment with regard to two target measures – the survival probability of the plant community and their robustness against climate variability.

# Land and soil capability on rehabilitated mined land – prediction v's reality

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There is an increased social and political focus on the preservation of prime agricultural land in Australia and the importance of minimising the long term impacts on this land from mining activities. Consequently, there is also higher scrutiny on the predictions of achievable post mining land and soil capability outcomes made at the EIS stage. These predictions assume the implementation of best practice rehabilitation techniques, however the mining industry lacks a robust database of examples that can showcase actual results of land and soil capability achievements on post mining landforms. Therefore, the government and stakeholder assessment of long term impacts on agricultural land by mining projects is currently based on anecdotal evidence of past mine rehabilitation activities.

Given the potential for discrepancies between the predicted post mining land and soil capability versus the reality achieved over the long term, there is a rising need for well documented evidence of rehabilitation techniques that have achieved specific land and soil capability results, as assessed by certified professional soil scientists against agreed criteria. This documented evidence will increase the confidence level of the mining industry, government and stakeholders that the predicted results will be achieved.

The outcomes from this investigation, documentation and extension to the mining industry, can expand the focus of mine rehabilitation from an erosion sediment control and land use focus to include a targeted land and soil capability outcome, which can sustain various land uses, therefore raising the bar of what is considered 'best practice' mine rehabilitation in Australia.

# Degradation behaviour of sulfamethoxazole antibiotic in New Zealand agricultural soils under different environmental conditions

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### Abstract

A laboratory degradation study was carried out for sulfamethoxazole (SMO) antibiotic in three contrasting soils with varying organic carbon (OC) and microbial biomass. Experiments were carried out under varying environmental conditions following a destructive sampling technique. The activity of the microbial community was monitored during the incubation period by measuring the dehydrogenase activity. A simple first-order exponential decay (SFO) model was used to fit the measured degradation data for SMO and the corresponding degradation parameters and endpoints ($DT_{50}$ and $DT_{90}$) values were estimated. Dissipation times ($DT_{50}$) for SMO in Hamilton, Te Kowhai and Horotiu soils under non-sterile conditions were 9.2, 4.3 and 13.3 days respectively. The results showed good correlations between microbial activity and degradation rates. This study highlights the ability of farm soils to degrade sulfonamide antibiotics under different environmental conditions. Based on this study, it is concluded that it is unlikely that SMO will persist or accumulate in these soils, and its degradation in soils was a combined effect of biotic and abiotic processes.

### Key Words

Antibiotics, degradation, sulfonamides, dehydrogenase, microbial biomass, dissipation.

### Introduction

New Zealand has a rapidly expanding dairy industry, and well established beef, sheep, pig and poultry production, with its livestock population excreting about 40 times more waste than the human population (Sarmah *et al*. 2006). According to the New Zealand Food Safety Authority (NZFSA) the sulfonamide group of antibiotic contributes ~10 % of the total antibiotic usage of ~55 tons per annum. They are of considerable benefit to New Zealand livestock for prevention and treatment of diseases; however, they do have negative environmental and health impacts. Given the amount of antibiotics usage in New Zealand, direct excretal inputs from grazing animals along with permitted activity such as direct land application of animal waste is a cause for concern for regulatory bodies. Potential exists for the occurrence of sulfonamide residues in New Zealand's terrestrial and aquatic eco-systems. There are no published studies available in New Zealand regarding the occurrence and fate of sulfonamide antibiotic residues in soils and overseas studies show that degradation rates vary widely depending on the properties of the soil such as its moisture content, organic carbon, pH and soil bioactivity; temperature and physico-chemical properties of the compound.

Sulfamethoxazole which belongs to this group of antibiotic was chosen for this study as limited information exists on its fate in soils. Degradation half-life is an important input parameter required in antibiotic fate modeling exercises and for risk assessment purposes. While degradation half-lives for SMO are available in the literature (Holtge and Kreuzig 2007), given the varied soil and climatic conditions of New Zealand, extrapolation of degradation data obtained from overseas studies to New Zealand conditions are not desirable. Currently, there is no information on the half-lives or $DT_{50}$ values for SMO antibiotic under varied climatic and soil conditions in New Zealand.

The main objective of this study was to conduct laboratory incubation studies in order to investigate the degradation kinetics of sulfamethoxazole antibiotic in three different pastural soils (Te Kowhai, Hamilton Clay and Horotiu). The incubation conditions were maintained at 60% maximum water holding capacity (MWHC) and with two different initial antibiotic concentrations, different soil depths and at typical summer and winter temperatures (7.5°C & 25°C) in regions from where samples were collected and with sterilization at 60% MWHC. The study aimed to estimate the degradation parameters ($k_1$ and $M_o$) and endpoints ($DT_{50}$, $DT_{90}$ and $DT_{99}$) by fitting the observed SMO concentrations to simple first order kinetic model and to monitor and correlate microbial activity in soils by measuring dehydrogenase activity.

## Methods

*Soil*

Six topsoils (0-5 cm) and three subsoils (30-40 cm) supporting dairy pasture production were obtained from different regions of New Zealand. The soils were freshly collected, air dried at room temperature and sieved to < 2 mm. The soils varied in their pH, organic matter content and particle size distribution as shown in Table 1.

**Table 1: Selected properties of soils used in the study**

| Soils | pH 1:2 $H_2O$ | OC (%) | CEC ($cmol_c$ $kg^{-1}$) | Sand % | Silt % | Clay % | % MC | MWHC 60% | MBC (µgC $g^{-1}$) |
|---|---|---|---|---|---|---|---|---|---|
| Horotiu silt loam TS | 5.7 | 8.2 | 28.2 | 34 | 48 | 17 | 49.0 | 121.2 | 816 |
| Horotiu silt loam SS | 6.6 | 1.7 | -- | 34 | 48 | 17 | 60.9 | 134.8 | 584 |
| Te Kowhai silt loam TS | 6.7 | 5 | 21.7 | 9 | 54 | 37 | 23.9 | 79.6 | 1126 |
| Te Kowhai silt loam SS | 5.7 | 0.5 | --- | 12.3 | 62.8 | 24.9 | 41.4 | 84.5 | 536 |
| Hamilton clay TS | 5.7 | 4 | 17.2 | 13.7 | 51 | 30.4 | 23.7 | 77.6 | 1724 |
| Hamilton clay SS | 5.1 | 0.8 | -- | 13.4 | 40.3 | 46.2 | 23.8 | 75.5 | -- |

*Experimental protocol for degradation studies*

The soils from different depths (0-10; 30-40 cm) were freshly collected, sieved (2 mm) to remove plant roots and gravel and were stored at 5°C until use. The microbial biomass carbon (MBC) of the soils was measured by the fumigation extraction method (Wu *et al.* 1990). The moisture content (MC) of soils was determined gravimetrically at 105 °C and the water content adjusted to 60% of MWHC at -10 kpa, and soil was incubated at 25°C and 7.5°C for 2 days before spiking. Sterile controls were included, with sterilization being achieved by autoclaving twice (121°C, 103 kPa for 30 minutes). Duplicate soil samples (5 g) for a set time interval were placed in 35 mL Kimax centrifuge tubes, and appropriate amounts of SMO stock solution (1000 mg $L^{-1}$) prepared in methanol was spiked onto the soil to obtain an initial concentration of 5 or 0.5 mg $kg^{-1}$. After allowing the solvent (methanol) to evaporate, the contents were thoroughly mixed by vortexing and left to incubate. The moisture content in each vial was maintained at 60% of field capacity by adding water once every 3 days during the span of the experiment. The tubes were also aerated every day to ensure a constant oxygen atmosphere. The entire experiment was conducted in closed incubators with temperature control, in order to avoid photo-degradation. Experiments were also conducted with sterile soils so that role of micro-organisms on degradation of the antibiotic is established. Additionally, the activity of the microbial community was monitored during the incubation period by measuring the dehydrogenase activity of the soils at random sampling times during the length of the experiment.

*Extraction and analysis*

Degradation kinetics was measured by employing a destructive sampling technique. At selected time intervals, duplicate samples (5 g) were extracted with 10 mL Dichloromethane (DCM). The samples were vortexed for 1minute, followed by 15 minutes in an ultrasonic bath and shaken for 12 h in a rotary drum shaker. The tubes were then centrifuged at 1750 x g for 5 minutes and the DCM extract (~1.5 mL) was evaporated to dryness under a gentle stream of nitrogen, reconstituted in methanol (0.5 mL), and immediately analysed using HPLC-UV/Fluorescence detection. An elution scheme similar to that of Srinivasan *et al.* (2012) was developed. Separation was achieved using a Luna 5µ RP-$C_{18}$ column, eluted isocratically with mobile phase consisting of acetonitrile: 0.05 % trifluoroacetic acid: tetrahydrofuran in ratio of 40:55:5. The flow rate was 1.0 mL $min^{-1}$ and an injection volume of 20 µL. The limits of detection were calculated on a signal to noise ratio of 3 to 1. LOD for SMO was 0.02 µg $mL^{-1}$ using the UV detector, and was improved to 0.005 µg $mL^{-1}$ by using a fluorescence detector (excitation and emission wavelength were 272 and 340 nm respectively) in tandem with UV detection.

*Dehydrogenase activity (DHA)*

Dehydrogenase is a cellular enzyme involved in electron transport in the respiratory chain of fungi and bacteria. As dehydrogenase does not exist as an extra- cellular enzyme, any dehydrogenase activity can be taken as a measure of electron transfer activity in living cells. Determination of DHA is the most common method used to determine soil microbial activity. The method is based on measuring the DHA using a dye as an electron acceptor. Detailed experimental procedure can be found elsewhere (Friedel *et al.* 1994).

**Results and discussions**

The degradation rate of sulfamethoxazole in three New Zealand pasture soils varied with initial concentration, depth, incubation temperature and sterilization (Table 2). Increased initial spiked concentration resulted in decreased degradation rate constants and consequently increased $DT_{50}$ (Figure 1). Degradation rate was faster in topsoils when compared to subsoil samples in two of the soils examined. This clearly indicates the prominence of soil microbes in topsoil which strongly influenced antibiotic degradation (Figure 1). DHA activities for the top soils were a unit higher than those obtained for subsoils suggesting limited microbial activity at increasing depths (plot not shown).

SMO antibiotic did not tend to persist more than 90 days in any of the soils, at either depth, suggesting that natural bio-degradation is sufficient for the removal of these contaminants from the soil. The degradation rate constants were higher in soils incubated at 25°C than in soils incubated at 7.5°C (Figure 1). DHA activity measurement indicated very little or no activity for soils incubated at 7.5°C when compared to soils incubated at 25°C, thus emphasizing the role of micro-organisms in the degradation of SMO antibiotic and that a temperature of 25°C is preferred for enriched soil bioactivity rather than 7.5°C. Sterile soils irrespective of soil depth showed a marked decline in the degradation rate constant compared with non-sterilized soils (Figure 1), indicating role of abiotic processes in the dissipation of the sulfonamide antibiotic. Due to the hydrophobic nature of the compound there could be high sorption affinity of SMO to the soil due to its organic matter content. This could result in irreversible binding in topsoil and subsoil and lesser SMO bioavailability for dissipation in soils.

**Table 2: First order rate constants (day-1) and degradation endpoints (days) for SMO antibiotic in three different soils under varying conditions.**

| Sulfamethoxazole- First order kinetics | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | 25°C; 5 mg $kg^{-1}$ | | 25°C; 0.5 mg $kg^{-1}$ | | 7.5°C; 0.5 mg $kg^{-1}$ | | Sterile; 25°C; 0.5 mg $kg^{-1}$ | |
| Soils | | TS | SS | TS | SS | TS | SS | TS | SS |
| Hamilton | $DT_{50}$ | 11.36 | 12.38 | 9.24 | 11.75 | 25.39 | 29.88 | 11.00 | 34.66 |
| | $DT_{90}$ | 37.75 | 41.12 | 30.70 | 39.03 | 84.34 | 99.25 | 36.55 | 115.13 |
| | $DT_{99}$ | 75.49 | 82.24 | 61.40 | 78.05 | 168.69 | 198.50 | 73.10 | 230.26 |
| | $k_1$ | 0.061 | 0.056 | 0.075 | 0.059 | 0.0273 | 0.0232 | 0.063 | 0.020 |
| | $M_0$ | 3.10 | 4.62 | 0.283 | 0.385 | 0.356 | 0.418 | 0.395 | 0.347 |
| | $R^2$ | 0.99 | 0.94 | 0.84 | 0.97 | 0.97 | 0.91 | 0.93 | 0.61 |
| Te Kowhai | $DT_{50}$ | NA | | 4.31 | 14.15 | 20.69 | 20.95 | 13.00 | 22.43 |
| | $DT_{90}$ | | | 14.30 | 46.99 | 68.73 | 69.56 | 43.20 | 74.52 |
| | $DT_{99}$ | | | 28.60 | 93.98 | 137.47 | 139.13 | 86.40 | 149.03 |
| | $k_1$ | | | 0.161 | 0.049 | 0.0335 | 0.0331 | 0.0533 | 0.0309 |
| | $M_0$ | | | 0.318 | 0.401 | 0.310 | 0.396 | 0.315 | 0.394 |
| | $R^2$ | | | 0.966 | 0.958 | 0.94 | 1.00 | 0.93 | 0.90 |
| Horotiu | $DT_{50}$ | NA | | 13.33 | 12.38 | 23.18 | 19.69 | 18.10 | 22.65 |
| | $DT_{90}$ | | | 44.28 | 41.12 | 77.01 | 65.41 | 60.12 | 75.25 |
| | $DT_{99}$ | | | 88.56 | 82.24 | 154.02 | 130.83 | 120.24 | 150.50 |
| | $k_1$ | | | 0.052 | 0.056 | 0.0299 | 0.0352 | 0.0383 | 0.0306 |
| | $M_0$ | | | 0.216 | 0.360 | 0.263 | 0.326 | 0.366 | 0.378 |
| | $R^2$ | | | 0.98 | 0.97 | 0.91 | 0.96 | 0.89 | 0.88 |

# $k_1$ is the first order rate constant; $M_0$ is the initial observed concentration; $DT_{50}$, $DT_{90}$, and $DT_{99}$ are the degradation endpoints for 50, 90 and 99% antibiotic dissipation; $R^2$ is the measure of the goodness of fit of the model. TS = topsoil; SS = subsoil. NA= not applicable

## Conclusions

In general SMO dissipation in soils was due to the combined effect of biotic and abiotic processes as result of which it is unlikely that the antibiotic will persist or accumulate in these soils for a longtime. In the laboratory the experimental conditions can be easily varied one at a time, however under field conditions the degradation rates could be much faster as they could be affected by multitude of several factors such as moisture content, temperature, humidity, rainfall, sunlight and soil properties at the same time. The effect of anoxic conditions, varying moisture content and addition of manure along with antibiotic persistence in the presence of other contaminants was not investigated in this study.

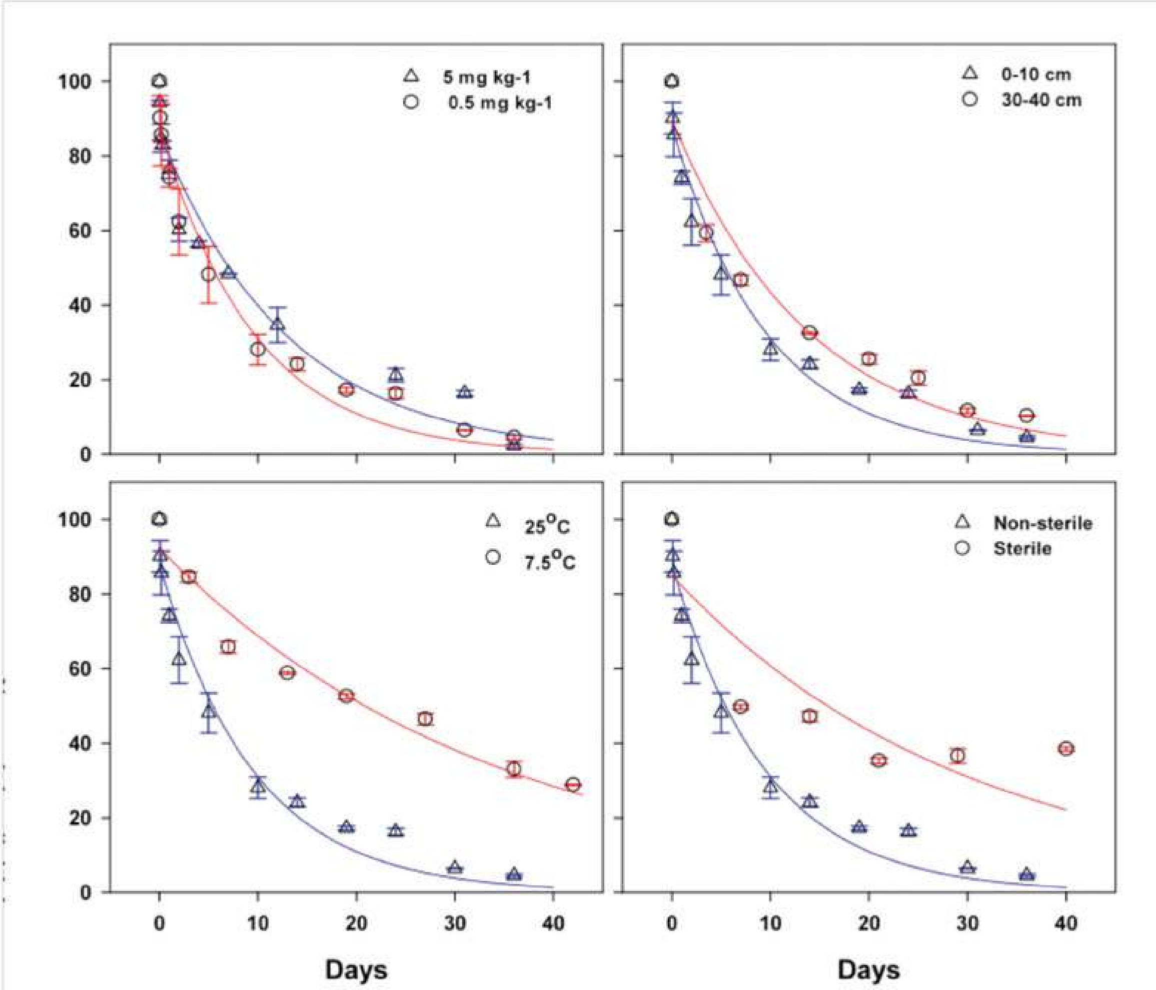


**Figure 1. Plots of measured residues of sulfamethoxazole antibiotic (SMO) in Hamilton clay soil as a function of time along with the fit for the simple first order kinetic decay (SFO) model. Vertical error bars represent the range of duplicate results for measured values.**

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# Soils of the Denniston–Stockton Plateaus, Buller, West Coast, New Zealand: Their role in minesite rehabilitation

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The Denniston–Stockton Plateaus of the Buller Region of New Zealand's South Island West Coast are renowned for coal mining. The erosion pavements and the associated regolith of early Tertiary (Eocene) quartzose sandstones are host to ecosystems adapted to these coal measures. Opencast coal mining has been active for many years in the area, and expansions are planned by a number of coal mining companies. The soils salvaged prior to opencast coal mining have become an important resource for effective rehabilitation of indigenous ecosystems.

The unique subalpine climate (ave. c. 6000 mm pa) at 500–850 m asl, about 6 km inland from the Tasman sea, and heath-rush-tussock 'pakihi' or mänuka (*Leptospermum scoparium*) scrub vegetation are associated with Rocky or Gley Raw Soils and Humose Acid Gley Soils, with small pockets of Humose Iron Podzols. The Raw Soils also occur under mountain beech (*Nothofagus solandri* var. *cliffortioides*) / podocarp forest on steeply sided, rocky, deeply incised stream gullies that dissect the tilted plateaus. The soils with pakihi and scrub vegetation occur in a mosaic matrix with weathered quartzose sandstone surface rock slabs. The soils are acidic (pH 4–5), siliceous, perennially saturated, leached, and infertile, with generally high organic matter contents in the Ah horizons. The preferred method of recycling these soils, which was developed for Stockton Mine by Solid Energy New Zealand Ltd, for minesite rehabilitation is by vegetation direct transfer (VDT) involving the relocation of entire root plates with pakihi, scrub, and forest vegetation. Salvaging, stockpiling, and respreading soils separately from the vegetation for replanting are lower priorities because of damage to soil invertebrates, potential for erosion, inevitable dilution of topsoil by subsoils and regolith materials, loss of organic matter (removing the main sources of N and P), and relatively slow rates of vegetation growth.

# Biodiverse floristic assemblage development and concomitant soil carbon dynamics along a restored forest chronosequence following post-mining perturbation

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A unique aspect of strip-mining (e.g. bauxite, mineral sands etc.) is the large tracts of forested land that is affected thus leaving large disturbance footprint compared to deep-pit mining operations. Many of the major bauxite deposits in Australia – the world's largest bauxite producer – occur in the unique jarrah forests of south western Australia (a biodiversity hotspot). In the context of understanding early ecosystem development and carbon sequestration processes in a de novo setting following vegetation establishment from a seed-mix, we investigated the impact of high-level mineral P fertilisation on soil C dynamics through a 15-year age series of 'restored' jarrah forest. This age series formed a 'chronosequence' which acted as a space-for-time substitution in which soil dynamics could be analysed over time and compared to a native, un-disturbed, benchmark. The soil profile was analysed to a depth of 20 cm for five restored forest age-classes ranging from 2 to 15 years. The quantity and quality of soil carbon in both bulk soils and particle-size based discrete carbon pools showed positive trends towards convergence with native forest levels. Parameters with a fast turnover such as litter layer and labile carbon pools were most successfully returned to pre-mining levels. C:N nutrient ratio for soil depths lower than 2 cm were progressively different from the native forest probably due to increased dominance of N-fixing pioneers in the younger restored sites. Floristic diversity and soil microbial function (determined by multiple carbon substrate utilisation) showed a skewed development of the restored towards the native benchmark.

# Monday 3 December

## SOIL CARBON

*Presented Posters*

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# Soil nitrous oxide and carbon dioxide emissions during pasture renewal.

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Pasture renewal practices in New Zealand often include a single season forage crop grazed in situ in winter, established using tillage following herbicide spraying of the old pasture. Although greenhouse gas emissions may be enhanced by herbicide application, tillage and winter grazing management, there is a lack of quantification of $N_2O$ and $CO_2$ emissions during pasture renewal.

We established a replicated, factorial field experiment to quantify how spraying, tillage and grazing management (including the effects of treading and urine) affect greenhouse gas emissions during pasture renewal. The main crop treatments were rape, established using intensive tillage or no-tillage, and grass established with no-tillage in autumn 2012. Six weeks before the tillage treatments were applied the pasture (>15 years old) was sprayed with glyphosate. Fluxes of $N_2O$ and $CO_2$ were estimated from headspace chamber concentrations measured two or three times weekly. Other variables measured to understand the key drivers of the emissions include soil mineral N, soil moisture and temperature, soil pore size distribution and bulk density.

Herbicide application had a large effect on $N_2O$ emissions, approximately 2.2 kg N $ha^{-1}$ was lost compared to non-sprayed pasture ($p = 0.003$). Soil organic matter mineralisation, the lack of plant N uptake and high water contents due to lack of plant water uptake were factors likely to contribute to these high losses. Intensive tillage (0.7 kg N $ha^{-1}$) almost doubled $N_2O$ emissions during the two weeks following tillage *compared to no-tillage ($p = 0.009$). Further results from this on-going trial will be presented.*

# Nitrous oxide emission from plants – development and testing of experimental apparatus

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Field nitrous oxide ($N_2O$) measurements typically include the sum of $N_2O$ evolution from both plant and soil. Quantifying the amount of $N_2O$ coming from each of these sources may help identify mitigation options, particularly if plant species effects are significant.

A double chamber apparatus has been developed that allows us to physically separate an intact, actively growing plant, into above and below ground components. During development of the apparatus consideration was given to:

- Replacement of carbon dioxide consumed by the plant in the upper (shoot) chamber
- An effective, simple and reproducible gastight seal between the upper and the lower (soil + root) chambers
- Chamber volumes both large enough to allow for uninhibited plant growth yet small enough to obtain a measurable $N_2O$ sample within a relatively short time period (1 hour)
- The capability of using either hydroponics or soil as the plant growth medium

Successful chamber seal tests were carried out using both ammonia ($NH_3$) and $N_2O$ as test gases.

We conducted an experiment in a climate controlled cabinet over a 6 hour period using the grass species *Poa annua* and *Paspalum dilatatum* (5 replicates), to assess the reproducibility of our method. The top chambers were in place for 60 out of every 90 minutes for three gas samplings using intact plants; the plants were then cut to leave 1 cm of stubble and a further $N_2O$ sample was taken.

Our initial results have confirmed the suitability of this apparatus for measuring $N_2O$ emissions from plants in isolation from soil emissions. Further experiments are being conducted to quantify pathways of $N_2O$ loss.

# The effect of external variables on nitrous oxide emission from nitrification process in soils

**Rui Liu**[1], Helen Suter[1], Tony Weatherly[1], Helen Hayden[2], Deli Chen[1]

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Due to spatial-temporal variability experienced with N2O field measurements, process based agro-ecosystem models are useful to predict N2O emissions under different situations. However, there is limited information about the source of N2O and many models estimate the contribution of nitrification using a fixed ratio. To improve modeling capability and our understanding of N2O emissions, it is essential to know how much N2O comes from nitrification and what is the affecting factor of N2O emission from this process. Nitrification can be autotrophic and heterotrophic but little information is available about the relative contribution of each to N2O production from nitrification. Ammonia-oxidizing bacteria (AOB) are generally believed to be the most important contributors to nitrification in agricultural soils, but ammonia-oxidizing archaea (AOA) can also be present in large numbers, but its significance to nitrification has not been understood well. My PhD research is focused on determining the contribution of nitrification to N2O production under different environmental conditions, and to identify the importance of the soil microbial population in this process, in order to improve computer models capability to define the sets of conditions that identify where and when nitrification is a significant source of N2O. This paper will report results from a laboratory experiment using the 15N labeling technique and the nitrification inhibitor (DMPP) to determine the contribution of nitrification to N2O emissions in a single soil and using molecular analysis (rRNA) technique to analyze the microbial community in nitrification at 40%water-filled pore space (WFPS), and 15, 25 and 35oC.

# Mitigation of nitrogenous gas ($NH_3$ and $N_2O$) emissions from cattle manure using organic amendments

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Ammonia ($NH_3$) and nitrous oxide ($N_2O$) emissions are two major pathways of nitrogen loss from cattle manure, lowering the fertilizer value and causing many negative environmental impacts. Currently, dietary manipulations, chemical additives and in field manure management techniques are practiced to mitigate those emissions. But the use of additives such as urease inhibitors is not cost effective under commercial feedlots as the procedure requires repeated applications for a longer term. Instead, only few research works have been done on the use of alternative organic manure amendments, such as lignite and biochar on reducing these emissions, which are having additional soil improvement properties. The effectiveness and mechanisms involved with these organic amendments on emission reduction are not fully understood. Lignite has a potential to be used as an acidifying agent to shift the equilibrium between $NH_4^+$ and $NH_3$ to promote the $NH_4^+$ form and subsequent adsorption. On the other hand, biochar has been proven to be effective on reducing the $N_2O$ emissions. Thus, a combination of lignite and biochar would have a potential synergistic effect on reducing both $NH_3$ and $N_2O$ emissions. To explore the above potential benefits, series of laboratory experiments are conducted to investigate nitrogen transformations of cattle manure treated with lignite and biochar, to understand the effects of different sources of amendments, application rates, mixing ratios, particle sizes, application timing and frequencies under simulated field conditions with varying temperatures and moisture levels along with the transformations of retained nitrogen in amended manure under field applications as a fertilizer.

# Use of nitrification inhibitors to reduce emissions of nitrous oxide from soil

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Nitrous oxide ($N_2O$) emissions from agriculture represent around 75% of the national total, and around three quarters of this comes from fertilised soils. Reducing $N_2O$ emissions from the application of nitrogen (N) fertilisers is one mitigation method that can potentially be achieved through the use of nitrification inhibitors such as DCD and DMPP. This paper reports a field experiment conducted on a Sodosol in southern Victoria growing a ryegrass seed crop. Urea (40 kg N/ha) was applied to a small plot study six times over an 8 month period (autumn to summer) alone or with amendment of the nitrification inhibitors DMPP (all times) or DCD (spring to summer only). Data on $N_2O$ emissions (manual chambers), soil mineral N transformations, N fertiliser fate (using $^{15}N$) and biomass production were measured regularly over that time period. Results showead that greatest emission of $N_2O$ occurred during spring after a wet winter, when the soil started to dry and temperatures increased. DMPP and DCD reduced $N_2O$ emissions equally well over the spring-summer period, with a reduction of 33 to 37% compared to urea. The inhibitors retained N as ammonium ($NH_4^+$) for longer time periods than urea particularly during wetter times. Biomass production benefits from use of the inhibitors were seen during wet periods and when fertiliser application was not made for 2 months, indicating that the DMPP could retain N in a plant available form for longer.

# Tuesday 4 December

## INVITED SPEAKER

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# Impacts of elevated carbon dioxide on soil physical properties

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The carbon dioxide concentration in the atmosphere is increasing. In the 53-year period between 1958, when the carbon dioxide concentration in the atmosphere was first monitored, and 2011, it increased from 316 ppm to 390 ppm. In this talk, I shall discuss the effects that elevated carbon dioxide has on the four soil physical properties that affect plant growth: aeration, water, mechanical impedance (compaction), and temperature. Because it is generally felt that diffusion is the main way that gases move in the soil, the amount of carbon dioxide in the air above the soil usually should be about the same as the carbon dioxide in the surface of the soil. Thus, since carbon dioxide in the atmosphere is increasing, it also is increasing in the soil. Therefore, I shall distinguish the effects of elevated carbon dioxide in the soil from elevated carbon dioxide in the atmosphere.

Roots need at least 10% by volume air space in the soil to survive, which means that the minimum oxygen concentration in the soil needs to be about 2% because air contains 21% oxygen. No such limiting (maximum) concentration has been established for carbon dioxide. Because oxygen is more important for root growth than carbon dioxide, most studies concerning the effects of soil water content and soil compaction on root growth have focused on oxygen in the soil rather than carbon dioxide. The absence of extreme toxicity at high levels of carbon dioxide (24%) in one study done with a sandy loam soil indicated that carbon dioxide would rarely, if ever, limit root growth in this soil. Excessive moisture, or lack of moisture, and mechanical impedance are more important in decreasing root growth than elevated levels of carbon dioxide in the soil. Although temperature has a direct effect on the growth rate of plant roots, apparently few or no studies have been done in which the interaction of temperature and elevated carbon dioxide in the soil has been studied. Research has concentrated on oxygen and temperature.

Studies done by my students and me showed that, during two growing seasons, soil in a grassland of mid-continental North America under doubled atmospheric carbon dioxide was wetter than soil under ambient carbon dioxide, and the difference in moisture was greater under dry conditions than under wet conditions. Less water was lost from the soil under elevated carbon dioxide than from the soil under ambient carbon dioxide, because elevated carbon dioxide closed the stomata and reduced the transpiration rate of the plants. Other studies confirmed our findings. When soil water has been measured, I have found no papers that report instances of soil water being less under elevated atmospheric carbon dioxide compared to ambient carbon dioxide. Few studies show how elevated atmospheric carbon dioxide affects plants grown in compacted soil, and, apparently, no information exists concerning the effects of elevated atmospheric carbon dioxide on soil temperature.

# Tuesday 4 December

## TEMPLE-SMITH LECTURE

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# Tasmania's soils – 'expect more...'

Bill Chilvers

This catchphrase is usually associated with the latest marketing campaign by your local rural merchant or an ad for the national health system.....but today I'm attaching it to our soils.

The gap is widening between world population growth on one hand, and gains in productivity farmers are achieving on the other. We are expecting great things of our soils. Can they deliver?

Tasmania has some major and exciting new irrigation schemes coming on line, the largest of which is in the region of my farm, 'Oakdene'. Productivity of Oakdene has lifted significantly in the previous decade, despite some major inherent soil limitations. Much has been observed and learnt in watching the soil respond to the changes in my farming lifetime from extensive pastoral grazing to dryland cropping, then irrigation, and now dairy conversion.

The path of development on Oakdene provides a valuable insight for how this new water might be utilised in the Tasmanian midlands, and how farmers might respond to growing demands, particularly for milk.

Can we 'expect more...' of the fragile soils and landscapes within the Midlands scheme? I intend to share some observations and insights on how the soils I farm are coping with the changes.

I believe range of social, economic and environmental factors may lead to a slightly different nature of dairy conversion in this region of Tasmania. I suggest moderate size conversions (400-600cow) on a truly mixed milking platform of grazing and cropping offers a sustainable outcome for the region. This farm model also gives a positive future for the regions fragile soils with the intensive dairy phase of the rotation building organic matter, while the cropping/dryland grazing phase is a time for lower water application and deep profile drying.

Monitoring soil, nutrient and water resources on Oakdene are discussed as well as strategies on how I might react to the trends, and finally what further research is needed for successful dairy conversions in the Midlands of Tasmania.

# Tuesday 4 December

## SOIL CARBON AND CLIMATE CHANGE (SEQUESTRATION – ACCOUNTING)

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# Effects of agricultural practices on soil carbon sequestration in Australia: A meta-analysis

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## Abstract

Do Australian agricultural soils play a role in climate change process and mitigation by sequestering carbon? The net carbon exchange of agro-ecosystems depends on the balance between carbon gain (input of crop residues) and carbon loss (decomposition of soil organic matter). There has been an ongoing debate globally as to whether agricultural practices can increase soil carbon stock. Australian soils are much older when compared to many soils in the northern hemisphere. Nutrient deficiency, salt accumulation and undesirable climate conditions probably restrict the potential of soil carbon sequestration in Australia. However, a quantitative review on the effects of agricultural practices on soil carbon change in Australian cropping systems is lacking. In this meta-analysis, we investigated the effect of agricultural practices viz. zero or reduced tillage, residue incorporation, rotation history and nitrogen fertilizer application on soil carbon change in Australian cropping systems. The potential patterns of variation in the effects of agricultural practices were also assessed by including categorical variables in the meta-analysis models. These variables include soil depth, cropping duration, soil clay content, and geographic location. The objective of this study was to systematically synthesize and quantify the effects of agricultural practices on soil carbon change in Australia. The potential, if any, of agricultural management practices on the mitigation of greenhouse gas emissions from Australian cropping systems will be discussed.

# Soil carbon sequestration potential of revegetated scalded soils following waterponding

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## Abstract

This study examined the carbon sequestration potential of scalded soils in the semi-arid rangelands of the Central West Catchment of New South Wales. The scalded soils develop due to the presence of a silty crust that inhibits infiltration of rainfall and prevents establishment of vegetation. Construction of a network of waterponds allows water to be trapped, infiltrate and rapidly leach soluble salts from the soil profile. The water also permits restoration of the shrink/swell characteristic of the high clay content sub soil and formation of large surface cracks (several centimetres in width). Soon after these two functional changes have occurred seed becomes trapped in newly formed surface cracks and is able to germinate by taking advantage of stored water and the pool of available P that is found in the scalds. Whilst waterponding was primarily developed to restore vegetation on degraded unproductive landscapes the substantial revegetation that occurred led to the question about their carbon sequestration potential. The scalds initially have a very low soil organic carbon (SOC) stock (approx 18.7 t/ha$^{-1}$ in the upper 30cm); however, once vegetation becomes established within the ponds SOC is rapidly sequestered and within five years amounts of up to 25 t/ha$^{-1}$ SOC in the upper 30 cm are present. The results of this research indicate that waterponding of scalded soils is an effective method that not only restores degraded unproductive landscapes but also facilitates the rapid sequestration of SOC. The results of our research may eventually lead to waterponding achieving eligibility as a carbon abatement activity under the Australia Government Carbon Farming Initiative (CFI).

## Introduction

Extensive areas of bare soil are present across the semi-arid rangelands of the Central West Catchment of New South Wales. These areas, referred to as scalds, occur on alluvial soils of prior streams in the western river flood plains such as those found on the Lower Macquarie catchment. They also include the meander plains of Marra Creek where this study was undertaken. The landscape is generally flat or has a slight slope of up to 1%. Originally the soils in the region were red duplex soils; however, as a result of wind and water erosion, they have lost their sandy $A_1$ horizon exposing a shallow silty-loam $A_2$ horizon. The remaining $A_2$ horizon overlies a saline and sodic, clayey B-horizon (Ringrose-Voase and McClure 1994). The shallow $A_2$ horizon forms a crust that prevents water infiltrating into the subsoil and facilitates high rainfall run-off. In this situation the soil assumes vertisol like characteristics (Isbell 1997) but is, in effect, a degraded chromosol (Isbell 1997).

Denudation of this landscape has arisen as a consequence of overgrazing by domestic, feral and native animals (Ringrose-Voase *et al*. 1989) and is exacerbated during drought when vegetation struggles to survive against the effects grazing and of lack of moisture. Approximately 100 000 ha of soil has been scalded in the Marra Ck region (Thompson 2010).

Natural regeneration of vegetation is sparse at best and even withholding stock from the scalds for long periods of time does not promote revegetation. Over the years several management options, including tyne pitting and mouldboard ploughing have been attempted to restore vegetation to the scalded areas but have all failed. Revegetation of the scalded soils can only effectively be achieved through a method of waterponding which has demonstrably been effective in achieving substantial increases in vegetative growth and species diversity on the scalded areas (Thompson 2008). Revegetation is achieved by establishing a network of horseshoe shaped banks of approximately 0.4 ha each, on gently sloping land (<0.5%) across the landscape. Ponds can be completely enclosed on the flatter areas. The ponds are best established to hold a maximum water depth of 10 cm. They naturally become revegetated with native grasses, and in some cases saltbush (*Atriplex* spp.) is manually seeded into the wall of the pond. Waterponding is an economically feasible method that successfully results in sufficient revegetation for grazing to be resumed. Up to 30 000 ha of scalded soil has been rehabilitated through waterponding in the Marra Ck region (Thompson 2010).

This study investigated the carbon sequestration potential of scalded soils that have been revegetated following construction of waterponds.

**Methods**

We compared changes to soil properties at 12 waterpond sites and three adjacent scald sites located on privately owned grazing properties along a 60km transect adjacent to Marra Ck in the Central West catchment of NSW. Soil cores to 40cm depth were collected from each waterpond and scald site. Each soil core was divided to obtain samples from four depth increments; 0-5cm, 5-10cm, 10-20cm and 20-30 cm.

The twelve waterpond sites were established between 1 and 25 years earlier. The waterpond treatments were divided into four age classes (age class 1 = 1 yrs, age class 2 = 5 yrs, age class 3 = 8-10 yrs and age class 4 = 20-25 yrs). At each pond site three replicate pond sites were sampled giving an overall total of 36 ponds. As there was a visible vegetation gradient within the waterponds, three soil cores were taken from close to the pond wall and bulked, three from the centre of the pond and bulked and three near the opening of the pond which were also bulked.

The three scalded areas were sampled using a 25 x 25m grid. Nine cores were taken from randomly selected cells within the grid and analysed separately for each depth increment.

Bulk density was measured on each sample using a subsample method (Murphy *et al*. 2003), as was pH/EC (1:5 water extract method 4A1 (Rayment and Lyons, 2011)), total organic carbon [Heanes method 6B1 (Rayment and Lyons, 2011)], total carbon and nitrogen [LECO method 6B2 (Rayment and Lyons, 2011)] and available P [Olsen method 9C2b (Rayment and Lyons, 2011)].

**Results**

The scalds were found to have a high EC of 1.84 dS/m at the surface increasing to >3 dS/m at depth (Fig 1). Soluble salts were found to rapidly leach from the soil profile with an EC of 0.27 dS/m at the surface of one year old waterponds.

Scalds also had a high available P concentration (> 19ppm) at the surface (Fig 2). The P is rapidly incorporated by the newly established vegetation with surface concentrations declining to <10ppm at the surface after one year.

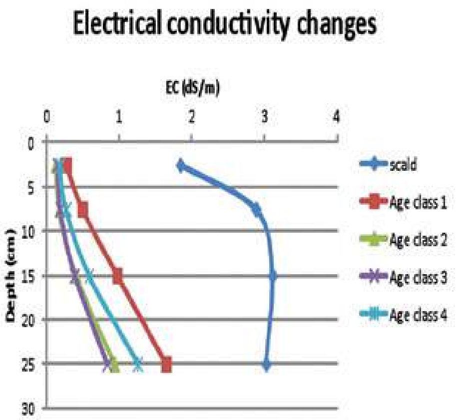


**Figure 1: Electrical conductivity of scalds and ponds (age class 1 – 1 yr, age class 2 – 5yrs, age class 3 – 8-10 yrs, age class 4 – 20-25 yrs)**

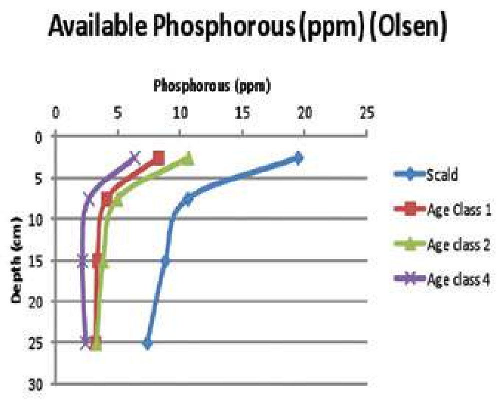


**Figure 2: Available P concentration in scalds and ponds (age class 1 – 1 yr, age class 2 – 5yrs, age class 3 – 8-10 yrs, age class 4 – 20-25 yrs)**

The results show that the carbon density within the scalds was 18.7 t/ha$^{-1}$ in the upper 30 cm (Fig 3). The waterpond results show that carbon density has increased up to 26.3 t/ha$^{-1}$ in ponds established for five years. However, there is a slight decrease in subsequent years with a constant carbon density of approximately 25 t/ha$^{-1}$. The results show higher levels of SOC in soil below 10cm depth in both the scalded and water pond sites (Fig 4).

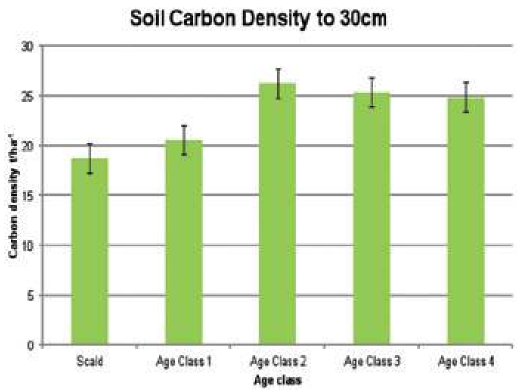


Figure 3: The soil carbon density in scalds and ponds to 30cm (error bars indicate standard error)

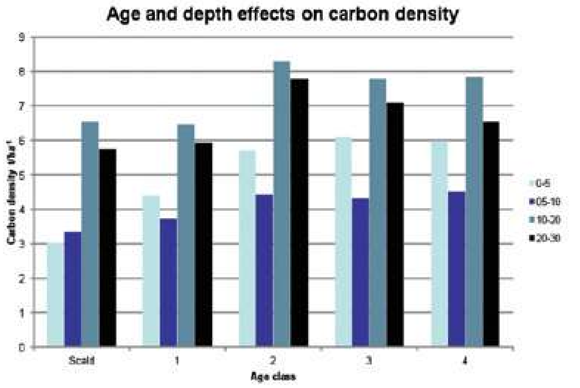


Figure 4: The distribution of SOC by depth increments showing a greater proportion below 10cm depth

Scalds had a mean C:N ratio of 5.8 at the surface increasing to 9.8 at the 20-30cm increment. Waterpond results indicate an increasing trend in C:N ratio with the mean surface results at 9.1 across the age classes. However, the 20-30cm increment indicated little change with a mean C:N ratio of 9.6 across the age classes. The scalds also have a higher bulk density (>1.5 Mg/m$^3$ at 0-5cm) compared with the waterpond sites (range 1.39- 1.25t/m$^3$ at 0-5cm varying with age class). Note that to take account of the change in soil volume that that occurs with waterponding the carbon density results for the waterponds (Figures 3 and 4) have been calculated using equivalent soil mass with the scald as the reference bulk density (Ellert and Bethany 1995).

**Discussion**

Retention of water in this landscape is critical before vegetation can become established on the scalded soils. Our research has found that within one year following waterpond construction (depending on the occurrence of rainfall) the soluble salts that have accumulated in scalds begin to be leached down through the soil profile.

The second critical change that is necessary before vegetation can be established is restoration of cracks in the soil. Initially, after the ponds receive a rainfall run-on event, shallow cracks develop. Over time, as the structure of the soil is modified, deeper vertical cracks establish (Ringrose-Voase and McClure 1994). These cracks permit infiltration of water deep into the soil profile where it is stored and protected from evaporation. Together, the leaching of the soluble salts and the restoration of the shrink/swell characteristic of these, clay rich soils causes large cracks (several centimetres in width) to appear in the surface of the scalds provide an environment suitable for germination of seed. The seed, which is naturally available in the landscape is moved across the scald surface as a result of rainfall induced water movement, but instead of being wash away, the seed is trapped within the cracks and waterponds.

Phosphorous (P) is an essential element for plant growth (Holford 1997). We have found that the scalds have a moderately high level (>19ppm) of available P in the top 5cm. This pool of P has originated as a result of rock weathering and has accumulated because no vegetation exists on the scalds to take up the P. Our results show that the available P levels are rapidly diminished due to plant uptake once the vegetation is established. There are a number of factors making it impractical and uneconomical to fertilise the soils in this region so the presence of this supply of P is a critical feature contributing to the successful revegetation of the waterponds.

Finally, once the vegetation becomes established in the waterponds, organic matter (OM) accumulates in the soil profile and SOC is sequestered. The results of our research show that the waterponds achieve approximately a 29% increase in carbon density to 30cm within five years of establishment when compared with the scalded soils. This is equivalent to an annual sequestration rate of > 1.0 t/ha$^{-1}$ year$^{-1}$; a relatively rapid rate when compared with the SOC sequestration rates found in European soils (Freibauer *et al.* 2004). Further, our results suggest that once the SOC is sequestered it is retained potentially permanently in the soil.

Typically, SOC is accrued and is stored in the surface layers of the soil profile where plant litter, roots and microbial activity is greatest. The concentration of SOC diminishes with increasing depth where such activities are low. Our results show that the SOC concentration is greatest at depths below 10cm.

Skjemstad *et al.* (1998) corroborate this finding suggesting that the shrink/swell characteristic of some soils can result in the redistribution of OM down through the deep cracks into the subsoil. Consequently, we conclude that the increase in SOC at depth is a result of redistribution of the new OM down through the soil profile as a result of the cracking and shrink/swell activity of the waterpond soils.

Skjemstad *et al.* (1998) also found that the radiocarbon age of the OM in an old sample of a vertisol profile they examined was around 2000 years BP, substantially older than other soil types they investigated. In view of their findings we conclude that SOC in the scalded soil is probably also old and stable because there is very little disturbance and hence the turnover potential of these soils is very low. This is because the surface crust prevents water infiltration and there is no vegetation on the scalds to contribute to the SOC pool.

Although the principal source of the accumulated SOC in the waterpond sites is as a result of the sudden increase in vegetation biomass we suggest that because the SOC is accumulating at depth it is potentially contributing to the stable or recalcitrant pool of SOC. This is due in part to the spatial location of the OM deep in the soil profile where it is less likely to be decomposed by microbial activity. Secondly, the physical interaction that occurs between OM and clay minerals leads to physical protection and stability of OM. Baldock and Skjemstad (2000) demonstrated a strong relationship between soil surface area (SSA) and mineralogy of samples associated with TOC. Therefore given the clay rich nature of the waterpond soils it is probable that as C is sequestered in the waterpond soil profile it will be stored in the recalcitrant pool.

## Conclusion

Waterponding is effective at restoring vegetation on scalded soils at a large scale. With careful management of grazing animal stocking rates it is possible that the rehabilitated soils can sequester and retain SOC indefinitely. With more than 30 000 ha of scald in the region having been rehabilitated by waterponding it is estimated that at least 189 000 t / C has been sequestered across the region; a substantial quantity of which will have been sequestered in the recalcitrant pool. This quantifiable contribution makes waterponding a project worthy of serious consideration as a carbon abatement activity under the CFI.

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# Soil organic carbon sequestration and turnover in leucaena-grass pastures of southern Queensland

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## Abstract

The sequestration of soil organic carbon (SOC) in agricultural systems has the potential to partially mitigate climate change by removing carbon dioxide ($CO_2$) from the atmosphere. Furthermore, incorporating a forage legume into grazing systems may aid C storage, as typically mature pasture soils are nitrogen (N) limited. In Queensland, *Leucaena leucocephala*, a leguminous shrub, is known to improve the N fertility of grazing lands. Despite this, there is limited research on the C and N dynamics beneath leucaena-grass pastures. This study assessed the potential of leucaena to sequester SOC by estimating the origin ($C_3$ or $C_4$), quantity (t/ha) and vertical distribution (m) of C stocks, in bulk fractions. A chronosequence of leucaena stands was sampled to a depth of 1 m and then analysed for C, N and stable C isotopes ($\delta^{13}C$). Pasture type affected C stocks within the upper 0.3 m (grass pasture < leucaena mid-row = leucaena row). On average, stocks were 14 % higher beneath leucaena rows compared to pure pasture-grass swards. Stand age also affected C stocks in the 0.3 m zone, with a 16 – 23 % increase in SOC over 40 years. However, this trend was not evident below 0.3 m suggesting other factors were affecting SOC storage at lower depths. Cumulative $C_3$-C stocks were also significantly higher beneath leucaena stands. However, irrespective of pasture type, the majority of C displayed a $C_4$ signature. This suggests that direct $C_3$-C contributions to SOC from leucaena were small (25%) compared to the effect of improving N fertility and hence $C_4$ grass productivity in N-limited, mature pasture soils.

## Introduction

The global focus on climate change has led to an increasing interest in soil carbon (C) and in particular, on the potential for agricultural systems to sequester atmospheric $CO_2$ as a means of mitigating greenhouse gas emissions. In Australia, grazing for beef cattle production is a major form of land use (> 56 %) and represents a significant opportunity for SOC storage, especially in more intensively managed systems (Catchpoole and Blair 1990). One system of interest is leucaena-grass pastures. *Leucaena leucocephala* is a perennial leguminous shrub, often incorporated in subtropical beef production systems as twin hedgerows separated by a 3 – 10 m grass strip. Current literature suggests that pastures containing legumes have a greater capacity to sequester SOC (e.g. Barrios, Buresh *et al.* 1996). Leucaena is reported to improve the N status of grazed pastures (Burle, Shelton *et al.* 2003) and sequester C when applied as a green manure to alley cropping systems (Isaac, Wood *et al.* 2003). However, limited literature exists on the SOC status of grazed *in-situ* leucaena-grass pastures (Radrizzani, Shelton *et al.* 2011). Currently, over 150 000 ha of leucaena is planted throughout central and south-eastern Queensland and this area is likely to reach 300 000 - 500 000 ha within the next decade (Shelton and Dalzell 2007). The objective of this study was to assess the potential of leucaena-grass pastures to sequester SOC by (i) estimating the origin ($C_3$ or $C_4$), quantity (t/ha) and vertical distribution (m) of SOC in the profile; and (ii) assessing the difference in SOC stocks (t/ha) between leucaena and pure grass pastures and with differences in leucaena stand age.

## Materials and Methods

Field sampling was conducted at Brian Pastures Research Station (25°39'S, 151°45'E) in southern Queensland on a chronosequence of leucaena plots (9, 22, 34 and 40 years) and paired pasture sites. A systematic grid consisting of 5 transects each with 5 sampling points (25 sampling points at each site) was overlain on the 20 m x 20 m sites. In the leucaena plots, samples were collected from both within and between the hedgerows (i.e. leucaena row and mid-row respectively). The soil cores were extracted from 1 m in depth using a hydraulic auger and then split into 5 depth intervals (0-0.1, 0.1-0.2, 0.2-0.3, 0.3-0.6, and 0.6-1.0 m). The soil type had been previously classified as a self-mulching, brown Vertosol (ASC).

Dried and ground soil samples (< 100 µm) were analysed for organic C, total N and stable $\delta^{13}C$ and $\delta^{15}N$ isotopes using an isotope ratio mass spectrometer (IRMS). Carbonates were removed with HCl prior to analysis. Bulk density (g/cm$^3$) was derived for each soil core to enable calculation of absolute SOC stocks (t/ha). Cumulative stocks were scaled to an equivalent soil mass (soil depth x bulk density) using a series of polynomial regressions to enable comparison between sites. The proportion of C derived from leucaena ($C_3$) was estimated with a two-compartment mixing model (Balesdent and Mariotti 1996).

Statistical analysis was conducted on SAS using the Proc Mixed and GLM procedures. Leucaena age and pasture type were the main treatments. Profile depth was a repeated measure. Treatment means were compared with the post-hoc Tukey HSD test. Response variables included organic C concentration (%), SOC stocks (t/ha), $\delta^{13}C$ ratio and $C_3$-C stocks (t/ha) for whole soil samples.

## Results

Soil organic C stocks were significantly higher beneath leucaena stands compared to grass pasture sites, within the upper 0.3 m of the soil profile (grass pasture < leucaena mid-row = leucaena row; $P < 0.01$) (Table 1). On average, stocks were 14 % higher in the 0-0.1 and 0.1-0.2 m depths under leucaena, however, this trend was not evident deeper in the profile (> 0.3 m). Cumulative SOC stocks increased significantly with depth reaching values between 167 and 172 t/ha by 1 m in the profile. Additionally, leucaena stand age was a significant determinant of SOC stores above 0.3 m (0 year < 9 year < 22 year = 34 year < 40 year; $P < 0.01$). The $\delta^{13}C$ signatures under leucaena pastures were enriched compared to mean values for leucaena foliage (-27.6 ‰). Age and leucaena vigour also affected isotope signatures. In the 9 year and 40 year sites, $\delta^{13}C$ values were relatively similar amongst pasture types. However, the 22 and 34 year sites displayed broad separation in signatures between leucaena row, mid-row and grass pasture sites at each depth in the profile.

**Table 1: Soil organic C, N and stable $^{13}C$ and $^{15}N$ signatures in leucaena row, mid-row and pasture sites, in grazed paddocks near Gayndah, southern Queensland. Letters represent significant differences between pasture types at a certain depth.**

| Depth | Pasture type | Organic C (%) | Organic N (%) | SOC stock (t/ha) | C:N | $\delta^{13}C$ | $C_3$-C (%) | $C_4$-C (%) |
|---|---|---|---|---|---|---|---|---|
| 0-0.1 m | Pasture | 2.28$^a$ | 0.15$^a$ | 28.0$^a$ | 15.1$^a$ | -14.6$^a$ | 8 | 92 |
| | Leucaena mid-row | 2.49$^a$ | 0.15$^a$ | 29.2$^a$ | 16.7$^b$ | -15.3$^{ab}$ | 13 | 87 |
| | Leucaena row | 2.86$^b$ | 0.18$^b$ | 35.2$^b$ | 16.0$^{ab}$ | -16.9$^b$ | 24 | 76 |
| 0.1-0.2 m | Pasture | 1.68$^a$ | 0.10$^a$ | 18.6$^a$ | 16.6$^a$ | -13.9$^a$ | 3 | 97 |
| | Leucaena mid-row | 1.89$^b$ | 0.11$^a$ | 20.3$^{ab}$ | 17.3$^{ab}$ | -14.4$^{ab}$ | 7 | 93 |
| | Leucaena row | 2.00$^b$ | 0.11$^a$ | 21.3$^b$ | 17.7$^b$ | -15.2$^b$ | 12 | 88 |
| 0.2-0.3 m | Pasture | 1.41$^a$ | 0.08$^a$ | 18.2$^a$ | 17.3$^a$ | -13.3$^a$ | 1 | 99 |
| | Leucaena mid-row | 1.63$^b$ | 0.08$^a$ | 19.6$^b$ | 18.8$^b$ | -14.2$^a$ | 5 | 95 |
| | Leucaena row | 1.64$^b$ | 0.07$^a$ | 20.5$^b$ | 23.7$^c$ | -14.4$^a$ | 7 | 93 |

The relative contribution of leucaena ($C_3$) and grass species ($C_4$) to SOC stocks differed according to pasture type, depth and stand age (Figure 1 and 2). As expected, $C_3$-C stocks were significantly higher beneath leucaena row and mid-row sites compared to grass pasture and this difference increased with depth ($P < 0.01$). Interestingly, however, the majority (> 76 %) of SOC displayed a $C_4$ signature, irrespective of vegetation type. In general, $C_3$-C stocks increased with stand age to reach a maximum under the 20 year site before declining to baseline pasture levels by 40 years. This trend was strongest under leucaena hedgerows and in the upper 0.3 m of the soil profile.

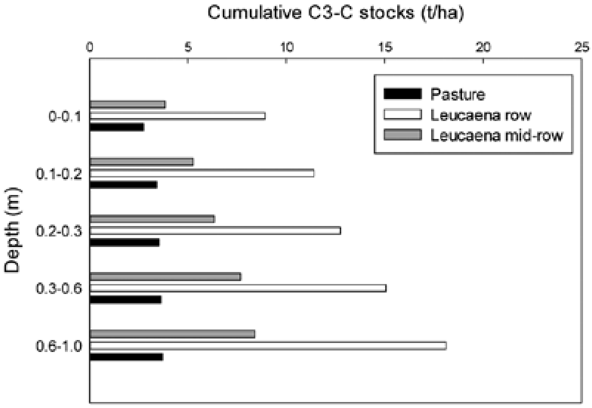


**Figure 1: Cumulative C3-C stocks beneath grass pasture, leucaena row and leucaena mid-row sites in grazed paddocks, near Gayndah, Queensland.**

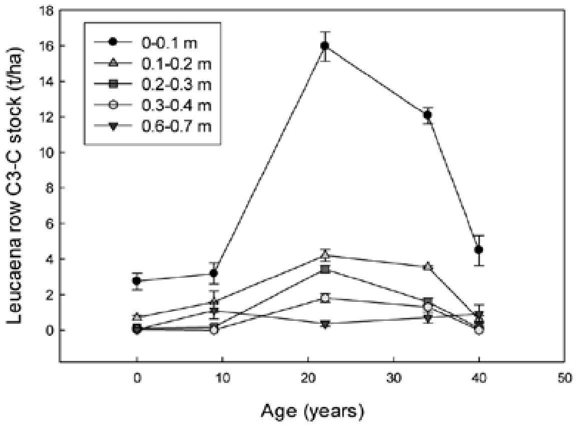


**Figure 2: C3-C stocks with depth in the soil profile for a chronosequence of leucaena row sites in grazed paddocks, near Gayndah, Queensland.**

## Discussion

Carbon sequestration is defined as the transfer and long-term storage of atmospheric $CO_2$ into terrestrial, geologic or oceanic C pools. One method for increasing SOC inputs involves the specific selection of pasture species based on biophysical criteria (i.e. $N_2$-fixing or deeply rooted). Incorporating a legume component into pasture (either herbaceous or tree form) has known benefits for the N status of the soil (Burle, Shelton *et al.* 2003). Likewise, several studies have demonstrated that legumes promote SOC sequestration across a range of soil types and climatic conditions (e.g. Diels, Vanlauwe *et al.* 2004; Isaac, Wood *et al.* 2003).

This study demonstrates that leucaena hedgerows positively affect the C and N balance of mature, grazed pastures. On average, leucaena hedgerows stored 35.2 t/ha of SOC in the upper 0.1 m of the profile, which corresponds to a 14 % increase over baseline pasture-grass levels. These findings are consistent with a previous study by Radrizzani, Shelton *et al.* (2011) who found leucaena stands stored 17 % more SOC than adjacent pure grass swards in the 0-15 cm depth. Interestingly, leucaena rows also recorded higher C levels (8 – 11 %) than the inter-row spaces. Along with the distinct differences in $\delta^{13}C$ signatures, this suggests that organic matter inputs (leaf fall and root turnover) were predominantly localised to the parent plant. While cattle dung can be distributed widely, direct leaf fall is the primary source of residue in systems subject to low intensity grazing. This is also the case in drought or frost prone areas.

Sequestering SOC deep in the profile is considered a potential benefit of seeding legume and pasture species with extensive root systems. However, this study indicates that leucaena stands primarily store SOC in the topsoil horizons. Below 0.3 m, pasture type was insignificant compared with inherent site variability. Hence, it appears that leucaena contributes the majority of biomass C through above ground processes and shallow root turnover. The age of leucaena stands also affected SOC stocks, however as previously noted, the most significant trend was observed in the upper 0.3 m of the profile. On average, there was an extra 12 t/ha of SOC sequestered over 40 years, corresponding to a rate of 0.3 t C/ha/year.

While N-fixing plants typically accumulate more SOC than equivalent tree or pasture species, there is still a question over the source ($C_3$ or $C_4$) and stability of C storage. In this study, the relative composition of SOC differed significantly between pasture types. Overall, leucaena rows sequestered 8.9 t / ha of $C_3$-C to a depth of 1 m compared to 3.8 t / ha and 2.7 t / ha for leucaena mid-rows and pasture respectively. This represents a 60 - 70 % increase in $C_3$-C from baseline pasture levels. However, $C_3$-C formed only a small proportion of total stocks. On average, the profile beneath leucaena rows stored 115 t / ha of $C_4$-C, correlating to 88 % of SOC by 1 m in depth. Hence, the direct contribution of SOC (i.e. $C_3$-C) by leucaena stands was relatively small despite a significantly higher total C stock in the upper 0.3 m of the profile. There are three potential explanations for the disproportionate input of $C_4$-C within leucaena pastures. Firstly, leucaena residues contain high levels of organic N (2.8 %), thus providing an effective source of fertilizer which may promote $C_4$ grass productivity. Typically, mature grass pasture soils are N limited, constraining their ability to sequester SOC (Radrizzani, Shelton *et al.* 2011). Significant amounts of N and phosphorus (P) are required to support plant growth, maximize net primary productivity and sustain a functioning microbial community (Radrizzani, Shelton *et al.* 2011). Hence, leucaena residue may act as a priming mechanism for soil microbes to mineralise organic N along with other essential plant nutrients. Secondly, the high litter quality of leucaena residues, including low C:N ratio and high decomposition rate, may potentially accelerate the removal of $C_3$-C from the soil profile. Lastly, differences in leucaena and pasture-grass productivity may be related to the inherent grazing tolerance of different species and diet selection by cattle.

In this study, $C_3$-C production was tightly coupled to leucaena age and stand productivity over time. The sequestration of $C_3$-C was highest in the 22 year stand and significantly lower beneath the 10 year and 40 year hedgerows. The reduction in $C_3$-C stocks can be attributed to leucaena rundown, which was evident as reduced stand density and plant productivity. The depletion of P and sulfur (S) and an associated decline in biological $N_2$ fixation was found to be the primary cause for leucaena rundown in a study by Radrizzani, Shelton *et al.* (2011). Hence, grazed leucaena-grass pastures in which product is removed from the system, will require additional nutrients to maintain and sequester SOC.

## Conclusion

Soil organic carbon sequestration in grazing agro-ecosystems has the potential to partially offset fossil fuel emissions. As a management practice, planting leucaena hedgerows may aid in sequestering additional SOC within the profile. However, this study suggests that C is only stored in the upper 0.3 m and is significantly higher beneath the rows compared to adjacent inter-row strips. Additionally, the majority of C is derived from $C_4$ grass pastures, most likely from the injection of high-N leucaena residues into the profile. The permanence of SOC stores need to be examined in systems in which are actively grazed. Long term product removal has led to the depletion of soil nutrients and subsequent leucaena rundown (Radrizzani, Shelton *et al.* 2011). This in turn has significantly reduced SOC stores beneath aged leucaena stands. While leucaena represents a profitable fodder species for beef cattle production and a source of N for mature grass pasture soils, SOC storage is tightly coupled with stand productivity.

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# Soil carbon science to support a scheme for the payment of changes in soil carbon – lessons and experiences from the CAMBI pilot scheme

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## Abstract

The application of soil carbon science to a pilot scheme to implement a market based instrument for soil carbon is described. The steps included: 1. Development of a set of potential soil carbon levels based on land management, 2. The determination of the initial soil carbon levels, 3. Estimating the expected rates of change in soil carbon, and 4. The initial and final measurements of soil carbon levels for the successful contracts. At each step it was necessary to apply the known soil carbon science to address specific issues.

## Key Words

Soil carbon, market based instrument, land management, measurement, rates of change

## Introduction

The potential for soil carbon sequestration to play a significant role in meeting Australia's greenhouse gas reduction targets has attracted widespread interest. Development of schemes to make payments to land managers for changes in soil carbon levels is frequently advocated as a possible way to engage large areas of agricultural land for carbon sequestration. While the amount of carbon dioxide that could be removed from the atmosphere in this way is finite and relatively small in overall terms, it is potentially one way to limit the increases in atmospheric carbon dioxide while improving soil condition. However, there is much uncertainty and perhaps confusion about the optimal mechanisms that would successfully engage land managers in such schemes. Any potential scheme is beset with uncertainties in the soil science, economic aspects of contract design, verification procedures and ultimately farmer acceptance. This paper reports on the methods and experiences of a pilot market based instrument (MBI) for soil carbon in a small region in Lachlan Catchment, Central West NSW, known as the CAMBI Project. It briefly describes the MBI but concentrates on the science aspects of soil carbon measurement and estimates of potential change that were required to make the scheme effective. It identifies four fundamental scientific operations that were required to implement the MBI within the target area within the Lachlan Catchment.

## Overview of the Catchment Action Market Based Instrument (CAMBI) Project

The CAMBI Project was funded by Catchment Action NSW towards the end of 2009 when significant uncertainty existed around the agriculture sector's role in climate change mitigation. The aim of the project was to fill a number of important gaps in current land management policy through the design and implementation of a MBI to enhance soil carbon sequestration. The pilot was implemented with the support of the NSW Department of Primary Industries, the NSW Office of Environment and Heritage and the Lachlan Catchment Management Authority from 2011 to 2012 (Lachlan CMA 2011). Farmers were funded through the MBI to adopt land management actions with the specific aim of increasing soil carbon. The pilot project concentrated on a small part of the catchment on a defined area with a specific climate and soil type. This was the Cowra Trough Red Soils and consisted of a belt of Red Chromosols used primarily for cropping in rotation with pasture phases and permanent perennial pastures. To participate in the MBI, land holders were required to submit a bid reflecting the price that they would need to be paid ($ / t of $CO_2$-e), to undertake land management changes that would sequester soil carbon. Three kinds of contracts were offered:

- An action-based contract where landholders were paid to change to standardised actions that would increase soil carbon stores. The three actions were: conservation tillage, permanent pasture and environmental plantings
- An outcome-based contract where landholders were paid on the basis of the amount of carbon actually sequestered over the five years of the pilot. Landholder-defined actions needed to result in an increase soil carbon through: increased biomass and/or increased quality of herbage mass input, decreased soil disturbance, increased retention of stubble or increased biomass returns to soil.

- A hybrid contract, where landholders are paid partly (50%) on the basis of adoption of standardised actions and partly (50%) on the amount of carbon actually sequestered over the five years of the contract.

To determine the competitiveness of the bids for each of these contract types, the expected rate of soil carbon sequestration was assessed. This served as the underlying metric for the MBI. In determining this expected rate, it was judged necessary to obtain an initial or starting level of soil carbon for each site. All the bids were then ranked on the basis of cost effectiveness expressed in $ per tonne of CO2-e (see Lachlan CMA 2011). Contracts were awarded up to the CO2-e price where the total project budget was expended.

**Steps in the Application of Soil Carbon Science**

The project required a synthesis of existing soil carbon data sets and strategic soil sampling to underpin the soil carbon science of the project. This led to the development of a soil carbon metric that was used to assess rates of soil carbon sequestration, a critical element in determining which land managers would receive payments under the MBI. The following four basic requirements were identified.

1. *The development of a set of potential levels of soil carbon and the likely changes in soil carbon that could be expected.*

To achieve this, the Lachlan Catchment was originally stratified on the basis of soils, climate and land management and a map of soil carbon zones or soil carbon management units produced. All new data collection for soil carbon was guided by this stratification and used standard sampling procedures (DECC 2009, Sanderman et al. 2011). The aim of the sampling program was to estimate the long term equilibrium level of soil carbon. Several features of this overall approach are described.

a. The stratification was undertaken using existing climate data and existing soil and geology maps. This allowed the identification of a series of likely soil carbon zones or soil carbon management units that were intended to have a relatively uniform soil type and climate and soil carbon potential.

b. On a regional basis these soil carbon zones or soil carbon management units gave some confidence in the capacity to make predictions about soil carbon levels under different land management practices.

c. Analysis of the soil carbon data to estimate the soil carbon levels for each land management practice in each soil carbon zone showed some useful general trends but it soon became apparent there was insufficient data to implement a catchment-wide MBI. Hence to implement the pilot scheme to set up a MBI for soil carbon sequestration, it was necessary to concentrate on a smaller area (the Cowra Trough) where there was sufficient information on soils, climate and the soil carbon levels under different land management practices.

d. Given that it takes a substantial time for the soil carbon levels to reach long term equilibrium (20 to 50 years), and based on the land management information recorded at each site, few, if any landholders tend to implement a single, simple land management regime on an individual paddock over a long period of time. Therefore any soil carbon measurements are likely to be only approximations of the dominant land x soil x climate combinations. A series of soil carbon modelling exercises were undertaken with FullCAM and with RothC to support the measured data.

e. A simple grouping of the land management data into the routine divisions such as direct drill, traditional till and pasture was too simple and did not necessarily reflect the soil carbon levels. The possible reasons for this are described.

    i. The model outputs suggesting that soil carbon levels followed the simple progression traditional tillage < reduced tillage <direct drill < zero tillage < pasture was not confirmed clearly in the data. While there was a general trend it was not confirmed with statistically significant levels of confidence. Modern crop management includes a range of options, each potentially contributing to soil carbon sequestration potential. The differences in soil carbon levels between pasture and cropping were also very dependent on the nature and management of the pasture. This is consistent with Liu et al. (2011) who showed that pasture management, especially the level of nutrients is critical to the level of soil carbon.

    ii. All new sampling was done with the standard 25m quadrat proposed in the standard protocols (DECC 2009, Sanderman et al. 2011). However, field observations of pasture composition and herbage mass suggested that in some cases the 25m quadrat is possibly too small and that some of the variability detected was a consequence of within paddock variation. Published information on variograms for soil carbon (eg McBratney and Pringle 1999) suggest that soil carbon commonly varies at a range of 100m rather than 25m.

*2. Determination of the Initial Levels of Soil Carbon for each Paddock in the Initial Bid Process*

It is a reasonable assumption that soils with carbon levels close to the optimum for a region or area are unlikely to gain much from changes in management. Thus in the pilot the determination of an initial level of soil carbon at a site or paddock became a critical step to identify and potentially exclude sites where little change in soil carbon is likely. The initial soil carbon level defines the potential for changes in soil carbon levels that are possible as a result of changes in land management. The target paddocks were those that were likely to gain the largest changes in soil carbon stores and were therefore those with soil carbon levels well below the regional or area optimum. Predicting soil carbon levels using land management histories alone is limited by the following:

a. It was clear from the land management questionnaires that few if any land managers maintain a simple, uniform land management system over a long period of time. The general practice is that the land management practices vary with the seasons and economic conditions.

b. The level of information available from land holders on the land management history varied greatly. In some cases it was difficult to predict the likely soil carbon levels because of missing information on specific tillage operations, stubble management, nutrient management and grazing management.

c. The time and expertise required to interview a land manager to obtain sufficient information to make a prediction about soil carbon levels may not be any more cost effective than undertaking some modified soil carbon measurements based on pedotransfer functions and ultimately be significantly less accurate.

For the CAMBI project the initial soil carbon levels was of such critical importance to the bidding process that baseline sampling was conducted to estimate these. This was a critical input to estimate the potential change in soil carbon and help calculate the cost per tonne of soil carbon and the payments to be made. To keep transaction costs to a minimum, soil carbon was not measured to 30 cm for all potential paddocks for the soil carbon MBI. Rather a pedotransfer function was developed using the soil carbon content for 0 – 10 cm to predict the soil carbon store to 30cm. The use of the pedotransfer function meant that it was possible to obtain an estimated value of the soil carbon store for all the bids, but only for the targeted soil carbon management unit, the Cowra Trough Red Soils. Once the pedotransfer function was applied outside the targeted area, the confidence in the predictions fell below an acceptable level.

*3. Estimating the expected rates of change in soil carbon following changes in land management practices.*

For the CAMBI Project, payments to farmers are based on either predicted or actual change in soil carbon after 5 years. Estimation of these expected changes in soil carbon are dependent on the following:

a. The initial starting level of the soil carbon store ($¢_{I)}$.

b. The expected final long term equilibrium level of the soil carbon store under the contracted land management system (¢E). This information was estimated from the stratification procedure described in Section 1.

It is the difference between the initial level and the final level which is important in predicting the potential for soil carbon sequestration. The next requirement is to normalise or calibrate this estimate to the local region to account for the local climate, soil types and climate. This can be achieved by estimating the following:

a. The maximum change in soil carbon level that can be expected from the land management system with the lowest soil carbon levels to the highest soil carbon levels in an agricultural land use ($¢_{max)}$. For example this may be the difference between the following:
   - Traditional tillage regime with stubble burning soon after harvest and numerous tillage operations prior to sowing and low nutrient regime, and,
   - A pasture that is well managed with moderate utilisation to maintain high levels of photosynthetically active plant material and a high level of nutrients.

   For the Cowra Trough Red Soils this was estimated at 35 t/ha/30 cm.

The maximum rate of change that can be expected based on the local climate and soils and so the potential biomass growth for a region (Smax). This was estimated at about 1000 kg/ha/30 cm/yr based on published information.

The predicted initial rate of change (first 5 years) of soil carbon stores (Ŝ) was given by the equation below:

$$Ŝ\,(t/ha/30cm/yr) = s_{max\,*}\,(¢E - ¢I) / ¢_{max}$$

The derivation and testing of this linear equation is discussed further in Rawson et al. (in prep.). While this is very much an empirical equation, it is useful for establishing differences between land management practices and soil carbon zones as the normalising factors smax and ¢max will vary between soil carbon zones. The rates estimated with the equation were comparable to modelled rates of soil carbon change using the Roth_C and FullCAM models for the area and with published data. At longer periods of time the relationship will become non-linear. It does not appear to be applicable for environmental plantings; hence ¢max is based solely on agricultural systems.

*4. The Measurement of Changes in Soil Carbon Stores at the Paddock Scale for Outcome-based Contracts*

Payments for Outcome –based contracts were based on the actual changes in soil carbon levels at the paddock scale which required actual detailed measurements of the soil carbon stores. The bid process, the predictions of rates of change in soil carbon levels and the 5 year term of the contracts indicated that the required accuracy for detecting changes in soil carbon stores between the start and the end of the contracts was of the order of 1.5 to 2.0 t/ha/30cm. Such accuracies were not possible using the pedotransfer function developed earlier. The estimation of soil carbon levels at the paddock scale to this level of accuracy was potentially restricted by limits on the transaction costs as the pilot project was being conducted as a realistic scheme to implement a MBI for soil carbon. Published information on sampling for soil carbon and a study of a paddock in the Lower Macquarie in Central West NSW (Singh et al. submitted), as well as our own field experience was used to develop a protocol for sampling to estimate soil carbon stores at the paddock scale (Murphy et al. in prep.) A review of this work indicated that it was possible to achieve the required level of accuracy within the limits of the transaction costs. Estimating soil carbon levels for a whole paddock or set of paddocks is a very different operation to the more generally accepted sampling methodology using a 25 m quadrat which is used to compare paired sites and to obtain regionally based estimates of soil carbon stores. At the paddock scale it is necessary to account for much greater spatial variability which requires a different sampling strategy.

## Conclusion

The CAMBI pilot scheme was a test case for the application of soil carbon science in a trading scenario. While in general the current state of soil carbon science was adequate to support the implementation of a MBI for soil carbon, the project has demonstrated that it was necessary to apply the available soil carbon science in a more rigorous and demanding set of circumstances imposed by the market based instrument that focussed on the land management outcomes at the paddock scale. The practical application required by CAMBI required a much sharper focus on the practical application of soil carbon science.

## Acknowledgements

Luke Beange and Mary Kovac of DPI, John Young of the Office Environment and Heritage, Lachlan Catchment Management Authority, especially David Trengove, Ian Packer and Allan McGufficke; and especially the land holders in the Cowra Trough Red Soils who have participated so constructively in this participatory research model.

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# A pragmatic model for estimation of pre-clearing soil organic carbon levels in eastern Australia

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### Abstract

An understanding of initial soil organic carbon (OC) levels prior to vegetation clearing is important for modelling processes of soil carbon sequestration. It is a key component of carbon accounting methods such as FullCAM as used in Australia. A pragmatic readily applied model for pre-clearing topsoil organic carbon levels in eastern Australia has been developed, using a dataset of 466 soil profiles from native vegetation sites in eastern Australia, together with covariates of annual maximum temperature, annual precipitation, parent material composition (silica%), aspect and topsoil depth. The derived model is of moderate statistical strength with the multiple regression relationship for OC on natural log scale achieving an $R^2$ of 0.53 and a residual standard error of 0.63 $\log_e$ OC%. Validation against 15% of the data points initially withheld revealed a RMSE of 2.3% OC and a median absolute difference of 1.7% OC. The actual versus predicted relationship revealed a Lin's concordance correlation coefficient of 0.75 and an $R^2$ of 0.61. Of the environmental covariates, it appears that temperature has the greatest influence on topsoil OC levels, followed by parent material composition, then rainfall, with aspect having the least influence. The model (i) allows relatively easy modelling and prediction of topsoil OC over native vegetation sites in eastern Australia, (ii) can inform on pre-native vegetation clearing soil OC levels for carbon accounting schemes used in Australia such as FullCAM, and (iii) inform on natural factors controlling soil OC distribution in eastern Australia.

### Key Words

Soil organic carbon, modelling, pre-clearing, native vegetation, FullCAM

### Introduction

Knowledge of the role of soils in storing carbon is vital for understanding, modelling and managing the impacts of climate change. Soil organic carbon (OC) levels in soils have changed significantly since the arrival of mankind in Australia, particularly since European settlement with the associated large scale conversion of native ecosystems to agricultural landscapes. An understanding and knowledge of SOC contents in soils prior to vegetation clearing is necessary for reliable modelling of SOC levels.

Various soil carbon models and accounting schemes, such as Roth C and FullCAM, used in Australia aim to model OC levels under different climate and land management regimes (Jenkinson 1991, Richards & Evans 2004). These all rely on assumptions of the initial base levels of OC, but these are made at quite broad, crude levels. For example, in FullCAM, they are derived from a spatial layer of crude estimates of soil carbon mass to 30 cm for identified soil types or soil-landscape units within IBRA biogeographic regions, supported by typically very sparse laboratory data (Webb 2002). There is no direct consideration of climate, parent material composition, topographical or aspect factors. There appears to be considerable potential to significantly improve pre-clearing soil OC estimates used by carbon accounting models such as FullCAM.

Apart from the pre-clearing map mentioned above, there appears to be few previous attempts to model and map pre-native vegetation clearing soil OC levels in Australia. Bui (2012) examined and mapped soil carbon density under eucalypt forests in Australia. Digital maps and models of soil OC under current land use conditions have been prepared for the agricultural zones of Australia (Bui *et al.* 2009, Henderson *et al.* 2005, Wynn *et al.* 2006) as well as for several regions and localised areas. However, these do not directly inform on pre-clearing soil OC levels, but rather current levels for existing agricultural landscapes. Furthermore, the models upon which they are based are complex and use sophisticated data sources and are therefore usually not amenable to modification to derive the required pre-clearing soil OC predictions.

This study attempts to (i) produce a pragmatic model for the prediction of soil OC levels in naturally vegetated sites over eastern Australia, (ii) compare the predictive ability of the model against the system used by FullCAM and (iii) derive insights into factors controlling soil OC distribution in undisturbed sites of eastern Australia.

## Data and Methods

The overall strategy was to prepare a multiple regression model with associated statistics using a dataset with soil organic carbon (OC) and readily available covariates for vegetated sites in eastern Australia. The model was then validated against data points initially withheld from the analysis.

### *The dataset*

A soil data set of profiles with OC laboratory results from eastern Australian native vegetated sites was compiled. From this only those profiles with adequate site data and that met other criteria as outlined below were selected. Final profile numbers were as follows: Queensland (134), New South Wales (212), Victoria (6) Tasmania (46), CSIRO (eastern Australia, 68) amounting to a total of 466 profiles. The dataset was created using MS Access with further organising and sorting using MS Excel.

### *Carbon values and co-variates*

Only those profiles with less than 18% OC were included in the final dataset as levels above this are defined as organic materials in the Australian Soil Classification (Isbell 2002). For approximately 95% of profiles, the Walkley-Black (W-B) wet oxidation method was used to derive the OC values. LECO and other combustion methods were used for the remainder. No correction factors were applied to account for possible under-estimations of OC values by the W-B method used in earlier decades (Skjemstad *et al.* 2000).

Only those sites under native vegetation were included in our analysis. However, as sample numbers were low, and significantly under represented in other jurisdictions relative to NSW, profiles from lightly disturbed native vegetation regimes such as native forestry were included for non-NSW jurisdictions. This resulted in an additional 116 samples being added, but despite this NSW samples still dominated. Covariates were selected to represent the key soil forming factors of climate, parent material, relief and aspect, together with topsoil horizon depth as outlined below.

*(i) Climate* – simple annual precipitation (mm pa, precip) and annual maximum temperature ($^oC$, $T_{max}$) values were used. These were derived from 2.5 km Australia wide climate grids obtained from the Australian Bureau of Meteorology and represent mean values obtained over the 1961-1990 period.

*(ii) Parent material* – this is derived from the parent material descriptor or geologic unit recorded at each site by the soil surveyor. The material is converted into an equivalent silica content using recognised relationships (Gray *et al.* 2012, 2009), for example, granite being highly siliceous (average 73% silica) and basalt mafic (average 48% silica). Silica content provides a meaningful quantitative estimation of the chemical composition of parent material and generally has a direct relationship to quartz content and an inverse relationship with basic cation content.

*(iii) Relief* – a new *topo-slope index (TSI)* that combines topographic position and slope gradient was developed to represent this factor, see Gray *et al.* (2012). However, this variable was found to have only weak statistical significance in the derived relationships.

*(iv) Aspect* – a new 1 to 10 *aspect index* was developed to represent the influence of aspect as shown in Table 1. Sites that receive high solar radiation such as on gentle slopes and those facing north and north-west have low indices, whilst those that receive low solar radiation such as those with steep south and south-easterly facing slopes have high indices.

*v) Depth* – it is widely recognised that OC values decrease with depth, so it is necessary to include this factor in the model. Only profiles with topsoil upper A horizon down to 20 cm were included in the dataset and the thickness of this horizon was also included as a covariate.

**Table 1: Aspect index**

| Slope% | N | NE | E | SE | S | SW | W | NW |
|---|---|---|---|---|---|---|---|---|
| <15 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| 15-30 | 2 | 3 | 4 | 6 | 5 | 4 | 2 | 1 |
| >30 | 3 | 5 | 6 | 10 | 8 | 6 | 3 | 2 |

*Statistical analysis and validation*

Approximately 15 percent of points from the dataset, stratified by jurisdiction, were randomly extracted for validation purposes. Over the remaining approximate 85% of points (i.e., the training data) multiple linear regression relationships, with associated statistics, were derived for soil OC based on the above-mentioned covariates, using R statistical software. A natural log transformation was applied to the OC% to address the observed skewness in results. Validation involved the determination of root mean square error (RMSE), mean error (ME), mean absolute error and median absolute error. A plot of observed versus predicted OC% values with associated statistics was prepared. Lin's concordance correlation coefficient (CCC) was used to measure the level of agreement of predicted values with a 45° line through the origin (Lin 1989).

## Results and Discussion

The relationship between topsoil organic carbon and the key environmental factors influencing native vegetation sites in eastern Australia is expressed by the multiple linear regression model of Equation 1, with associated statistics presented in Table 2.

$$\log_e(\text{OrgC\%}) = 4.30 + 0.00035(\text{precip}) - 0.097(T_{max}) - 0.022(\text{silica\%}) + 0.047(\text{aspect}) - 1.55(\text{depth}) \quad (1)$$

**Table 2. Regression parameters of final model**

| | Estimate | Std error | t value | Pr(>\|t\|) |
|---|---|---|---|---|
| Intercept | 4.30 | 0.029 | 15.1 | <0.0001 |
| Precip | 0.00035 | 0.000094 | 3.7 | 0.0002 |
| Tmax | -0.097 | 0.0074 | -13.1 | <0.0001 |
| Silica | -0.022 | 0.0030 | -7.3 | <0.0001 |
| Aspect | 0.047 | 0.017 | 2.7 | 0.0063 |
| Depth | -1.55 | 0.64 | -2.4 | 0.017 |

$R^2 = 0.53$; n = 407; residual std error = 0.63;
F statistic = 91.2; model P value <0.0001

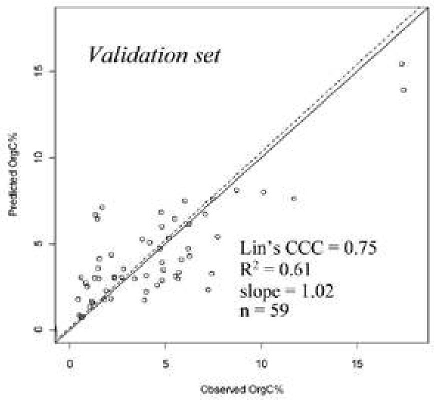


**Figure 1. Observed vs predicted soil OC values**

Validation of the relationship against approximately 15% of withheld points, following back transformation to original non-log scale, revealed an RMSE of 2.3% OC and a mean error of +0.20% OC. The mean and median absolute differences between observed and predicted values were 1.7% OC and 1.5% OC respectively. These errors appear quite high, but when the high variation in soil organic carbon in natural ecosystems is considered (e.g. Wilson *et al.* 2010), then they may be deemed reasonable. The mean error reflects an overall slight under estimation of OC values by the model. A plot of observed versus predicted organic carbon values is shown in Figure 1. It demonstrates a line of best fit (dashed line) only slightly offset from the 1 to 1 concordance line with a moderate Lin's concordance correlation coefficient and $R^2$ value.

The model presented here can provide a useful first approximation of soil OC levels under undisturbed native vegetation as would represent pre-native vegetation clearing conditions. It can inform on the potential capacity of soils across eastern Australia to store organic carbon, i.e. the likely upper limits of OC mass per unit area. It may assist in deriving the initial OC levels in carbon modelling and accounting tools such as RothC and FullCAM. It provides an alternate mechanism to the current approach of FullCAM of deriving coarse estimates based on broad soil units and limited laboratory data. The inclusion of location specific climate, parent material and aspect indicators may significantly improve the FullCAM predictions. The additional inclusion of soil texture in the model was found to slightly improve its strength, but this covariate relies on actual field data for which there is no complete spatial layer, thus it cannot be used for digital soil map production, so the results are not presented here.

The model could potentially be used to develop a first approximation digital soil map of pre-native vegetation clearing topsoil organic carbon. Such a map could be made available for incorporation into the carbon accounting models for different areas. Further investigation will test the feasibility of obtaining all the required co-variates and producing such a digital soil map.

The model sheds light on the factors controlling soil OC in a natural state. The quantitative influence of each covariate can be estimated by their partial regression coefficients, for example:

Precipitation – each 100 mm increase equates to 0.035 $\log_e$ % (3.6% proportional) increase in soil OC
Tmax – each 1 degree C temperature rise equates to 0.097 $\log_e$ % (9.2% proportional) decrease
Parent material - each 10% silica% increase equates to 0.22 $\log_e$ % (19.7% proportional) decrease
Aspect - each unit increase in the index equates to 0. 046 $\log_e$ % (4.8% proportional) increase
Depth - each 1 cm increase equates to 0.015 $\log_e$ % (1.5% proportional) decrease

The overall potential influence of each covariate on soil OC levels can be estimated by multiplying the partial regression coefficients by the maximum likely variation of each covariate. The *t* values in Table 2 also indicate the level of confidence that can be attached to the influence of each variable in the model. It appears that temperature is the dominant controlling influence, being a primary controller of organic decomposition rates and evaporation/soil moisture regimes. Parent material composition appears to have the next highest influence; clearly the more mafic materials are giving rise to higher fertility soils that is reflected in higher OC levels. Annual precipitation also appears to play a major role, but is clearly subordinate to temperature and parent material factors. Aspect plays a smaller but still significant role, as moist south-facing slopes have higher OC levels than drier north-facing slopes; the increase being more pronounced in steeper terrain. The depth factor does not come out strongly in this model, being restricted to depths less than 20 cm. The statistically weak influence of topography (apart from its connection with aspect) is noteworthy.

## Conclusion

A relatively simple and easily applied model for the estimation of pre-clearing soil organic carbon levels over eastern Australia has been developed. There are several sources of potential error and weakness in the model, such as simplification of covariate categories, lack of representation of some environments, and possible laboratory analytical errors. Nevertheless, it appears to provide satisfactory predictive performance given the wide and often inconsistent variation in soil organic carbon levels in natural ecosystems. The model can give useful first approximations of pre-native vegetation clearing topsoil OC levels and may provide a basis for improving these estimates in Australian soil carbon accounting models. It can also provide useful insights into the environmental factors controlling organic carbon levels in Australian soils.

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# Tuesday 4 December

## SOIL CARBON AND CLIMATE CHANGE (SEQUESTRATION - ACCOUNTING)

*Presented Posters*

# Influence of soil parent material and clay content of Dermosols on soil carbon content in Tasmania

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A wide variety of Dermosol order soils were sampled and analysed for Total Organic Carbon (TOC) throughout Tasmania as part of the Soil Carbon Research Project (SCaRP). Dermosols are the most common soil order in Tasmania but they vary considerably in colour, clay content and texture. Much of this variability can be attributed to differences in Soil Parent Material (SPM). Alluvial parent materials often result in dark grey sandy loam or loam A horizons. Igneous parent materials often result in brown clay loam A horizons with well developed blocky structured B horizons. Tertiary sediments often result in grey loam A horizons grading to yellow-brown light clays. These three broad parent material classes have distinctly different mineralogy and provide a good case study to examine whether these differences in SPM and associated mineralogy influence TOC. The effect of SPM on TOC within the Dermosol order was analysed by a spatial mixed model following normalisation of explanatory variables such as rainfall and cropping intensity. Clay contents were estimated by texture tests performed on homogenised subsamples. The effect of clay content on TOC was analysed by attributing a quantitative clay content value, so that regression models could be used.

# Vertical and lateral variability of soil carbon stocks in an apple orchard: Implications for soil carbon monitoring & carbon footprinting

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Soil carbon sequestration can mitigate climate change, enhance the provision of ecosystems services, strengthen global food security and ensure the sustainability of food production systems but could also benefit farmers by reducing the carbon footprint of their products. However, correctly monitoring soil carbon stocks demands a statistically significant and powerful detection of small changes over time. The number of samples required depends primarily on the embedded variability. Even though prevailing inputs of carbon in the soil are from the rhizosphere, particularly in the deeper soil horizons, very little soil carbon stock data are available for orchard systems. We carried out sampling in the tree row and the inter-row of a 4 year-old commercial apple orchard block. Soil carbon stocks decreased rapidly with increasing depth and ranged from 40.7 t C/ha in the 0 to 10 cm layer, down to 3.1 t C/ha in the 90 to 100 cm depth. Coefficients of variations seemed to increase with increasing depth and were between 1% (10-20cm depth increment - tree row) and 40% (80-90cm depth increment - inter-row). The number of replicates required depends as well on the depth considered. For both treatments, only 64% of the carbon stock of the top metre was present in the top 30 cm of soil. Deeper depths need to be considered when monitoring soil carbon stocks. Horizontal variability was assessed by intensively sampling 10 locations across the orchard block. From these measurements we will develop rules for verifying changes in soil carbon stocks for multiple purposes.

# Allocation and storage into soil organic matter of 14C fixed by perennial-based grass pasture

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There has been growing interest in the use of perennial-based pastures to increase the sequestration of carbon (C) in soil to offset emissions of greenhouse gases. The aim of this study was to measure the allocation of photosynthate into roots and organic matter fractions below-ground. A $^{14}CO_2$ pulse-labelling technique was applied to an eleven-year old Kikuyu pasture in the field. Plants were labelled in late Spring and early Autumn following a simulated grazing. Above-ground plant material and soil to 70 cm were sampled 1 week, 6 weeks and 52 weeks after labelling. After soil drying, roots (> 2mm) were separated from the soil in five separate depth intervals prior to fractionating organic matter between the particulate organic matter (POM) fraction (>50 µm < 2 mm) and the humus fraction (< 50 µm). Each soil fraction as well as shoots and roots were finely ground before combustion in a Roboprep CN analyser and trapping $CO_2$ in alkali for analysis of total C and in scintillation cocktail for $^{14}C$ determination. Analyses indicated that more than 40% of the $^{14}C$ applied was allocated below-ground. The majority (32% of $^{14}C$ applied) was located in the top 10 cm, where initially the distribution was 25% in roots, 6 % in POM and 1% in humus. After 12 months, this reduced to 7% in roots with a slight increase in POM to 7.5%, but the humus fraction remained unchanged at 1%. These results imply that the build-up of carbon below-ground is largely in the less protected POM fraction with little or no accumulation in the more stable humus fraction.

# Soil carbon stocks in Victorian agricultural soils: Influence of climate, soil type and management

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In order to assess the potential for soil carbon sequestration on farms in Australia, improved understanding is required of current carbon stocks and the relative influence of climate, soil characteristics and management on soil carbon. In this study, part of the Australia-wide Soil Carbon Research Program, we measured soil carbon stocks under four management classes (continuous cereal cropping, cereal cropping-pasture rotations, pasture grazed by dairy cattle, and pasture grazed by sheep or beef cattle) in key production regions and soil types in the state of Victoria. Six hundred sampling sites were selected according to region, soil type (Australian Soil Classification) and management class, with approximately 25 representations of each region x soil x management combination. Annual rainfall across the sites ranged from 250 to 1800 mm. At each site (25 x 25 m area in a farm paddock), soil was randomly sampled to 30 cm depth in 10 cm increments, and a 10-year paddock management history was obtained from the landholder. Soil organic carbon was measured by dry combustion with infrared detection. Data were analysed using REML variance components analysis and regression. In this paper we describe relationships between soil organic carbon and climate, soil properties and management.

# Measuring soil carbon sequestration for carbon trading purposes in the Upper Hunter Valley

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Currently there are no approved methodologies for soil carbon trading under the Carbon Farming Initiative (CFI). Measurement of baseline soil carbon levels and the estimation of potential sequestration rates under best management practices are significant barriers (time & cost) to progressing methodologies. In a soil carbon project undertaken in the upper Hunter Valley the sampling and analysis method utilised the findings of the CAMBI project (Murphy et al. in prep). Soil samples were taken in a method similar to those described in the Central West CMA SoilWatch Project; initial samples were randomly collected from a representative area in each 10ha of land using a 35m diameter circle and 10 x 0-10cm soil cores taken. One composite sample per site was then analysed with the objective of using the appropriate ANOVA to quantify site differences on soil carbon stocks. This allows for comparisons between paddocks and for comparisons over time to be made.

Once the results were received a pedotransfer function was used to determine the carbon content in the top 30cm. To verify this function a further 10% additional soil cores were taken to 30cm which would satisfy Kyoto requirements (Baldock. pers com). At this stage the results of this sampling and analysis system are encouraging and indicate that it is possible to determine soil carbon to 30cm depth with a high degree of certainty without the cost of sampling to this depth at all sites.

# Soil carbon model Upper Hunter valley - a way to offset carbon emissions from the mining industry

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The purpose of this project is to produce a GIS model that collates information on land use, land management and soil type to develop a spatial model to assess the regional potential to sequester carbon. This information will be compiled using aerial photography and cadastre information to define land use. District agricultural yields and management practices will be used to develop current and historical land use management regimes. Finally, soil landscapes together with topographical information will be used to estimate soil texture. Once compiled, the GIS model will be able to define a unique set of variables for any point within the study area; these variables will be analysed using the RothC and CENTURY carbon models to develop a spatial model of regional soil carbon levels. This model will be calibrated by assessing soil sampling data collected by NSW Soil Condition and Land Management within Capability 2008 baseline project by the Department of Environment, Climate Change and Water.

Additionally, best practice land use management regimes will be used to estimate soil carbon sequestration potential of the region in the future. Further, the potential for these regimes to offset agricultural production and carbon emissions for an operational coal mine will also be investigated.

# Tuesday 4 December

## SOIL FERTILITY AND SOIL CONTAMINANTS (CONTAMINANTS)

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# Natural toxins in the environment – why care?

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Plants, animals, bacteria, fungi, algae and other organisms produce a large diversity of bioactive metabolites, some of which are very toxic such as aflatoxins, alkaloids as cytisin and strychnine, peptides such as amanitin, or lectins (glycoproteins) such as ricin. In addition natural toxins are likely to be abundant in the environment due to the many sources and continuous production of large quantities – often in annual amounts of kilograms per hectare. Even crop plants contain toxins, e.g. cyanogenic glucosides in sorghum and cassava, glycoalkaloids in potato, glucosinolates in rapeseed, benzoxanoids in wheat and isoflavones in soya bean. Despite the widespread occurrence and toxicity, natural toxins seem more or less neglected as contaminants in soils, surface water and groundwater, even though the toxins are deposited in the open landscape in intimate contact with soil and water. Studies in our laboratory of more than ten different natural toxin groups show that the rates of initial degradation of natural toxins is high to intermediate (half lives from hours to one month) although strongly depending on specific soil conditions such as pH, moisture, temperature, microbial activity, mineralogical properties, and appropriate enzymes. For some toxins complete initial degradation can take months. Some natural toxins form from precursor compounds, e.g. cyanide which is produced from cyanogenic glucosides or isothiocyanates which form by hydrolysis of glucosinolates. This further complicates prediction and modelling of soil and water concentrations of the toxins. Many toxins occur as glucosides which are very weakly sorbed to soils resulting in high mobility and leaching potential. For instance the glucosidic and carcinogenic ptaquiloside produced from Bracken has been detected in upper groundwater. Several of the tested toxins are more toxic to soil organisms than conventional pesticides and if commercialised the natural toxins would never be approved for use as pesticides. Natural toxins represent an important but almost non-explored group of emerging contaminants and needs attention as new biomedicinal crops and gene modified allellopathic crops as well as spreading of invasive plants and other organisms lead to increasing releases of these toxins to the environment. The presentation provides examples of our work with selected classes of natural toxins in the environment during the last 10 years.

# Prediction of total petroleum hydrocarbon concentration in soils using diffuse reflectance infrared spectroscopy

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Diffuse reflectance infrared Fourier transform (DRIFT) spectroscopy, with partial least-squares (PLS) regression, was used to develop calibration models for the prediction of total petroleum hydrocarbon (TPH) concentrations in soils spiked with TPH and in field-contaminated soils from sites in south eastern Australia (n=205). Concentrations ranged from zero to 32,600 mg kg$^{-1}$ and PLS models were derived for three concentration sets; 0-5,000, 0-15,000 and 5,000-32,600 mg kg$^{-1}$. For each concentration set, a number of selected frequency ranges were tested. The full aliphatic alkyl (-CH) stretching vibrations were found to be the most sensitive for intermediate TPH concentrations, with NIR at 4500-4100 cm$^{-1}$ and MIR at 3,000-2600 cm$^{-1}$. In particular, the MIR range included two specific alkyl peaks, one at 4950 cm$^{-1}$ and the other near 2730 cm$^{-1}$, both shown to have a strong correlation with TPH at low and high TPH concentrations, respectively. The PLS regression analysis using the 3000-2600 cm$^{-1}$ MIR frequency region, for a TPH range of 0-15000 mg kg$^{-1}$, was the most accurate model with a coefficient of determination ($R^2$) = 0.92 and root mean square error (RMSE) = 601 mg kg$^{-1}$. Further validation was carried out by random selection of a separate 50-field sample “test” set that resulted in an $R^2$ = 0.89, RMSE =767 mg kg$^{-1}$, and a residual prediction deviation (RPD) = 2.7. Results showed that the infrared PLS method was capable of good precision and accuracy for determination of TPH concentrations, with minimal sample preparation and thus application for rapid in-field TPH screening.

# Linking 3D-pore structure to the distribution of copper and organic matter in Andisols

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Sustainable horticulture depends on the integrity of various soil functions, which in turn directly depend on soil structure affecting aggregation, root growth, biota, and liquid and gas permeability. A necessary step to develop eco-efficient horticultural systems is to identify how soil structure can be preserved and manipulated by orchard management.

We hypothesized that changes in pore structure resulting from the feedback mechanisms between organic carbon (OC) management and soil biota can be captured with 3D X-ray computed tomography (CT), and that such changes can explain differences in the filtering function of soils. We compared soils that have been under organic and integrated kiwifruit production for 30 years. Copper was chosen as model contaminant to analyze the mobility of contaminants in orchard soils. It is widely used to control fungi and bacterial diseases in orchards, which led to concerns about its potential accumulation, toxicity and mobility.

The 3D-macro pore networks and some morphological parameters of undisturbed topsoil cores from the vine-rows of both orchards were derived with X-ray CT combined with image processing techniques. Then, a pulse of bromide and copper was leached through the same cores. Leachate samples were analyzed for both solutes. After leaching of three pore volumes, the soil cores were cut into slices, which were analyzed for total copper and OC contents. We discuss the correlation of the 2D-distribution of total copper and OC with the 3D-pore structure, as well as its impact on copper mobility.

# Assessment of arsenic accumulation in lettuce grown in a contaminated Ferrosol

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## Abstract

The re-development of arsenic (As) contaminated land for residential purposes presents a human and environmental health risk. Such land requires remediation to manage risks if residents produce fruit and vegetables on their land. The current phytotoxicity investigation level (PIL) used in NSW is 20 mg total As kg$^{-1}$ soil. This study's aim was to directly assess As plant availability using a quick growing, common garden vegetable grown in an As contaminated Ferrosol. A randomised complete block design pot trial growing buttercrunch lettuce (*Lactuca sativa*) in five As soil concentrations (10, 86, 169, 315 and 656 mg As kg$^{-1}$) with four replicates was undertaken. After six weeks plants were harvested and leaf biomass and plant tissue As concentration measured. Results showed plant growth was uninhibited at all As concentrations. Significantly higher biomass was shown in the 86 As kg$^{-1}$ treatment than both the control and the highest soil As concentration. The lowest plant biomass was at the 315 mg As kg$^{-1}$. Plant tissue As concentration did not exceed 1 mg As kg$^{-1}$ (dry weight) for all treatments. The current health limit for human consumption is 1 mg As kg$^{-1}$ (wet weight of food). The use of 20 mg As kg$^{-1}$ as a default remediation goal for As concentration is inappropriate in this Ferrosol. Site-specific remediation goals, using the health investigation level (HIL) of 100 mg kg$^{-1}$, should consider plant As availability. A pot trial using a common garden vegetable may provide a rapid and reliable soil assessment method.

## Key Words

Arsenic, contamination, ferrosol, lettuce, phytotoxicity, remediation

## Introduction

At some sites throughout Australia, arsenic (As) based pesticides, used extensively in the early to mid 1900s, have caused soil contamination in excess of 1000 mg As kg$^{-1}$ soil (Smith *et al*., 1998). Many of these highly contaminated sites are cropped areas such as abandoned banana farms and decommissioned livestock dip sites. NSW State Environmental Planning Policy No 55 – Remediation of Land, (SEPP 55) (DUAP, 1998), requires planning authorities to consider land contamination as a land use constraint and to determine appropriate actions to be undertaken as part of their redevelopment.

As part of the conditions of a re-development approval, a remediation action plan (RAP) may be formulated to rehabilitate the contaminated site. The National Environment Protection Measures (NEPM) and current NSW guidelines for As site contamination assessment are based on total As concentration. The remediation goals for residual soil As normally use a health investigation level (HIL e.g. 100 mg As kg$^{-1}$ for residential areas) and a phytotoxicity investigation level (PIL e.g. 20 mg As kg$^{-1}$) (NSW EPA, 1998). The HIL forms part of the Australia-wide NEPM and is based on significant research. However, soil As phytotoxicity, and hence the appropriate PIL, may change significantly with the soil type and properties (e.g. pH, clay content and type, organic matter, iron and aluminium oxides, phosphate-sorptive characteristics) and plant species (e.g. As sensitivity, preferred uptake and accumulation characteristics).

Soil total As concentration alone is an inappropriate indicator for assessing phytotoxic impacts at contaminated sites. To improve the assessment of As phytotoxicity in soil, a direct plant based measure of soil As availability for any soil type is needed. An investigation was undertaken for an As contaminated Ferrosol in the Tweed Shire (NSW) to determine the phytotoxicity and plant uptake of As. A germination trial was used to assess the toxicity of the soil at varying levels of the soil contaminant and it showed the soil had limited As toxicity even at soil total As concentrations of 656 mg kg$^{-1}$ (Foster *et al*., 2010). A pot trial used to investigate plant uptake of As from the same soil found that radish plants did not show a significant loss in plant production, even at the highest soil As concentration (Foster *et al*., 2011). However, the plant root tissue accumulation of As at the highest soil As concentration of 656 mg kg$^{-1}$ was in excess of the safe food As level. That trial focused on the root mass as a human food source, however plant tops are also consumed and these plant parts may accumulate different concentrations than roots.

Plant sensitivity to As varies considerably between species (Woolson *et al*., 1973). To ascertain whether plants with edible leaves may assimilate enough As to pose a health risk to consumers, this study investigated plant uptake of soil As using buttercrunch lettuce (*Lactuca sativa*).

## Methods

### *Soil sampling*

Soil was taken from a proposed redevelopment site on a Red Ferrosol at Bilambil Heights on the North Coast of NSW on 11 November 2007. Based on a preliminary study, samples were collected from 0.00 m (surface) to 0.25 m below the natural surface level (NSL) at locations selected to represent the spectrum of soil As concentrations at the site. Sampling equipment (auger and spade) was cleaned by brush (dry method) to remove any residual soil from the sampling apparatus before the next sample event. The samples were transferred to 'clip-seal' plastic bags that were expelled of air, sealed and placed in a cool esky for transport to the laboratory and further testing.

### *Pot trial*

Plant production and As uptake of buttercrunch lettuce (*Lactuca sativa*) was assessed in a pot trial with a randomised complete block design. Treatments consisted of five samples of the study soil with As concentrations of 10 (as control), 86, 169, 315 and 656 mg kg$^{-1}$. Four replications of each soil sample were potted in lined, 100 mm pots and planted with lettuce seeds. Pots were weighed and watered to field capacity as determined in the laboratory. A layer of vermiculite was placed on the soil surface to reduce evaporation and the pots were placed in a glasshouse. The pots were rotated regularly to reduce light and heat variation.

Following germination, plants were thinned, leaving pots with one lettuce seedling. Pots were reweighed regularly and watered to maintain field capacity with the mass of water lost recorded. Four pots containing soil only were included to determine the quantity of water lost to evaporation. At the end of the trial period the lettuce leaves were harvested, dried and weighed prior to preparation for laboratory analysis for tissue As concentration.

### *Laboratory analysis*

Soil samples were analysed by NATA-accredited laboratories to determine soil characteristics and total As concentration. In addition, an assessment of the water content at field capacity was undertaken to determine the amount of water required per gram of soil to reach field capacity. Following the six-week trial, the plants were washed, weighed, oven-dried (60ºC) and re-weighed. Oven-dried lettuce leaves were analysed to determine tissue concentration of As.

### *Statistical analysis*

All statistical analysis used a 0.05 ($\alpha$) level of significance (to minimise a type II error) and only the resultant $R^2 > 0.70$ were reported. The pot trial analysis was undertaken in two parts: biomass assessment and plant tissue As accumulation using a two-factor without replication analysis of variance (ANOVA).

## Results and discussion

The results for the dry matter production of the lettuce leaves are shown in Table 1.

**Table 1. Average lettuce leaf dry matter (g) with increasing soil total As concentration showing significant differences ($\alpha$ = 0.05, lsd = 0.236).**

| Soil As (mg kg-1) | Mean Leaf Dry Wt.(g/pot) |
|---|---|
| 10 | 0.608 b |
| 86 | 0.980 c |
| 169 | 0.433 ab |
| 315 | 0.315 a |
| 656 | 0.650 b |

The fresh weight of the lettuce ranged from 3 to 10 g (the small size was not unexpected given the size of pot). As small plants are more susceptible to soil chemical stressors (e.g. As) it was not considered necessary to grow the lettuce to a larger size. The soil samples were air-dried and ground to eliminate structural variance and ensure maximum soil to root contact throughout the growth period.

The analysis indicates that the smallest lettuce dry biomass was recorded for plants grown in 315 mg As $kg^{-1}$ and the greatest biomass was recorded in 86 mg As $kg^{-1}$. However, there was no significant difference between the control (10 mg As $kg^{-1}$) and 656 mg As $kg^{-1}$.

The As tissue concentration in plants grown in soil containing 656 mg As $kg^{-1}$ was much higher than in plants grown in all other soil samples where As tissue concentration was below 1 mg $kg^{-1}$ (Table 2).

**Table 2. Average lettuce leaf tissue concentration of As (mg kg-1 dry matter) with increasing soil total As concentration showing significant differences ($\alpha$ = 0.05, lsd = 0.641).**

| Soil As (mg kg-1) | Leaf As concentration (mg kg-1 dry matter) |
|---|---|
| 10 | 0.125 a |
| 86 | 0.425 a |
| 169 | 0.500 a |
| 315 | 0.575 a |
| 656 | 5.300 b |

The increase in leaf tissue As concentration showed a strong linear relationship with increasing soil As as shown by the regression equation below.

$T = 0.0084\ (\pm 0.002)* S - 3.472\ (\pm 0.588)$
$R^2 = 0.8262$; standard error of the estimate = 0.8488
Where T = leaf tissue As Conc.; S = soil total As.
Note: Error term is the 95% confidence interval for the coefficients

The regression equation gradient suggests that the lettuce accumulates less than 1% of the total soil As of this Ferrosol.

This experiment showed that there was lettuce plant uptake of the soil As and accumulation in the leaf at concentrations that were below the accepted human health limits used in Australia at all but the highest soil As concentration (656 mg $kg^{-1}$). Interpretation (in terms of human health risk) of these dry matter results is augmented by considering the wet weight implications on As concentration. Considering the moisture content of the tissue samples averaged 90%, the actual wet weight As concentrations may be approximately one-ninth of the reported As concentration as dry weight. The highest tissue As in wet weight terms is significantly diminished to approximately 0.59 mg $kg^{-1}$. This is a recognised safe level of As in food (National Food Authority, 1993). Comparing this tissue concentration outcome to a previous radish root As accumulation trial, the wet weight As concentrations in the radish roots were less than 1 mg $kg^{-1}$ in all but the highest soil As concentration (656 mg $kg^{-1}$). The lettuce and radish responses appear similar at soil concentration of 169 mg $kg^{-1}$ or less.

## Conclusions

The As contamination concentration in the test ferrosol was of low availability to the plant. The lettuce plants did not show a significant loss in plant production, even at the highest soil As concentration. Lettuce plant uptake (in terms of consumable wet leaf concentration) was below the maximum safe food As level of 1 mg $kg^{-1}$ for all soil As concentrations. A plant based examination of phytotoxicity and the setting of remediation targets must take cognisance of both the plant response in terms of survival and uptake in edible parts. The remediation target in this case may be more appropriately set by specific soil assessment using common garden vegetable plants and in this instance at the Health Investigation Level (HIL) of 100 mg As $kg^{-1}$ instead of the default NSW PIL of 20 mg $kg^{-1}$. A two-stage approach of germination testing followed by plant uptake assessment (in both a root and leaf vegetable) may be an appropriate procedure to define remediation targets.

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## SOIL FERTILITY AND SOIL CONTAMINANTS (CONTAMINANTS)

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# Organochlorine pesticides in irrigated Vertosols of the Namoi Valley, north-western New South Wales

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Organochlorine pesticides (OCPs) such as DDT and DDE have been detected in the surface 0.2 m of Vertosols in the lower Namoi Valley of north western New South Wales even though they have not been applied to crops since 1982. However, their presence in the deeper soil horizons has not been investigated. The objective of this study was to determine if OCPs were present to a depth of 1.2 m in Vertosols under irrigated cotton farming systems in the lower Namoi Valley. Soil was sampled from the Australian Cotton Research Institute, near Narrabri, and two cotton farms near Wee Waa and Merah North in northern New South Wales. The OCPs and metabolites detected in order of concentration were: DDE > endosulfan sulphate >endrin > α-endosulfan > β-endosulfan >DDT and DDD. DDT was sprayed extensively in the lower Namoi Valley up to the early 1980's and may explain the persistence of DDE, a metabolite of DDT, in the majority of soil samples. Dicofol and Dieldrin, previously undocumented in these soils were also detected. The movement of OCPs into the subsoil of Vertosols may occur when irrigation or rain transports soil colloids and organic matter via preferential flow systems into the deeper layers of a soil profile.
The persistence in soil of OCP's applied to cotton crops grown more than two decades ago suggests that they could enter the food chain. Their presence at depths of 1.2 m suggests that they could move into groundwater that may be used for domestic and stock consumption.

# Arsenic uptake of vegetables under contaminated irrigation

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The uptake of arsenic (As) by vegetables from contaminated irrigation water in a soil environment is poorly understood. During a pot trial, four vegetable species (carrot, radish, spinach and tomato) were cultivated in a free-draining soil and irrigated with various As levels ranging from 50-1000 ug L-1, prepared from sodium arsenate heptahydrate ($Na_2HA_sO4.7H_2O$) salt. Irrigation water was maintained in the soil at two water management regimes, aerobic (70% field capacity for all studied crops throughout the experiment), and saturated-aerobic (110% field capacity initially followed by aerobic till next irrigation) for carrot and spinach only. The irrigation was repeated every 10 days for aerobic plants and fortnightly for saturated-aerobic plants. The soil used in experiment was an acidic silt loam, low in nutrient elements and arsenic. The results revealed that As concentration in edible parts of radish (taproot) and spinach (leaves) was significantly ($p<0.05$) higher under 1,000 ug As L-1 irrigation than under other treatments. In most cases, a comparatively higher concentration of As in the leaves of spinach was observed under saturated-aerobic conditions compared with irrigation at aerobic situation. The soil concentration of As increased as a function of the irrigation concentration, indicating As was adsorbed onto the soil. Arsenic concentration in leaves of spinach exceeded the maximum permissible concentration (MPC) for inorganic As (0.05 ug g-1 fresh weight) for 100, 200 and 1,000 ug As L-1 treatments under saturated-aerobic conditions, showing a potential risk to human health.

# The effect of urinary nitrogen content and DCD nitrification inhibitor on nitrogen emissions from grassland lysimeters in Ireland

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In grazed pasture systems, the nitrogen (N) contained in a cattle urine patch may be up to 1200 kg N ha$^{-1}$. The majority of this N is in excess of plant requirements and is vulnerable to environmental loss. In this study, cattle urine was applied at 5 rates of nitrogen, 0, 300, 500, 700 and 1000 kg N ha$^{-1}$ to soil monolith lysimeters in late autumn in Ireland. Dicyandiamide (DCD) nitrification inhibitor was applied in solution form at 30 kg DCD ha$^{-1}$ in two split applications following urine. Measurements of gaseous N emissions, nitrate ($NO_3^-$) leaching and pasture N uptake were made for a calendar year following urine application in two consecutive experiments. Increasing the rate of urine N applied increased the cumulative nitrous oxide ($N_2O$) emissions, $NO_3^-$ leaching and pasture N uptake in years one and two, and these relationships are described. The application of DCD reduced the cumulative $N_2O$ emissions and $NO_3^-$ leaching in year one but not in year two. The pasture N uptake response to the increasing urine N rate was variable. A nitrogen balance for the fate of urine patch N attempts to answer the question – where is the missing N? We suggest the use of nitrification inhibitors and the reduction of urinary N content as two mitigation strategies for N loss from grazed pasture.

# Release patterns of controlled release fertilisers and nitrogen use efficiency

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Increasing the nitrogen use efficiency (NUE) of crops enables greater yield and protein content to be achieved per unit of N supplied. Controlled release fertilisers (CRFs) can match the supply of nitrogen to crop needs more closely, meaning less is lost to nitrate leaching, ammonia volatilisation and denitrification. This reduces the environmental impact of fertiliser use, and potentially increases economic efficiency. Inappropriate selection of fertiliser means release rates can be too fast, resulting in losses early after application, or too slow, resulting in inadequate supply to the growing plant, and nitrogen release after harvest. Variations in soil texture, temperature, moisture, EC and pH may alter the nitrogen release rates from these fertilisers and retention in soil. Incubation experiments were conducted to measure the release pattern of nitrogen over 56 days from granular urea and three controlled release fertilisers (polymer coated urea, polymer coated sulfur coated urea and lipid coated urea) in three horticultural soils from south-eastern Australia. Data obtained from the incubation experiment guided the selection of a soil and fertiliser combination with a release pattern showing promise to improve NUE relative to granular urea. Lettuce was grown in pots in a glasshouse with four treatments: a CRF, a single application of urea, split application of urea, and control. Leachate was collected and N content of leachate was measured. After harvesting, the yield and plant biomass were measured. Nitrogen use efficiency was then calculated for each treatment.

# Quantifying the effects of land management interventions on water quality in the Coal River Catchment

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In this study we examined spatial variation in water quality and its relationship to riparian land management in the Coal River Valley, SE Tasmania. Historical water quality data from stations at Baden, downstream of the Craigbourne Dam, Richmond and White Kangaroo Rivulet collected between 1999 and 2008 were obtained from DPIPWE. Riparian land use within one kilometre of the river was assessed and digitised using 2005/7 colour aerial photographs and water quality data for that period were examined and possible linkages investigated. The historical data demonstrates complex spatial patterns of in-stream water quality parameters in the Coal River Valley. There was a significant negative correlation between dissolved oxygen and water temperature observed in the Coal River. However, positive correlations were found between stream flow and rainfall with turbidity at all stations except downstream of the reservoir. Stream nitrogen and phosphorus showed a significant relationship with rainfall at Richmond. Positive correlations of turbidity with nitrogen and phosphorus show nutrients bound to sediment are a likely source of nutrients in the river. Subcatchment riparian land use and water quality data from 2005/7 suggests that lower turbidity at Richmond (2.82 NTU) compared to White Kangaroo Rivulet (4.25 NTU) and Baden (4.70 NTU) may be due to the impact of higher percentages of riparian vegetation. Similarly, riparian land management works such as planting native vegetation and fencing on the river banks could have reduced the sediment load and nutrient in the river by preventing erosion caused by stock access to river water.

# Can the urease inhibitor NBPT improve nitrogen use efficiency from urea surface applied to ryegrass?

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Ammonia ($NH_3$) loss from surface applied urea can be as high as 50% of applied nitrogen (N). A urease inhibitor can reduce $NH_3$ volatilisation meaning more N is retained for plant uptake. This paper reports results from a field experiment on a ryegrass seed crop grown on a Sodosol in southern Victoria. Ammonia loss from no fertiliser and surface applied granular urea (40 kg N/ha) and urea amended with a urease inhibitor (Green Urea 14®) in autumn and spring were compared using micrometeorological techniques over a 4 week period.  Soil mineral N and biomass production data was collected. Ammonia loss depended upon climatic conditions with 30% loss of applied N in autumn and 2% loss in spring (rain fell within 24 hours of fertilisation). The urease inhibitor reduced $NH_3$ loss to 9% of applied N in autumn and to 1% in spring. Granular urea was hydrolysed within 3 and 7 days in autumn and spring. The urease inhibitor increased the retention time of urea, and in autumn this led to greater retention of ammonium ($NH_4^+$). In spring $NH_4^+$ and $NO_3^-$ levels rapidly dropped to those of the unfertilised treatment. Biomass production increased with the use of urea by 8 and 18 kg dry matter (DM)/unit of N in autumn and spring respectively. The urease inhibitor increased biomass production compared to urea in autumn (an additional 1 kg DM/unit of N) but not in spring. The benefit of using the urease inhibitor was dependent on climatic conditions, and the biomass response was less than expected based upon the N savings.

# Tuesday 4 December

## SOIL AND LAND DEGRADATION

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# Spatial and temporal prediction of rainfall erosivity and its impact on soil erosion in New South Wales

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## Abstract

Historical rainfall data indicate considerable inter-annual variability and significant change over the past 100 years in New South Wales, Australia. While this change in rainfall amount and intensity is expected to have significant effect on rainfall erosivity and soil erosion, the magnitude of the impact is not well quantified because of the non-linear nature of the relationship between rainfall amount and rainfall erosivity, and the extreme nature of large erosive events. The aim of this study was to predict and map the spatial and temporal variations of rainfall erosivity across New South Wales for soil erosion assessment and monitoring using the Revised Universal Soil Loss Equation. In this paper, we present a set of equations for calculating RUSLE rainfall erosivity suitable for New South Wales based on commonly available daily rainfall data. We calculated both mean and extreme erosivity values (90th percentile) on monthly and annual bases. We produced finer scale (up to 30m resolution) maps of rainfall erosivity using appropriate spatial interpolation techniques. We implemented the calculation in a geographic information system using automated processes so that it is efficient, repeatable and easily updated. The predicted rainfall erosivity was compared with previous calculation using detailed pluviograph records in New South Wales showing good agreement ($R^2 = 0.89$). The rainfall erosivity products are being used to model the impacts of rainfall changes on soil erosion spatially and temporally across New South Wales at regional and community scales to inform measurable and cost effective means in sheet erosion identification and rehabilitation.

## Key Words

Rainfall erosivity, soil erosion, precipitation, RUSLE, GIS.

## Introduction

Soil erosion by water is widespread in Australia and many parts of the world. Water erosion removes fertile topsoil that contains most of the soil's plant nutrients and soil micro-organisms that contribute to soil health. Subsequent increase in sediment concentration and deposition may cause environment and water quality degradation in rivers, lakes and reservoirs.

Sheet and rill erosion of hillslopes is one of the major forms of soil erosion in New South Wales (NSW), Australia. An important feature of these erosion processes is that erosive episodes are frequently preceded by periods of low rainfall. When this happens, soils are especially exposed to water erosion where vegetative ground cover is sparse and offers little protection (Hairsine et al. 2010).

Historical rainfall data indicate considerable inter-annual variability and significant change over the past 100 years in NSW. While this change in rainfall amount and intensity is expected to have significant effect on rainfall erosivity and soil erosion, the magnitude of the impact is not well quantified because of the non-linear nature of the relationship between rainfall amount and rainfall erosivity, and the extreme nature of large erosive events. Natural resource management targets can be more easily achieved if the soil erosion hazard and land degradation is well understood, and if the mean as well as extreme erosivity values are accurately mapped at a range of temporal and spatial scales. Time series of maps as inputs to erosion prediction models are, in their own right, most valuable products to address one of NSW most serious emerging natural resource threats.

The Universal Soil Loss Equation (USLE, Wischmeier and Smith 1978) model and its successor, the Revised Universal Soil Loss Equation (RUSLE, Renard et al. 1997) are the most widely used method to predict the average annual soil loss due to water erosion. In the USLE/RUSLE, the climatic influence on water-related soil erosion is characterized by a rainfall-runoff erosivity factor, known as the R-factor. By definition, the R-factor is the mean annual sum of individual storm erosivity values, EI30, where E is the total storm kinetic energy and I30 is the maximum 30-min rainfall intensity. When factors other than rainfall are held constant, soil losses due to water erosion are directly proportional to the level of rainfall erosivity (Wischmeier and Smith 1978). When using the RUSLE, the R-factor is multiplied with other component factors relating to slope and slope-length (LS-factor), soil erodibility (K-factor), ground cover (C-factor) and soil conservation practices (P-factor) to predict the average annual soil loss per unit area.

The main objective of this study is to predict and map monthly and annual rainfall erosivity across NSW for erosion hazard assessment and monitoring. In this paper, we apply a set of equations for calculating monthly and annual erosivity values using 100 year daily rainfall data for NSW. We further produce finer scale (up to 30m resolution) R-factor maps using spatial interpolation techniques. We implemented the calculation in a geographic information system (GIS) using automated processes so that the calculation is efficient, repeatable and easily updated. The R-factor and erosion hazard maps can inform measurable and cost effective means in sheet erosion identification and rehabilitation.

## Methods

The rainfall erosivity (R) factor) is the product of two components: total energy and maximum 30-minute intensity for each storm. Historically, a rainfall erosivity contour (isoerodent) map was produced for NSW from point measurements at 29 meteorological stations across NSW with over 20 years of records (Rosewell and Turner 1992). The R-factor contour layer was interpolated to create a continuous rainfall erosivity surface (Yang et al. 2006). But only a single static layer does not represent the dynamic nature of rainfall erosivity.

Lu and Yu (2002) predicted seasonal rainfall erosivity and spatial distribution from 20 years rainfall data and produced seasonal rainfall erosivity for Australia at a ground resolution of 5 km. Further to their studies, we improved the daily rainfall erosivity model to estimate the R-factor based on NSW condition using long-term daily rainfall records (1889-2011) and implemented the calculation in GIS environment using automated scripts. Rainfall erosivity layers produced at monthly and annual bases are now available for more than 100 years.

Briefly, the model used to estimate EI30 for the month j from daily rainfall amounts can be written in the form (Yu and Rosewell 1996; Lu and Yu 2002)

$$\hat{E}_j = \alpha[1+\eta\cos(2\pi fj-\omega)]\sum_{d=1}^{N} R_d^{\beta} \quad \text{when } R_d > R_0 \qquad (1)$$

where $R_d$ is the daily rainfall amount and $R_0$ is the threshold rainfall amount to generate runoff, and $N$ is the number of rain days with rainfall amount in excess of $R_0$ in the month, and α, β, η, and ω are model parameters suitable for NSW. The parameter $R_0$ is fixed at 12.7 mm because storms with less than 12.7 mm rainfall are generally discarded in the R-factor calculations (Wischmeier and Smith 1978; Lu and Yu 2002).

The following sets of equations are used for the case of $R_0$ = 12.7 mm and $R_0$ = 0 mm (Yu, 1998, Lu and Yu 2002):

$$\alpha = 0.395[1+0.098\exp(3.26\psi / M_R)] \text{ for } R_0 = 12.7 \qquad (2)$$

$$\alpha = 1/47:\backslash 2+ 1/1: 9f yq)4/37\psi\, 0N_S{}^{*'} \text{ for } R_0 = 0.0 \qquad (3)$$

where $M_R$ is the mean annual rainfall, and Ψ is the mean summer (November-April) rainfall. A set of regional parameters depending on seasonal rainfall (Yu and Rosewell 1996) were used to estimate monthly R-factor from daily rainfall data over NSW (β = 1.48, η = 0.34).

The 90th percentile rainfall erosivity was calculated for each month representing the highest 10% monthly erosivity values. It is regarded as having high potential erosion hazard for a given month in a certain period (e.g. 10 years) if its rainfall erosivity value is greater than the 90$^{th}$ percentile value in the same month.

The original daily rainfall grids have a rather coarse spatial resolution (5 km) compared to other RUSLE factors in NSW, such as LS-factor with 30 m resolution. To produce high resolution final erosion estimation, the coarse R-factor layers were spatially interpolated to finer resolution to match with the resolution of other RUSLE factors. We used ANUDEM method (Hutchinson 1989, available in ArcGIS as TOPOGRID) as it has been proven to be better than global interpolation methods such as Kriging and Splines (Yang and Leys 2009). Automated scripts have been developed in GIS to process the daily rainfall grids, spatial interpolation and production of all the above mentioned rainfall erosivity products.

## Results

We produced monthly and annual rainfall erosivity maps (1889-2011) for NSW from daily rainfall grids using the above equations in GIS. The erosivity maps were further spatially interpolated to 30 m resolution for individual catchments or catchment management authorities (CMA), and sample maps are presented in Figure 1a. The R-factor map (at 30 m resolution) shows a strong correlation with corresponding point R-factor values calculated using pluviograph data in NSW (Rosewell 1993) with $R^2$ of 0.89 (Figure 1b).

a b

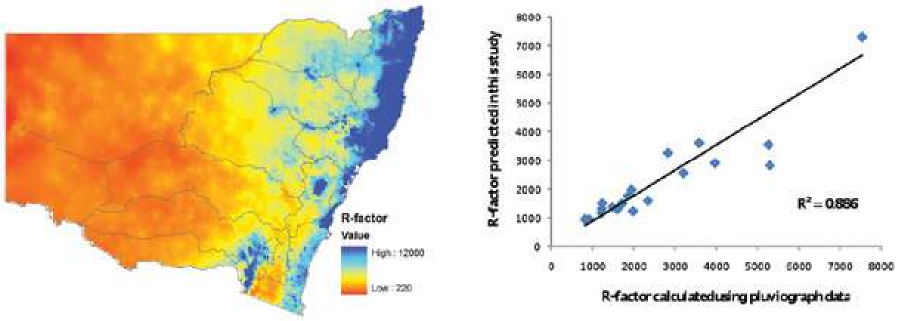


**Figure 1: Predicted annual mean R-factor (a) and its comparison with pluviograph records in NSW.**

The predicted R-factor (in unit MJ mm/ha.h.year), with a mean value about 1166, varies from less than 250 in western NSW to over 12,000 on parts of the North Coast. This means that erosion hazard varies about 50-fold across the State, while all other RUSLE factors being constant. The relative large spatial variations (13-fold) were noted in a previous study in NSW (Landcom 2004), but our estimates (on 30 m pixel basis) show a much higher variation than that reported in Landcom (2004). Statistcs analysis shows that about 90% NSW has an erosivity value less than 2250, and 50% area less than 820. However, the spatial variation in rainfall erosivity (both monthly and annual) is only about 7 times difference across NSW on catchment or CMA basis (Figure 2). Lower Murray Darling CMA has the lowest mean erosivity (500), while North Rivers CMA has the highest (3500, Figure 2). Our study on RUSLE erosion modelling so far (for period 2000-2011) reveals that the rainfall erosivity distribution has a significant impact on sheet erosion even when all other RUSLE factors (except P-factor) have been considered (Figure 3). The monthly variation of erosivity shows a similar pattern with that of the modelled sheet erosion in the same period (Figure 3a).

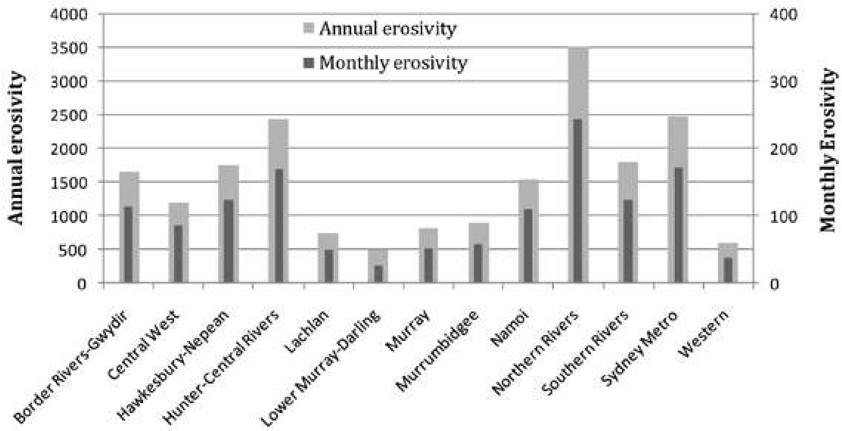


**Figure 2: Spatial variation of monthly and annual rainfall erosivity at NSW catchments (CMAs).**

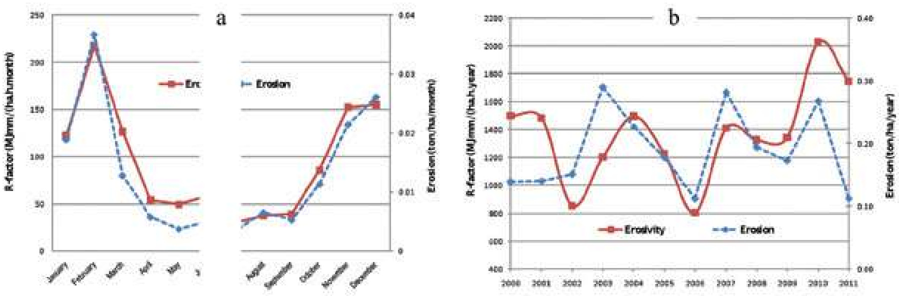


**Figure 3: Temporal variation of rainfall erosivity and sheet erosion (a: monthly, b: annually).**

## Conclusion

This study has demonstrated a suitable method for calculating monthly and annual rainfall erosivity values based on historic daily rainfall data for NSW. The methods have been successfully implemented in GIS for efficient calculation and mapping of the spatial and temporal variation of rainfall erosivity and, potentially sheet erosion across NSW. The spatial interpolation greatly enhanced the level of details about erosivity maps which can be further used for assessing erosion hazard and determining the timing of erosion control practices. With the automated GIS process developed in this study, the erosivity maps and erosion modelling can be readily upgraded when better rainfall data and models become available.

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# Influence of land use/land cover change on soil acidity and salinity in the lower Hunter Valley, NSW Australia

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### Abstract

In this paper, the environmental impacts of land use/ land cover change (LUCC) on selected soil properties (soil acidity and salinity) were explored for the lower Hunter Valley of New South Wales. As soil is the basis of all terrestrial ecosystems, soil degradation means lower fertility, reduced biodiversity and reduced human welfare. The monitoring of soil acidity and salinity as to how they are influenced by LUCC, is important, as these soil attributes are good indicators of soil quality in terms of soil productivity. Thus in this study two multivariate ordination techniques- correspondence analysis (CA) and canonical correspondence analysis (CCA) - were used to unravel the complex interrelationships of the landform attributes and LUCC derivatives with soil pH and EC. The results show that while elevation was the dominant influence on pH in vineyard production systems, natural vegetation in the form of woodland had greater influence on salinity in subsoil layers. The environmental implications of LULC change as evidenced by this study provides some insights for the future land use planning for the area.

### Keywords

Soil salinity, soil reaction, land use, terrain attributes, multivariate ordination.

### Introduction

Land use and land cover change (LUCC) has been recognized as a major driving force of global environmental perturbations. LUCC, through clearing of natural forests, subsistence agriculture, intensification of farmland production, modification of rangeland and expanding urban areas, has led to changes in the world's landscape in pervasive ways (Foley *et al.*, 2005; Chhabra *et al.* 2006). LUCC, and has led to unpleasant consequences such as global and regional climate change, distressed global biogeochemical cycles (e.g. carbon, nitrogen, and water cycles), and declining biodiversity and soil degradation. As soil is the basis of all terrestrial ecosystems, degraded soil means lower soil fertility, reduced biodiversity and consequent decline in human welfare (Braimoh and Vlek 2008). The focus of this paper is on the effects of LUCC on two important soil quality indicators, soil acidity and soil salinity. Specifically this paper aims to test the hypothesis that the influence of LUCC on soil acidity and salinity can be differentiated from the interactions of other landscape features controlling landscape processes influencing soil acidity/salinity. In other words the landscape features, combined with LUCC, can be used to robustly account for the spatial variation of soil acidity and salinity. To test this hypothesis we explored terrain attributes/LUCC/soil quality interactions by performing the multivariate ordination analyses correspondence analysis (CA) and canonical correspondence analysis (CCA).

### Materials and Methods

The historical LULC maps, derived from Landsat images for 1985, 1991, 1995, 2000 and 2005, were used to decipher the patterns of LUCC and explore the effect of LUCC on soil acidity/salinity. Soil samples were collected from 246 soil sample locations using the Latin hypercube sampling scheme (Odgers 2009). The samples from 0-10 cm depth were used to measure pH and electrical conductivity in a 1:5 soil: water suspension. Terrain attributes, such as DEM, slope, upslope area, and compound topographic index, which could influence the spatial patterns of soil properties such as acidity and salinity (Odeh *et al.* 1994), were derived from a digital elevation model.

Of the different multivariate ordination techniques, correspondence analysis (CA) and canonical correspondence analysis (CCA) were chosen for this study based on percentage variance accounted for in the species-environment relation. CA is indirect gradient analysis, while CCA is a restricted or canonical form of CA, in which the site scores are restricted to be linear combinations of measured environmental variables (Ter Braak *et al.* 1988; Odeh *et al.* 1991). In order to incorporate the effect of LUCC over the period between 1985 and 2005, three types of LULC information were derived from the series of LULC maps: i) vineyard consistency (Vine_C), ii) pasture/scrubland consistency (Past_C), and iii) woodland consistency (Wood_C). The list of LULC and other landform attributes and soil parameters used in CA and CCA, and the resulting biplots are shown in Table 1.

**Table 1: Description of LULC abbreviations**

| Abbreviation | Descriptions |
|---|---|
| Wood_C | Woodland consistency- woodland throughout the accounted period 1985 to 2005. |
| Past_C | Pasture/scrubland consistency- pasture/scrubland throughout 1985 to 2005. |
| Vine_C | Vineyard consistency- vineyard throughout 1985 to 2005. |
| F91vine | Area converted to vineyard between 1985 and 1991. |
| F95vine | Area converted to vineyard between 1991 and 1995. |
| F2000vine | Area converted to vineyard between 1995 and 2000. |
| F05vine | Area converted to vineyard between 2000 and 2005. |
| Aforest | Area converted to woodland, i.e. pasture/scrubland in earlier maps but woodland in the final map |
| Wood2 past | Area converted from woodland to pasture/scrubland, i.e. woodland in earlier maps but Pasture/scrubland in the final map |
| Others | Other than above, such as vineyard converting to pasture/scrubland, area etc. |
| pH10 | pH of soil from a depth of 0-10 cm |
| pH20 | pH of soil from a depth of 10-20 cm |
| pH30 | pH of soil from a depth of 20-30 cm |
| pH50 | pH of soil from a depth of 40-50 cm |
| EC10 | EC of soil from a depth of 0-10 cm |
| EC20 | EC of soil from a depth of 10-20 cm |
| EC30 | EC of soil from a depth of 20-30 cm |
| EC50 | EC of soil from a depth of 40-50 cm |

**Results and Discussion**

Multivariate ordination techniques such as CA and CCA are very robust in elucidating the complex interactions as they enable the inclusion of numerous target variables in the analysis and thus highlight their associations with the predictor variables. Useful outcomes of these multivariate techniques are biplots which are an extension of the ordination diagram. In order to understand the ordination diagram or biplot, each arrow represents a predictor variable, in a given direction or axis in the diagram (Figure 1); soil parameters (pH and EC) are projected onto this axis. The lengths of the vectors and angles between them are more important rather than their inter-point distances (Ter Braak 1995; Odeh *et al.* 1991; Greenacre 2007). Environmental variables with long arrows, closer to a given ordination axis, are more strongly correlated with the ordination axis than those with short arrows, and therefore more closely related to the pattern of landscape variation shown in the ordination diagram. Thus the ordination diagram shows the main pattern of variation of soil variables as accounted for by the predictor variables, in this case LULC categories and selected landform attributes.

As shown in Figure 1, both CA and CCA biplots convey similar overall information about the complex interrelationships, although the effect of the predictor variables is less prominent using CA in comparison to CCA. The biplot of CCA demonstrates that Vine_C (vineyard consistency) is more closely associated with slope; the arrows (predictor variables) pointing roughly in the same direction, indicating high positive correlation, while arrows pointing in the opposite directions imply high negative correlation and that arrows crossing at right angle indicate zero correlation (Qie *et al.* 2001). This is expected, because lands characterised by steep terrain are preferred for vineyards because of their better drainage and reduced risk of waterlogging (Taylor 2004). The CCA biplot shows no evidence of a close association of pH with Vine_C, rather elevation has close association with pH. Thus, the higher the elevation the larger is the pH level, especially in the topsoil layers. Wood_C has negative association with pH, indicating lower pH values for woodland compared to other LULC categories (pasture/scrubland and vineyard).

In considering the effect of LULC categories and landform attributes on salinity, EC values at different layers were found to spread out within the biplot indicating EC in various soil depths is associated with different LULC: Wood_C has negative association with topsoil EC (i.e. EC10) but somewhat closer association with EC50 and EC30. CTI and Past_C have somewhat close association with ECdeep which is logical in the sense that soluble salts were leached to the lower depths. Additionally, the larger values of CTI are usually found in the lower parts of watersheds and convergent hollow areas associated with soils with low hydraulic conductivity or areas of low slope (Minasny and McBratney 2006).

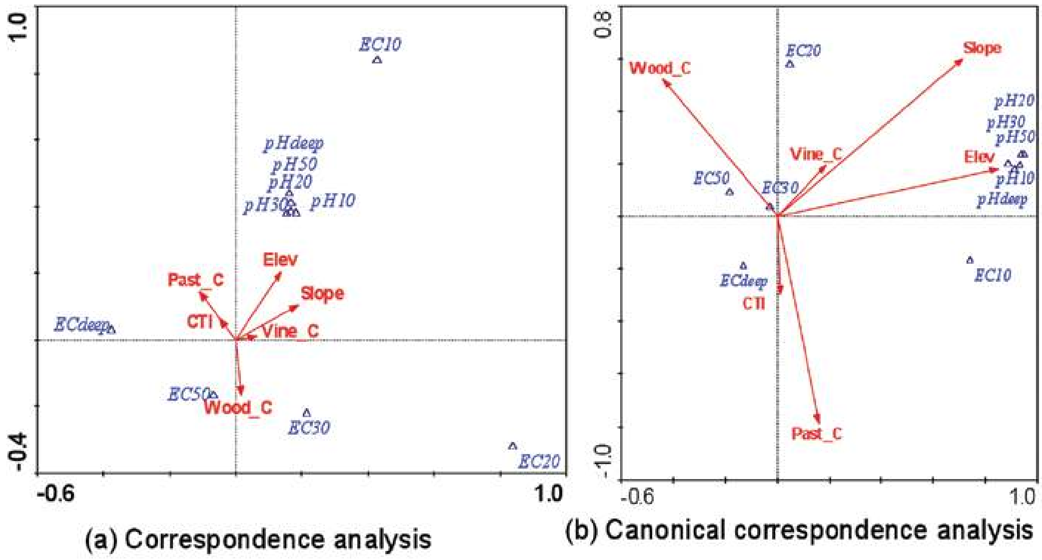


**Figure 1. Biplots: (a) Correspondence analysis; (b) Canonical correspondence analysis.**

*Note that: Wood_C= woodland consistency, Past_C= pasture/scrubland consistency, Vine_C: vineyard consistency, Elev: elevation, CTI: Compound topographic index. pH10: pH at 0-10cm, pH20: pH at 10-20cm, pH30: pH at 20-30cm, pH50: pH at 40-50cm and pHdeep: pH at a depth >50 cm; EC10: EC at 0-10cm, EC20: EC at 10-20cm, EC30: EC at 20-30cm, EC50: EC at 40-50cm and pHdeep: EC at a depth >50 cm.*

Table 2 shows the eigenvalues and variance accounted for by the first four axes of CA and CCA. The variance accounted for by the first two axes of CA (i.e. represented by the biplots) is 75.5% for CA, rising to 82.2% in case of CCA. The soil-environment correlation also improved slightly from 0.514 for CA to 0.545 for CCA.

**Table 2: Variance accounted for by the first four axes of CA and CCA. [Sum total eigenvalue for CA is 0.4 and for CCA is 0.021]**

| Axis | Eigenvalue (λ) | % of total λ | R (Soil, environment) | Cumulative % variance (Soil, environment) |
|---|---|---|---|---|
| | | Correspondence analysis (CA) | | |
| 1 | 0.161 | 40.3 | 0.212 | 33.5 |
| 2 | 0.099 | 24.8 | 0.302 | 75.5 |
| 3 | 0.083 | 20.8 | 0.184 | 88.5 |
| 4 | 0.044 | 11 | 0.168 | 94.3 |
| | | Canonical correspondence analysis (CCA) | | |
| 1 | 0.011 | 52.4 | 0.313 | 49.5 |
| 2 | 0.007 | 33.3 | 0.232 | 82.2 |
| 3 | 0.003 | 14.3 | 0.187 | 95.5 |
| 4 | 0.001 | 4.8 | 0.146 | 99.8 |

Thus ordination analysis through the biplot can inform us as to whether important environmental variables have been overlooked: an important variable has definitely been missed if there is no relation between the mutual positions of the sites in the ordination biplots and the measured environmental variables (Ter Braak 1995). In this study, the biplot of CCA is indicative of higher pH in the vineyard areas, probably due to cultivation of vineyards in the upper slopes which has close association with higher pH (Figure 1B). Vineyard cultivation is preferred in areas of slightly steeper slopes and a higher elevation rather than the lower part of watershed in order to have katabatic drainage of air, in addition to the requirement of good water drainage and reduced risk of waterlogging (Taylor 2004). Therefore, the higher pH of vineyard soils may have resulted from the cultivation of vineyards in mid- to upper slopes as well as from any lime application under vineyard cultivation.

## Conclusion

The effect of LULC on soil acidity and salinity indicates that the soils under vineyard have significantly higher pH than those under different LULC categories, which may be attributable to land use practices under vineyard, and/or due to inherent soil properties or landform that attracted vineyard cultivation. However, there is no significant effect of LULC on soil salinity indicating that vineyard land use has not led to increased soil salinity over the period. Elevation was found to have the dominant influence on pH in vineyards, while woodland had the closer association with deeper soil salinity as measured by EC50 and EC30 but negative association with topsoil EC and pH. Thus ordination analysis, in has enabled our understanding of the complex but minute variation in pH and EC in relation to change in land use/land cove and the causal effect of other landform attributes.

## Acknowledgements

We acknowledge the contribution of soil data by Dr Nathan Odgers of the University of Sydney.

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# Gully erosion mapping for risk assessment in Sydney's drinking water catchments

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**Abstract**

Erosion gullies are a source of sediment and other pollutants for Sydney's water supply however there is a lack of information to allow optimum allocation of limited gully remediation funds. The gully erosion trial was initiated to develop a gully erosion pollutant export model and to establish the methodology for mapping and classifying erosion gullies within the Sydney Catchment Authority (SCA) area of operations. The trial has been conducted in 2 stages with stage 1 being a mapping exercise and stage 2 an analysis of different LiDAR scenes before and after a significant rainfall event. This paper relates to stage 1.

The mapping was carried out with a Planar stereo mirror using ADS40 20cm air photography captured in May 2011. Gullies were split into segments based on small catchment areas. Gully treatment and severity was determined using the stereo mirror in ArcGIS. The presence or absence of bare soil was used to determine the risk hazard to the gully floor, walls and heads.

A prioritisation tool was developed to prioritise individual gullies and drainage units for treatment. Untreated gullies with bare soil at the head and a catchment greater than 30ha were considered highest risk. Site assessment in May 2012 showed that gullies mapped as high risk had all advanced due to a significant rainfall event in March 2012. The tool will enable gullies posing the highest risk to water quality to be treated first, leading to a decrease in ongoing land degradation and an improvement in water quality in the Sydney drinking water catchments.

## Introduction

Gullies are recognised as a major sediment source to rivers (Rustomji 2006). Fine sediment (clay and fine silt: <10um) carries a large proportion of nutrients and adsorbed pollutants and is readily transported (Caitcheon *et al.* 2006).

The Sydney Catchment Authority (SCA) funds gully remedial works through the Catchment Protection Scheme (CPS). The CPS aims to maintain and improve the quality of waters yielded from the Sydney's drinking water catchments by implementing erosion gully remedial works and through community education and training. The CPS operates as a partnership program that includes SCA, Hawkesbury Nepean Catchment Management Authority (HNCMA) and Southern Rivers Catchment Management Authority (SRCMA).

Previously remedial works have focused on priority subcatchments derived from gully erosion mapping undertaken by the Soil Conservation Service in 1984. It was considered by the SCA that this dataset no longer represented the extent and severity of gully erosion in the drinking water catchments due to many gullies being remediated or naturally stabilising and new evidence suggesting that contemporary rates of erosion were minimal.

The Gully Erosion Evaluation was initiated to establish a method for mapping and classifying erosion gullies in Sydney's drinking water catchments and to develop a gully erosion pollutant export model.

## Materials and Methods

### Study Area

Three drainage units in the SCA area of operations were selected as trial sites for the project (Fig. 1). Dixons Creek was chosen due to a significant rainfall event in December 2010 that caused observable erosion. Eden Forest was chosen due to the long history of soil conservation treatments. Oallen Ford was chosen as representative of severe erosion in the SRCMA area. Dixons Creek and Eden Forest are within the HNCMA area.

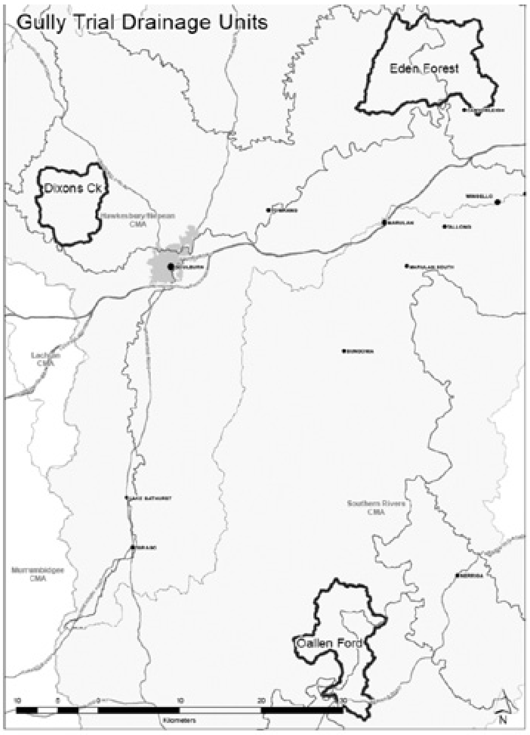


Figure 1: Gully Trial Drainage Units

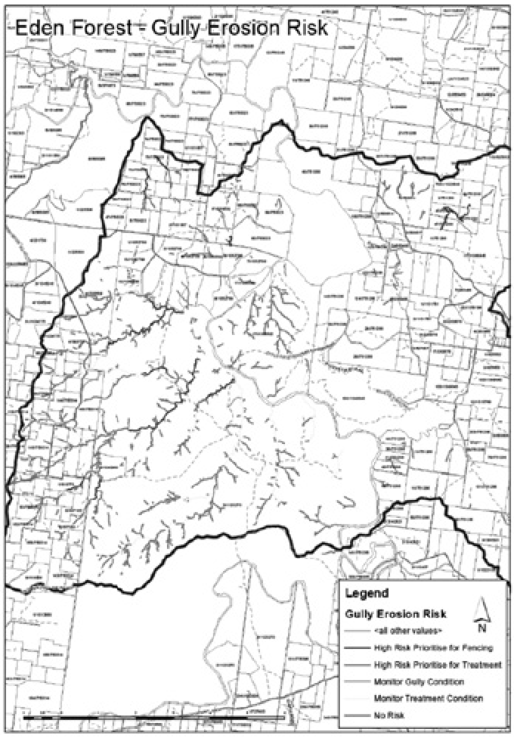


Figure 2: Gully Erosion prioritisation for Eden Forest Drainage Unit

**Air Photo Interpretation (API)**

20cm ADS40 Aerial photography was obtained from NSW Land and Property Information (LPI) in May 2011 for parts of the three trial areas.

Gullies were mapped with ADS40 imagery captured in 2008 at a 50cm resolution and also with ADS40 imagery captured in May 2011 at 20cm resolution. The ADS40 imagery was viewed in 3D using a digital stereoscope (Planar) unit with associated ArcMap software.

Gully centrelines (line feature) and gully heads (point feature) were digitised at approximately 1:1000 scale. Gullies were split into segments based on catchment areas derived using Arc Hydro Geoprocessing tools (ESRI, 2009), which were also used to determine the cumulative catchment area for all gully segments. Gully treatment and severity was determined using the stereo mirror in ArcGIS. Gullies were attributed with the type of treatment which included sediment trap below, engineering structure above (flume, dam, bank) and or fencing, as well as the stability of the treatment. The presence or absence of bare soil was used to determine the risk hazard to the gully floor, walls and heads. Gully depth was also recorded. Point features were also created for major engineering structures and also for all failed structures.

**Prioritisation**

The Gully Prioritisation Tool (Fig. 3) used to describe gullies in the trial areas was based on the Gully Erosion Ratings used as part of the NSW Gully Monitoring Evaluation & Reporting (MER) procedure.

Field visits were undertaken to verify gully classification and a differential GPS was used to record the current extent of selected gullies for comparison with aerial photography.

**Results and Discussion**

**Imagery Resolution and 3d interpretation**

Increased imagery resolution and stereoscopic viewing allowed mapping of gullies and gully treatments of much greater accuracy than low resolution mapping using non stereoscopic interpretation. Gullies with tree cover and secondary gullies were identified more consistently with 20cm imagery compared to 50cm imagery. Field verification using 20cm imagery was greatly reduced due to greater confidence in attribution. Also field verification showed greater correlation with mapping based on 20cm imagery compared to mapping based on 50cm imagery.

**Comparison of 1984 mapping.**

The 1984 mapping was done using much lower resolution imagery and when viewed over current high resolution aerial photography the line work does not align with actual gully locations. In most cases it is obvious which gullies are represented by each line. Only 58% of gullies mapped were mapped in 1984. Many gullies that were mapped under tree cover had not been identified in 1984.

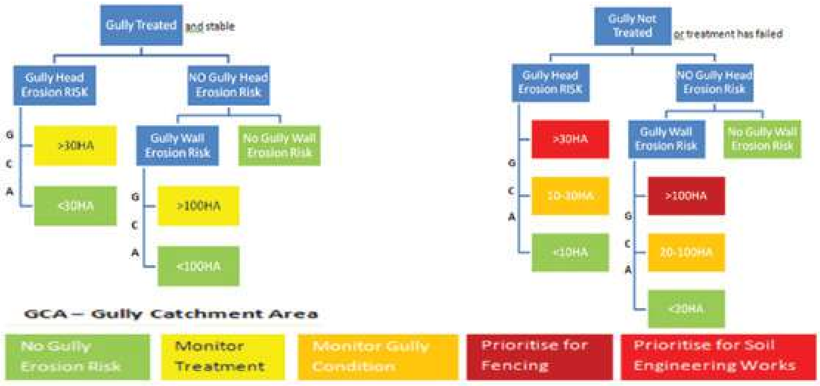


**Figure 3: Gully Prioritisation**

**Prioritisation**

The prioritisation rating was modified to be risk based rather than assigning an erosion status. Based on the presence or absence of bare soil, alone it is much more valid to attribute a gully as being at risk of erosion than to be classified as actively eroding. The Gully Prioritisation ratings also consider catchment size and the presence and integrity of gully treatments. Many of the soils in the trial sites are characterised by shallow topsoils and highly dispersive subsoils. The soils are of Ordovician metasediment or Granitic origin. Soil analysis could further differentiate erosion risk using exchangeable cations or dispersion percentage.

The tool will allow gullies outside the trial areas to be evaluated. The prioritisation initially looks at the treatment status of a gully and highlights gullies whose treatments should be monitored for failure. If a gully is untreated the gully head risk is looked at first then the gully sidewalls as high risk gully heads will require potentially much greater financial investment in remediation with engineering structures usually required. Generally sidewall erosion is treated by excluding stock access through fencing to allow natural rehabilitation. The catchment area cutoffs were derived from analysis of gullies within the three trial drainage units and should be reviewed of more drainage units are mapped.

Analysis of all gullies show that Oallen Ford has the most number and metres of high risk gully heads while Eden Forest and Dixons Creek had more high risk gully wall sections (Table 1). Eden Forest and Dixons Creek drainage units have more high risk side wall gullies due to large gully sections with greater catchment areas while Oallen Ford has shorter, smaller catchments that discharge directly into the Shoalhaven River. Figure 2 shows the location of high risk gullies in Eden Forest drainage unit.

**Are gullies active?**

There is much evidence that selected gullies are active within the 3 trial drainage units. This evidence includes changes in gully extent based on measurement of gully extent with a differential GPS of specific gullies compared to imagery. Other evidence from API includes sediment fans at gully bases and sediment deposits in the larger streams which change over time. This is also supported by field visits where undercutting of walls and slumping of walls was evident in many gullies and turbid water was observed discharging from gullies.

From comparison of GPS data and aerial photography at specific locations it is evident that many gullies have increased since May 2011. One gully in Dixons Ck drainage unit, which was initiated in December 2010 grew by over 9 metres as a result of a significant rainfall event in March 2012. Much of the literature suggests that the majority of change in gully extent occurred soon after gully initiation shortly after European settlement (Armstrong 2002); Rustomji *et al.* 2006). The literature also suggests that much of the sediment coming from gullies is sourced from the gully itself (Armstrong 2002). With aerial photograph interpretation a change in gully extent can be detected, however change in gully depth could not be determined. Because the surface area of gully floors and walls is considerable, a large volume of sediment can be derived from eroding just a few millimetres from the walls and floors of gullies but cannot be detected via imagery analysis. The second component of the gully trial will attempt to quantify erosion rates through analysis of LiDAR data over time. When LiDAR is reacquired the rate of change will be able to be determined for specific gullies.

**Table 1: Gully Erosion Prioritisation classes for trial areas. Metres of Gullies for different risk classes with percentages within drainage units and within risk classes. Eg 48% of 'No Risk' gullies occur in Eden Forest while 71% of Eden Forest gullies are 'No Risk'.**

| | Dixons Creek | | Eden Forest | | Oallen Ford | | Total (m) |
|---|---|---|---|---|---|---|---|
| | (m) | % | (m) | % | (m) | % | |
| | 88,643 | 37 | 115,281 | 48 | 34,896 | 15 | 238,820 |
| No Risk | 65% | | 71% | | 52% | | 65% |
| | 22,439 | 43 | 22,269 | 43 | 7,489 | 14 | 52,197 |
| Monitor Treatments | 16% | | 14% | | 11% | | 14% |
| | 6,137 | 23 | 8,161 | 30 | 12,555 | 47 | 26,853 |
| Monitor Gully Condition | 5% | | 5% | | 19% | | 7% |
| | 1,738 | 20 | 1,777 | 20 | 5,295 | 60 | 8,810 |
| Prioritise for Treatment | 1% | | 1% | | 8% | | 2% |
| | 16,941 | 44 | 14,981 | 39 | 6,671 | 17 | 38,593 |
| Prioritise for fencing | 12% | | 9% | | 10% | | 11% |
| | 245 | 30 | 554 | 67 | 24 | 3 | 823 |
| Other | 0% | | 0% | | 0% | | 0% |
| Total | 136,142 | 37 | 163,023 | 45 | 66,930 | 18 | 366,095 |
| Drainage Unit Area (ha) | 6,483 | | 14,118 | | 9,334 | | |
| Treatment Score (m/ha) | 0.27 | | 0.13 | | 0.57 | | |
| Fencing Score (m/ha) | 2.61 | | 1.06 | | 0.71 | | |

## Conclusion

There is clear evidence that significant rainfall events such as experienced in the Dixons Creek area in December 2010 will cause significant gully head and sidewall erosion. The mapping and prioritisation of gullies within 3 drainage units allowed prioritisation of individual gullies and drainage units. Oallen Ford had the highest metres per ha of gullies requiring treatment, while Dixons Ck had the highest metres per ha of gullies requiring fencing. The tool will enable gullies posing the highest risk to water quality to be treated first, leading to a decrease in ongoing land degradation and an improvement in water quality in the Sydney drinking water catchments.

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# Controls on land salinisation in the Baldry Catchment, central west NSW: Influences of soil and regolith on fluid flow pathways

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## Abstract

Large (>1ha) saline scalds with bare soils and efflorescing sodium chloride salts extend across the footslopes, with smaller localised mid-slope sites, in the Baldry landscape. Baldry is a sub-catchment of the Little River system, in central west NSW, described as a salinity 'hotspot' in the Murray Darling Basin. This paper details the biophysical properties of the soil and regolith from a catena at Baldry and explores how interaction with these materials influences the observed biophysical and chemical properties of the water. A summary of the hydrological patterns observed in surface and near surface waters across the catena at Baldry from 2004 to 2008 underpins this interpretation (Acworth *et al.* 2009). The distribution and configuration of soil and regolith materials developed on the Kynuna granite at Baldry: controls recharge to the confined aquifer system; dictates the dominant fluid flow pathways; and, explains the spatial location of discharge sites. Knowledge of soil and regolith influences on water chemistry at this site constrains the potential sources for salt, and explains how sodium chloride salts were introduced to the Baldry landscape.

## Introduction

Baldry is a sub-catchment (111,134ha, 43%) of the Little River catchment, in central west NSW. The Baldry study site is approximately 50ha in area and is located near the catchment divide, upstream (south) of Dubbo (Fig. 1). The study site is characterised by an open low hill and rise landscape with a northward running creek and gently inclined slopes (typically <10% gradient) developed on moderately weathered to locally deeply weathered early Devonian pink hornblende-biotite granite rocks of the Kynuna granite (Yeoval suite). Exposed tors and outcrop are a common feature on hill crests and ridges, with subcrop and low rounded tors on midslopes. Lower colluvial slopes comprise clays, coarse sands and minor gravels. Alluvial sediments in valleys are quartzose gravels with a sand and clay matrix. Soils are typically Rudosols on the upper and mid-slopes with Kandisols and Chromosols on lower slopes and in depressions. Soils in lower parts of the landscape typically have moderate cation exchange capacity. Salt expression manifests as large scalds (>1ha) on the lower slopes and footslopes, with localised small scalds in the midslope, and surface flow through these areas gives rise to localised stream water quality issues. Recharge to the groundwater system is controlled by the configuration of permeable and less permeable soils formed on in-situ weathered and colluvial materials and through fractures in outcropping and subcropping rock. Although the creek receives a component of baseflow low in the valley, it is seasonally ephemeral. This landscape is responsive to rainfall events and hydrogeological landscape (HGL) characterisation in this area (Wilford *et al.* 2009) suggests that it is responsive to land use changes

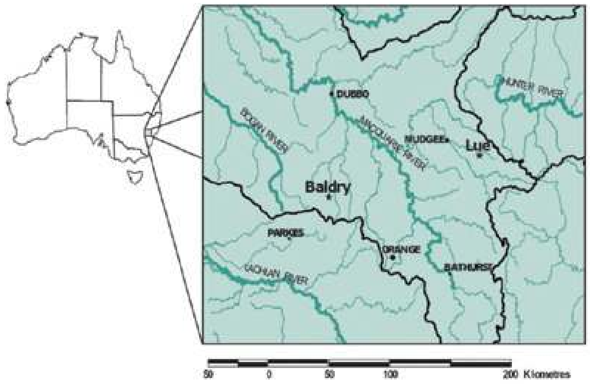


**Figure 1. Location of Baldry site, central NSW.**

**Figure 2. Location of piezometers and bore holes at Baldry. Water data from holes BH10, BH8, BH5, BH6 and BH2 used for soils/regolith comparison (Acworth et al. 2009).**

## Methods

Soils were described from a (SSW-NNE) catena sub-parallel to the streamline at Baldry (Isbell 1996; McDonald *et al.* 1989). Groundwater was characterised from nearby holes in the piezometer and bore array, including four piezometer/bore pairs at: Site 2 (footslope), Site 6 (lower slope), Site 8 (upper midslope) and Site 10 (upper slope); and, 1 hole at Site 5 (lower midslope) (Fig. 2). X-ray diffraction (XRD) analysis, for mineralogical characterisation of soil/regolith samples, was conducted (as per Moore and Reynolds 1989). Field (pH, EC, alkalinity) and hydrochemical analyses (major and minor ions; stable isotopes) for water are compiled from previous work with colleagues in this area (Acworth *et al.* 2009).

## Results

### *Soil and regolith development at Baldry*

Near the crest on the upper slopes weathered granite saprolite (Figure 3a) is overlain by sandy soils (Leptic Rudosol) with limited pedogenic modification of the profile and a simple, massive, two layer soil preserved (Figure 3b). In locations characterised by granite outcrop and subcrop, weathered granite corestones are exposed at the land surface as low tors, and eroded to form an apron of angular quartzose and lithic fragment clasts of granule to cobble size (Figure 3c). Locally, subcrop occurs in midslope settings (Site 5), with deeper soils preserved upslope and downslope of the saprock highs. A 2003 electromagnetic induction survey (EM31) measured elevated electrical conductivity associated with the creek valley, and some localised 'high' signals that correspond to areas upstream of subcrop (e.g. Site 5). This is consistent with localised ponding of water in the shallow subsurface behind impediments to natural flow in this landscape. Elsewhere the midslope (Site 8 - Yellow Kandisol) and lower slope (Site 6 - Yellow and Brown Chromosol) profiles illustrate more pedogenically developed gradational to texture contrast soil profiles with clay loam and sandy clay to clay B horizons respectively (Figure 3d). A 2cm thick low permeability, clay layer is described at approximately 90cm depth in the upper midslope, midslope and lower slope profiles, broadly coincident with the base of the B horizon, that locally inhibits vertical transfer and may cause ponding in the shallow subsurface during wetter periods. In addition to internal permeability variation in the soil zone, theta probes in upper midslope to lower slope settings indicate limited vertical transfer between the water table and the deeper aquifer system in this part of the landscape. The implication is that the mid-section of the catena, from the mid-upper slope to the break in slope at the bottom of the valley, is segregated into an upper relatively unconfined aquifer that can receive recharge across the landscape, but has internal texture contrast related permeability variation, and a deeper confined system with an upper boundary in the lower saprolite.

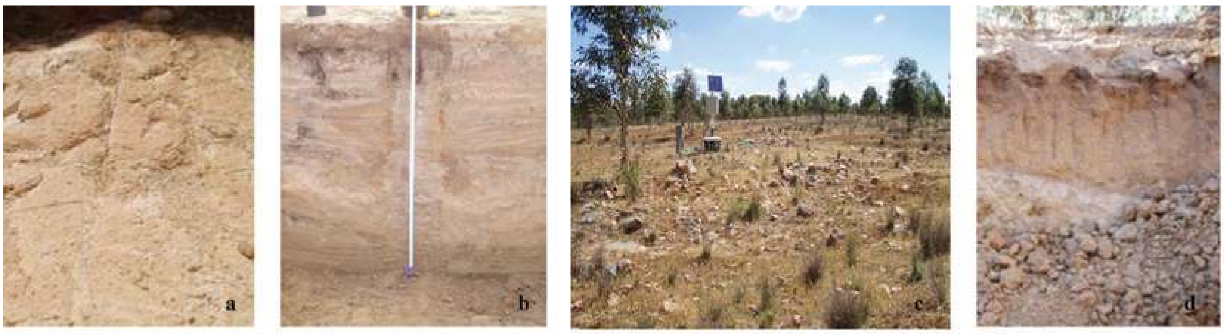


**Figure 3a: Saprolite on weathered Kynuna granite at Baldry**

**Figure 3b: Leptic Rudosol from an upper slope setting (near Site 10) at Baldry**

**Figure 3c: Weathered granite subcrop in a midslope setting (Site 5) at Baldry**

**Figure 3d: Yellow Chromosol in a lower slope setting (near Site 6) at Baldry**

On the footslope there is a colluvial accumulation of angular quartzose and lithic granular clastic sediment with associated clay loam soils overlying and interdigitated with, pallid and variably saline sandy clay and clay soils in the creek. Historically the salinity flux in the lower creek has shown a notable drop when the water levels in the deep bores dropped below stream level, indicating baseflow interconnectivity.

### *Secondary mineralogy in the soil/regolith*

For all profiles the composition of the uppermost materials (0-20cm) reflects a primary (rock) mineral composition of quartz, Na-plagioclase feldspar (albite) and K-feldspar (orthoclase) with little to no preserved clay fraction. This skeletal, lithic fragment and primary-mineral-bearing granular sand at the land surface is formed when clays move down profile during pedogenesis, and may also be an artefact of historical cropping in this area. In crestal settings primary minerals are present because there is rock in the shallow subsurface. Weathering of reactive mafic minerals (e.g. hornblende) through smectite to kaolinite clays and Fe/Mg sesquioxides, in parallel with incipient weathering of the plagioclase feldspars and K-feldspars to smectite clays and some kaolinite clay, and limited formation of illite clays after layer silicates (micas), results in a secondary mineralogy dominated by quartz, albite and orthoclase with a mixed smectite and kaolinite with minor illite, clay assemblage. In midslope to footslope areas where weathering profiles are deeper a similar pattern is observed but with higher kaolinite to smectite ratios as weathering of the feldspars and remaining mafic minerals proceeds through intermediate phases to form kaolinite clay. Smectite clays have a higher cation exchange capacity than kaolinite clays, so soils with this mixed clay assemblage can chemically interact with, and potentially buffer, saline solutions as they pass through the soil zone. Soils lower in the valley are increasingly dominated by quartz and kaolinite clay which are both relatively inert chemically, so there is reduced capacity for chemical buffering of salts in the discharge zone. The rock is high in Na and the weathering products preserve a sodic signature. Soils low in the landscape are dispersive and susceptible to erosion, so there is a combined dryland salinity and erosive sodic soil hazard in the footslope area.

*Water in the soil/regolith*

Values for $EC_w$ (shallow) at 25°C range from: 15.8 mS/cm for water in a Leptic Rudisol on the upper slopes; 9.7 to 13.5 mS/cm for water in a Yellow Kandisol on the upper mid-slope; 5.4 to 5.8 mS/cm for water in a Yellow Chromosol on the mid to lower slopes; and, 5.8 to 6.8 mS/cm for water on a Brown Chromosol on the footslope near the creek. For the soil (unconfined aquifer) zone the $EC_w$ values are higher near the top of the hill, implying there is a store of salt high in the landscape. For deeper holes (piezometers and bores) there is a broad relationship between thickness of the regolith and groundwater salinity with lower $EC_w$ values associated with thinner profiles (weathered saprolite to 2.5 m) near rocky outcrop, on the upper slope and on the lower slope, and higher $EC_w$ values associated with thicker profiles (weathered saprolite to 5m) on the midslope and footslope. The depth to weathered bedrock is highly variable, and three-dimensional subsurface regolith geometries complex, because of joint associated corestone formation on the granite. The irregular soil-saprolite-rock configuration influences where salt can be stored in the landscape and how groundwater moves vertically and laterally.

*Influences of soil/regolith ion water chemistry*

In general deeper waters are Na - Mg (Mn) - Ca (Sr) - K bicarbonate fluids with a stronger NaCl signal high in the landscape. The cation-rich bicarbonate chemistry reflects the influence of rock weathering on waters passing through the confined aquifer, and the pattern of elevated NaCl values indicates a salt store near the hill crest Evidence of mixing illustrates connectivity between the shallow and deep systems in crestal areas. There is probably also direct recharge to deep groundwater in areas where weathered corestones (fractured rock) are present near the land surface (Site 5). Shallow groundwater is uniformly very slightly acid to neutral (pH 6.5-6.9), which is less acidic than might be expected on a granitic substrate, and may reflect the presence of dissolved salts. Waters are typically oxidising (DO: 0.04 mg/L to 4.89 mg/L; variable Eh -88mV to 197mV, very low $Fe^{2+}$ and no detectable $S^{2-}$) and there is no apparent manifestation of acid sulphate soils at this locality. For most of the catena shallow waters are NaCl dominant with more Mg (Ca) in rocky areas, but Na bicarbonate waters are dominant in footslope soils. A crestal and upper slope NaCl salt store strongly influences the chemistry of the shallow unconfined system with NaCl transported downslope in near surface water. Saline (NaCl) waters discharge in lower slope to footslope (break in slope) settings causing land salinisation. Below the discharge zone the presence of Na-bicarbonate waters reflects the interconnectivity with the Na-bicarbonate water from the deep aquifer (Site 2). Deep bores in the lowermost parts of the landscape have water levels above ground level, but hydraulic head in this low relief subcatchment is not high. The implication is that there may be a more regional influence on the deep aquifer system in the Baldry area.

*Stable isotopes in groundwater*

Groundwater isotope values indicate that it has a meteoric (rainfall recharge) origin (Acworth *et al.* 2009) and that recharge typically takes place in cooler periods (winter dominant rainfall). Values show no relationship between groundwater salinity and isotope enrichment illustrating that the salinisation of groundwater cannot be attributed to evaporative concentration across the landscape. It is clear from the field expression of land salinisation (puffy soils, NaCl salt efflorescence) that there is some localised shallow evaporative concentration of salts associated with the discharge zone, but isotope values indicate that evaporative concentration is not the principal mechanism to explain the presence of salt in this landscape.

## Discussion

Interpretation of hydrologic and shallow water biophysical and chemical patterns in the piezometer and bore array at Baldry, indicate that there is a local groundwater flow system (GFS), but there may be a more regional influence on the confined aquifer system, indicated by bore water levels above land surface, and this warrants further investigation. Recharge to the confined aquifer occurs wherever weathered granite corestones (fractured rock) are present at or just below the land surface, typically in crestal locations, but commonly as subcrop in midslope settings. Gradational (Yellow Kandisol) to texture contrast (Yellow-Brown Chromosol) soils in the mid-section of the catena, host an upper relatively unconfined aquifer that has it's lower boundary on in the lower saprolite, overlying a deeper confined system. The irregular three-dimensional configuration of weathered granite saprolite and more indurated corestones in the subsurface introduces a complexity to fluid flow pathways at depth. Both aquifers have mixed chloride and bicarbonate waters but generally there is a NaCl signature in the shallow system and a more complex Na bicarbonate signature in the deeper system, and on the valley floor. Evidence for mixing occurs in crestal and footslope areas where there is greater connectivity between the aquifer systems. Higher NaCl concentrations in crestal and upper slope settings indicate a salt store high in the landscape. In addition to inland deposition of oceanic cyclic salt from the east across the entire landscape, there has been long term deposition of NaCl coupled with aeolian dust from the west, derived from ablation of soils and salt lakes in inland NSW. Preferential deposition of dust in lenses at hill crests would explain elevated NaCl concentrations in shallow groundwater in crestal and upper slope settings at Baldry. Mineralogical and chemical analyses indicate that the rock and saprolite in this landscape are still actively weathering and contributing Na, Mg with some Ca (Sr) and K ions to bicarbonate waters. The crestal NaCl signature overprints this local chemistry. The availability of Na for incorporation into secondary mineral phases during weathering explains the presence of sodic soils that are susceptible to erosion in lower slopes and footslopes at Baldry. Large (>1ha) saline scalds extend across the footslopes, with smaller localised mid-slope sites, due to discharge of saline (NaCl) water from the unconfined shallow aquifer system, at the break in slope and in areas of ponding behind granite outcrop and subcrop. In the same landscape there is sufficient hydraulic head for Na bicarbonate water to emerge in footslope settings.

## Conclusion

Landscapes with superimposed: NaCl land salinisation and associated runoff, soil sodicity and erosion; emergence of groundwater in lower parts of the landscape; and, linked impacts on vegetation cover and potential land-use, present a complex natural resource management (NRM) challenge. Understanding the spatial configuration of natural materials (soils, regolith and rock), where in the landscape salts are likely to store, the species of salt present, how salts are generated and mobilised; and, how water moves through and over the landscape, enables more strategic land and water management actions to be implemented at Baldry and in similar landscapes.

## Acknowledgements

The authors acknowledge support from colleagues at the University of NSW, in particular Ian Acworth and Wendy Timms, who worked with Tony in the initial phases of this study at Baldry (Acworth *et al.* 2009).

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# The soil specific nature of threshold electrolyte concentration analysis

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## Abstract

Maintenance of soil permeability is paramount to irrigation, especially as the use of saline-sodic waters increases. While research has shown that soil permeability can be maintained under high sodicity conditions, provided the electrolyte concentration is sufficiently high, there is relatively little information depicting threshold electrolyte concentration (TEC) relationships. Furthermore, even though TEC curves have been shown to be soil specific and dependent on soil properties such as clay mineralogy, clay content and organic matter content, guidelines for irrigation management in Australia do not currently acknowledge this. The work reported in this paper provides examples of TEC relationships for a range of soils from southern Queensland. Through correlation analysis, the work also investigates the role of clay content, mineralogy and organic matter in determining these relationships. Calculation of TEC curves for 6 south-eastern Queensland soils illustrated that TEC relationships are soil specific, even within soil orders; contrary to current guidelines. Additionally, correlation analysis revealed that there were no apparent relationships between the critical EC and SAR values (those determining TEC functions) and soil properties such as clay mineralogy, clay content and organic matter content.

## Introduction

Saline and sodic waters are increasingly being used for irrigation purposes. In particular, the rapid development of the coal seam gas (CSG) industry throughout eastern Australia has raised interest in the productive use of saline-sodic water. A key determinant to the sustainable use of saline and sodic water for irrigation is the maintenance of soil permeability. The application of sodic water without appropriate management has been shown to increase reactive clay swelling and dispersion (McNeal *et al.* 1968), change pore size distribution (Jayawardane and Beattie 1978) and decrease the saturated hydraulic conductivity ($K_{sat}$) of soils (McNeal and Coleman 1966; Quirk and Schofield 1955). However, Quirk and Schofield (1955) demonstrated that soil permeability can be maintained even under conditions of high sodicity (sodium adsorption ratio – SAR) provided the electrolyte concentration (EC) of the soil solution is greater than a critical value, known as the Threshold Electrolyte Concentration (TEC).

Quirk and Schofield (1955) suggested that the TEC could be identified on the basis of a 10% reduction in $K_{sat}$ from the stable condition. However, McNeal and Coleman (1966) subsequently proposed using a 25% $K_{sat}$ reduction and Cook *et al*. (2006) suggested a 20% $K_{sat}$ reduction as the TEC value. Importantly, the TEC varies with soil type (Quirk 2001; Rengasamy and Olsson 1991), with the key soil properties known to affect the permeability being clay content (Frenkel *et al.* 1977; Goldberg *et al.* 1991; McNeal *et al.* 1966), mineralogy (Churchman *et al.* 1995) and organic matter type and content (Nelson and Oades 1998). Despite this, there are very few examples of TEC relationships found in the published literature and the ANZECC (2000) guidelines for water quality are commonly used as a guide to the appropriate selection of saline-sodic water to maintain soil permeability. However, these TEC curves (Figure 4.2.2 in ANZECC 2000) were developed from a single study (DNR 1997) conducted on only two soils and cannot be considered representative of the range of soils encountered throughout Australia. The aim of the work reported in this paper is to provide examples of TEC relationships for a range of soils from southern Queensland and to investigate the role of clay content, mineralogy and organic matter in determining these relationships.

## Method

This study is reported in two parts. The first set of data provides an example of the TEC analysis for six soils only and the second set uses aggregated TEC data to investigate the relationships between TEC and selected soil properties for 36 soils from south-east Queensland.

Six soil samples were taken from Roma and the Darling Downs (Table 1). These were air-dried before being crushed to pass through a 2.36 mm sieve. Soil chemical measurements (exchangeable cations, exchangeable sodium percentage – ESP, cation exchange capacity – CEC) and organic matter content

(OMC) were calculated using standard procedures outlined in Rayment and Higginson (1992). The method for determining clay cation ratio (CCR), an indicator of clay mineralogy, was consistent with Shaw and Thorburn (1985). Clay content was obtained using particle size analysis consistent with (Gregorich *et al.* 1988).

**Table 1. Selected soil properties of six south-east Queensland soils used for the TEC comparison**

| Soil | Soil order | Texture | Clay content (%) | OMC (%) | ESP (%) | EC1:5 (dS/m) | Ca:Mg | CEC (meq/100 g) | CCR (meq/g) |
|---|---|---|---|---|---|---|---|---|---|
| 1 | Grey Vertosol | Medium clay | 44.3 | 1.7 | 3.7 | 0.05 | 1.14 | 21.40 | 2.07 |
| 2 | Black Vertosol | Heavy clay | 56.9 | 1.7 | 2.7 | 0.06 | 2.24 | 31.80 | 1.79 |
| 3 | Red Chromosol | Sandy loam | 12.7 | 0.8 | 1.2 | 0.04 | 3.08 | 5.13 | 2.48 |
| 4 | Brown Chromosol | Silty loam | 5.3 | 0.6 | 1.2 | 0.12 | 10.46 | 3.36 | 0.16 |
| 5 | Brown Chromosol | Sandy loam | 12.6 | 0.9 | 0.6 | 0.02 | 5.07 | 6.25 | 2.02 |
| 6 | Black Vertosol | Medium clay | 44.3 | 1.9 | 4.5 | 0.21 | 2.93 | 39.80 | 1.11 |

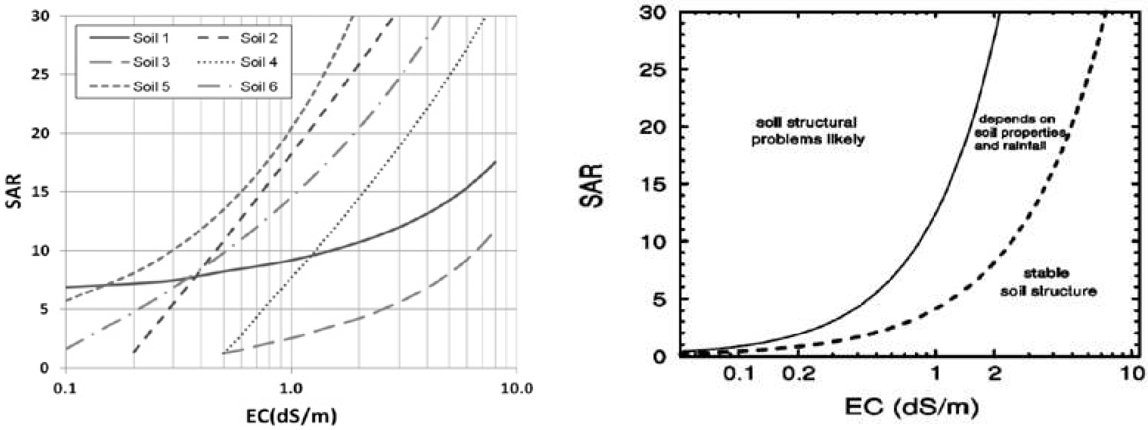
Five short soil columns (internal diameter 87.5 mm, length 50 mm) were prepared within stormwater pipe (75 mm length, 90 mm external diameter). A fast (Whatman No. 4) filter paper was placed beneath the soil and the soil samples were settled by dropping the core from a height of 50 mm, three times. The average bulk density of the settled soil samples was determined and all cores were subsequently re-packed to this bulk density. Two filter papers were placed on top of the soil column. The columns were placed into a pre-treatment calcium chloride (EC 2.0 dS/m) solution bath and allowed to capillary wet (-4 cm) for a minimum of 12 hours. The columns were then removed from the bath and 1000 $cm^3$ of calcium chloride pre-treatment solution was applied (head ~20 mm) to the top of each column which was placed in a Bucher funnel and open to the atmosphere at the bottom interface. The pre-treatment was allowed to drain for 2 h after the last of the pre-treatment solution had infiltrated and then a second pre-treatment calcium chloride (EC 2 dS/m) solution was applied with a constant hydraulic head (~20 mm measured from the upper surface of the soil column) to each column. The discharge (i.e. flux) from the base of each column was measured at contiguous time intervals until a constant flux was recorded. The hydraulic conductivity was then calculated using Darcy's equation.

A range of up to ten sequentially increasing SAR treatments (0 to ∞) were then applied to each of the five columns where each column was subjected to SAR treatments at a single EC (0.5, 1, 2, 4 and 8 dS/m). The SAR 0 treatment was applied first to each column. In each case, the hydraulic conductivity was measured with a constant head of ~20 mm as for the pre-treatment after a minimum of 1000 $cm^3$ of solution had infiltrated. The relative hydraulic conductivity ($rK_{sat}$) of the column was then calculated by dividing the hydraulic conductivity of the SAR>0 water quality treatments by the hydraulic conductivity measured when the SAR 0 water treatment was applied. The relative hydraulic conductivity data was then used to create a three dimensional response surface (in the form used by Ezlit 2009) for $rK_{sat}$ against solution SAR and EC. The 20% reduction in $K_{sat}$ ($0.8rK_{sat}$) contour was then calculated and represents the soil specific TEC relationship.

A correlation analysis was conducted using 36 soil samples from across south-east Queensland. The SAR required to produce a $0.8rK_{sat}$ was calculated for water with an EC of 1, 2 or 4 dS/m. This critical SAR was then correlated with the CCR, clay content and OMC.

### Results and Discussion

The six soil samples compared were all either Vertosols or Chromsols. While the soil properties were generally similar within each order (Table 1), the TEC curves obtained for these soils were not similar (Figure 1a). For example, where a solution EC of 1 dS/m was applied, the critical SAR resulting in a $0.8rK_{sat}$ ranged between 9 and 17 for the Vertosols and between 3 and 20 for the Chromosols (Figure 1a). These TEC functions clearly show soil specific responses. More importantly, the TEC curves (Figure 1a) within each soil order were not similar and confirm that the two TEC curves shown in the ANZECC (2000) guidelines (Figure 1b) are not appropriate for all soils. According to the ANZECC guidelines, the structural response for all soils when water with a specific SAR and EC is applied can be obtained directly from Figure 1b. However, considering as an example Soil 3 (Figure 1a) where water with an EC 3 dS/m and SAR 7 is applied, soil instability would be expected to occur, but according to the ANZECC diagram (Figure 1b) the soil would remain stable. This has important implications for irrigation management, especially as the incidence of irrigation with saline-sodic water increases.




**(a)** **(b)**

**Figure 1. Comparison of (a) the TEC (i.e. 20% reduction in Ksat) curves for the six soils in Table 1; and (b) the relationship between SAR and EC for soil structural stability (TEC) as it appears in ANZECC (2000), modified from DNR (1997).**

Soil properties that affect the permeability of soils include clay mineralogy, clay content and organic matter content. Hence, it could be expected that these properties should provide a relationship with soil specific TEC responses. However, no significant relationship was found between these properties and the SAR required to produce a $0.8rK_{sat}$ (Figure 2). In all cases the $r^2$ values were <0.1 for the relationships between the critical SAR and CCR, clay content, or OMC.

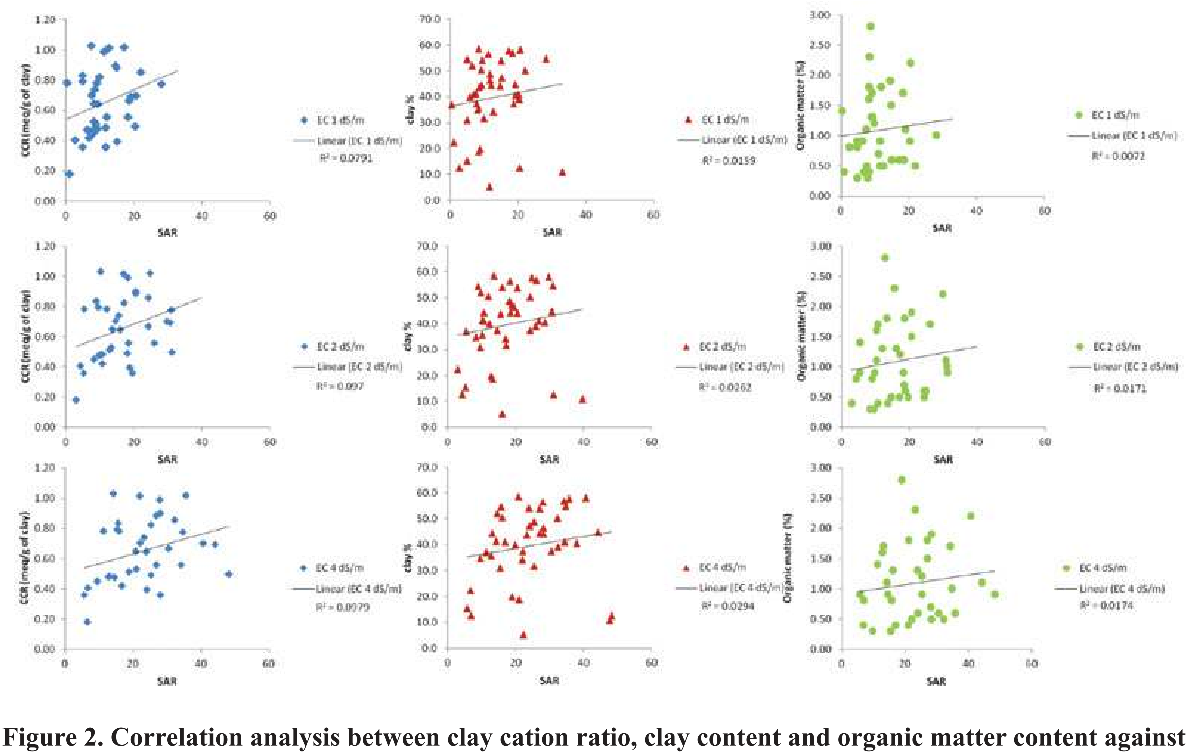



**Figure 2. Correlation analysis between clay cation ratio, clay content and organic matter content against the critical SAR at three threshold EC values for 36 soil samples. These threshold values represent a 20% reduction in saturated hydraulic conductivity ($0.8rK_{sat}$).**

The original depiction (DNR 1997) of the SAR/EC relationship diagram in ANZECC (2000) suggests that the CCR and clay content may be soil properties that are related to the TEC curves. In reference to Figure 1b, the hashed line is defined by a clay content of 55–65% and CCR of 0.55–0.75, while the solid line is defined by a clay content of 25–35% and CCR of 0.35–0.55 (DNR 1997). However, the results in Figure 2 suggest that while there is a weak trend consistent with the DNR (1997) observations, there is not a relationship that would enable the prediction of soil structural responses based on the CCR, clay content or OMC individually.

The CCR is only an indicator used for clay mineralogy, so a relationship between clay mineralogy and TEC cannot be ruled out. Future work could endeavour to broaden the range of soils, evaluate interactions between these soil properties and/or compare quantitative clay mineralogy with TEC values. Furthermore, the results presented in this paper refer to relative changes in $K_{sat}$, rather than actual hydraulic conductivity. Hence, the relationships between CCR, clay content, organic matter and absolute hydraulic conductivity should be investigated.

## Conclusion

This work illustrates that there are significant differences between soil TEC curves for soils, even within the same soil order. There is therefore a need to reconsider current guidelines used for irrigation management, especially as the requirement to irrigate with saline-sodic water increases. Furthermore, the results suggest that there is no apparent relationship between CCR, clay content or OMC and the critical SAR required to produce a $0.8rK_{sat}$. This means that a useful prediction of soil structural responses is unlikely to be obtained using these soil properties alone.

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# Tuesday 4 December

## SOIL AND LAND DEGRADATION

*Presented Posters*

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# Threshold electrolyte concentration for dispersive soils in relation to CROSS

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Most investigations of clay dispersion have concentrated on soils with high exchangeable sodium. Traditional indices for assessing soil structural stability, sodium adsorption ratio (SAR) and exchangeable sodium percentages (ESP) do not take into account the effects of K on soil clay dispersion and swelling. Therefore, a new quantitative index, cation ratio of structural stability (CROSS), was used as an alternative to SAR, to take into account the differential effects of Ca, Mg, Na and K on soil structural stability as well as exchangeable cation ratio percentage (ERC) instead of ESP to take into account the effect of exchangeable potassium. In this study we investigated the relationships between threshold electrolyte concentration (TEC) of soil solutions to CROSS for three soils of different mineralogy (illite-kaolinite and smectite), soil texture, net charge, pH and EC. The TEC, when all soils are included correlates with CROSS with R2=0.93. These relationships are different according to individual type of soil. The significant relationships between CROSS of the soil solution and exchangeable cation ratio (ECR), as well as CROSS and turbidity were established for all three soils when data combined together as well as for every individual soil.

Key words: clay dispersion, soil structure, flocculating power, dispersive potential.

## Introduction

Sodium adsorption ratio (SAR) and exchangeable sodium percentage (ESP) are currently used as indices for assessing the soil structural stability on interaction with water. A few studies have shown that potassium and magnesium ions in the exchange complex of soil can also be related to clay dispersion even when the exchangeable sodium levels are minimal (Arienzo, Christen et al. 2009; Rengasamy 2006; Robbins 1984; Smiles 2006; Subba Rao and Rao 1996). Rengasamy and Marchuk (2011) proposed a new ratio 'CROSS' (cation ratio of soil structural stability) analogous to SAR but which incorporates the differential effects of Na and K in dispersing soil clays, and also the differential effects of Mg and Ca in flocculating soil clays, and is defined as: CROSS = (Na + 0.56K) / [(Ca + 0.6Mg)/2]0.5 where the concentrations of these ions are expressed in millimole of charge/L. In studies conducted since the introduction of this concept, CROSS has been shown to be superior to SAR in predicting hydraulic conductivity changes and clay dispersion in a number of soils (Jayawardane, Christen et al. 2011; Laurenson, Bolan et al. 2012; Rengasamy and Marchuk 2011). We (Rengasamy and Marchuk, 2011) also found that CROSS measured in soil solutions was strongly correlated with to the ratio of exchangeable cations (ECR %). The primary aim of this paper is to establish threshold electrolyte concentration for a dispersive soil in relation to CROSS of the soil solution.

## Methods

Three soils viz. Urrbrae, McLaren and Claremont were used in the present study. Soil samples were air-dried and sieved to 2 mm particle diameter. Selection of these soils was based on the differences in their texture, clay mineralogy, electrical conductivity and pH (Table1). The soils were pre-treated using two sets of treatment solutions, with total cation concentrations 20 and 40 meq/L respectively of CROSStr values of 6, 8, 11 and 15, with gradual increase in concentrations of cations, particularly K. Soil samples were packed into Plexiglas columns at a bulk density of 1.3 Mg/m$^3$. The columns were percolated with three wetting, draining and drying cycles using each of eight CROSS treatment solutions (CROSStr) for Urrbrae and McLaren soils, and seven for Claremont soil. At the end of the last cycle the deionised water was passed through the columns to simulate the infiltration of the soils with rain water. The experiments were conducted using triplicate samples. The soils were removed from the columns, air dried, crushed and passed through a 2-mm sieve.

### *Spontaneous dispersion and turbidity measurements*

Spontaneous dispersion was assessed by a modification of the method described by Rengasamy (2002). Samples (10g) of dry soils were placed into transparent cylinders and 50 ml of distilled water was added slowly down the sides of the cylinders, taking care to avoid disturbance of the soil. After approximately 5 hr, any particles which had dispersed from the soils were gently stirred into suspension and left to stand for 2 hours. Suspensions were pipetted out from 10 cm depth for turbidity measurements. To quantify the amount of < 2μm particles dispersed, measurements were made on a Hach 2100N Laboratory Turbidimeter at 25°C and recorded in Nephelometric Turbidity Units (NTU).

**Table 1 Selected soil properties**

| Soil properties | Units | Soil | | |
|---|---|---|---|---|
| | | Urrbrae | Mc Laren | Claremont |
| Depth | cm | 15-40 | 15-40 | 15-40 |
| pH (1:5 soil water solution) | | 6.7 | 7.3 | 8.3 |
| EC(1:5 soil water solution) | dS/m | 0.061 | 0.139 | 1.03 |
| Total carbon | % | 1.49 | 0.76 | 4 |
| $CEC_{eff}$ | cmol $kg^{-1}$ | 8 | 10 | 33 |
| Dominant clay minerals | | Illite-kaolinite | Illite -kaolinite | Smectite |
| Australian Classification (Isbell 2002) | | Red Chromosol | Red-Brown Earth | Vertisol |
| Texture | | Sandy loam | Clay -loam | Clay |
| Location in South Australia | | Waite Research Institute | McLaren Vale | Waite Research Institute |

The EC and pH of the equilibrium solution were measured and the suspensions were then centrifuged. Soluble cations (Na+, K+, $Ca^2$+, $Mg^2$+) (concentrations mmolc /L) were determined in the 1:5 extracts by ICP. CROSSss of the soil solutions were calculated using the following equations:

CROSSss = (Na + 0.56K) / [(Ca + 0.6Mg)/$^2$]0.5 (mol0.5m-1.5) (1)

where the concentrations of the corresponding ions are expressed in mmolc/L.

The exchangeable cations (Na+, K+, $Mg^2$+ and $Ca^2$+) were determined by the method described in Rayment and Lyons (2011) and subsequently exchangeable cation ratio percentage (ECR %) were calculated as:

ECR %= [(Na++K+) / CECeff)] x100% (2)

where the quantities of the exchangeable cations are expressed in cmolc/kg.

**Results**

Clay dispersion in relation to CROSS

Chorom and Rengasamy (1995) concluded that clay dispersion was highly related to the net particle charge influenced by mineralogy and pH as well as organic carbon. Therefore, it cannot be expected that a relationship between the amount of clay dispersion and CROSS would be similar for the different types of soils. Nevertheless, for the three treated soils used in this study, the clay dispersion measured by the turbidity was highly correlated with CROSSss, with R2 = 0.88 (Fig.1). As for an individual soil type, although the slopes of the regressions differed significantly, correlations for all three soils were high: Urrbrae soil Y=1746.2X-3776 R2=0.89, McLaren soil Y=1889.6X-3199 R2=0.78, Claremont soil Y=1633X-2975. R2=0.76.

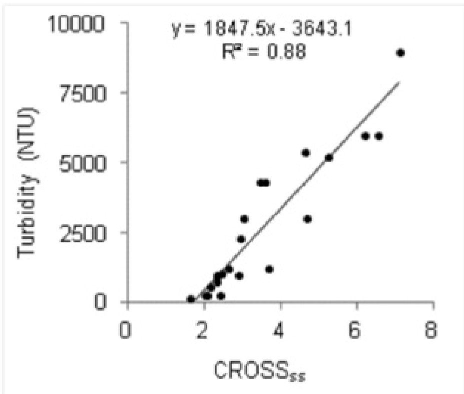


**Figure 1. Relationships between CROSSss and turbidity for all soils.**

Smectitic Claremont soil had clay dispersion even at low CROSSss values and the turbidity was lower than the illitic soils, confirming our earlier results (Rengasamy and Marchuk, 2011)

*Threshold electrolyte concentration in relation to CROSSss*

Cationic effects on soil structural features such as clay dispersion are highly influenced by the corresponding electrolyte concentration which promotes flocculation. For a given value of CROSSss, the electrolyte concentration needed to completely flocculate the clay suspension (or, in other words, completely prevent dispersion) is termed as 'threshold electrolyte concentration (TEC)'(Rengasamy 2002). Similar to the earlier derivations of TEC relating SAR or ESP for different soils (e.g. Quirk and Schofield, 1955; Rengasamy et al. 1984) we have obtained the relation between CROSSss and TEC (Fig 2) at the point of complete flocculation as follows: $Y = 0.45 X + 0.5$, where Y is TEC (dS/m) of the flocculated suspensions and X is CROSSss of the dispersed soil suspensions.

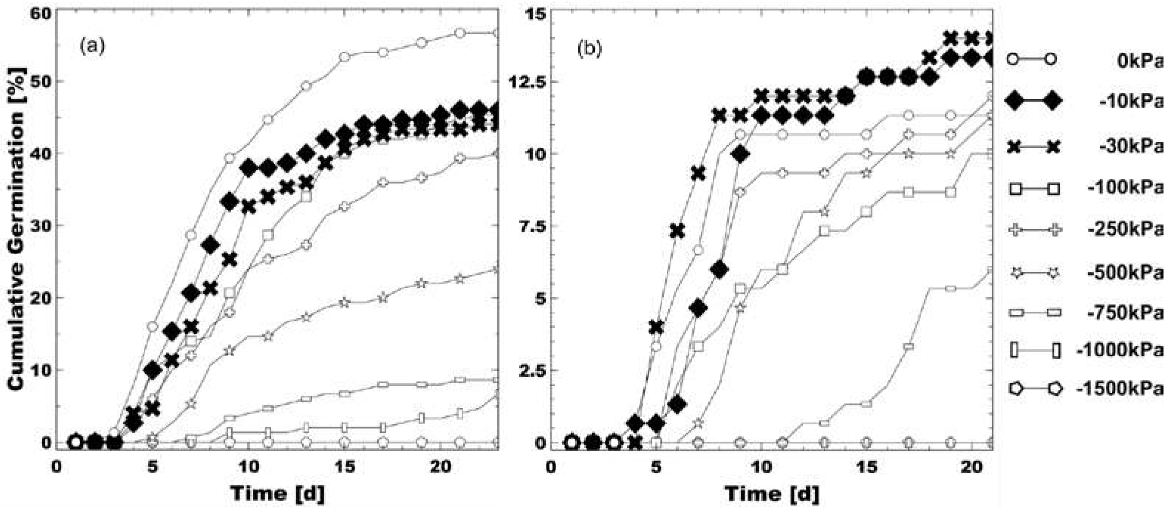


**Figure 2. Relationships between threshold electrolyte concentration (TEC) and cation ratio of structural stability of soil solutions (CROSSss) for all soils.**

Both CROSSss and ECR% of the dispersed soil solutions were highly correlated with the experimentally estimated TEC in individual soil types (Table 2). Similar quantitative relationships for different soil types are necessary for proper soil management. Once developed, these equations will be the basis for maintaining soil structure by adding or maintaining electrolytes below the concentration of which soil physical problems affect soil drainage and indirectly affect plant growth.

**Table 2 Statistical results of linear regression between threshold electrolyte concentration (TEC) of the soil solutions, cation ratio of structural stability of the soil solutions (CROSSss) and echangeable cation ratio (ERC%) of the treated soils solution**

| Soil | X | Y | Regression equation | $R^2$ |
|---|---|---|---|---|
| Urrbrae | | | Y=0.36X+0.91 | 0.96 |
| McLaren | CROSS | TEC | Y =0.35X-1.17 | 0.97 |
| Claremont | | | Y =0.31X+0.58 | 0.94 |
| Urrbrae | | | Y =0.09X-2.35 | 0.75 |
| McLaren | ECR% | TEC | Y =0.085X-0.9 | 0.89 |
| Claremont | | | Y =0.017X+0.58 | 0.89 |

*Statistical calculations and linear regression analysis were performed with the programme Graphpad Prism version 5.01(GraphPad Software, Inc., San Diego, USA).

## Conclusion

We studied dispersive soils using the newly developed concept of cation ratio of structural stability (CROSS). We investigated the relationships between threshold electrolyte concentration (TEC) of soil solutions to CROSS for three soils of different mineralogy (illite-kaolinite and smectite), soil texture, total carbon, pH and EC. Across all soils, the TEC highly correlated with CROSS ($R^2$=0.93). However, the relationships are different for individual soils. Illitic soils differ from smectitic soils, particularly in the slope values. Significant relationships between CROSS of the soil solution and exchangeable cation ratio (ECR), were established for all three soils as well as for each individual soil. Different cations with their specific flocculating power are involved in TEC. Future research will concentrate on using TEC for estimation of the amount of inorganic electrolytes needed to prevent clay dispersion by introducing the flocculating power of the individual cations in the flocculating suspension.

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# The impact of rain water on soil pore networks following irrigation with saline-sodic water

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Abstract

The soil pore network is an important factor affecting soil hydraulic conductivity (Ksat). In this study we examine the effect on the soil pore network of a Red Ferrosol caused by irrigation with good quality irrigation water (TW), as well as saline-sodic water with varying sodium absorption ratios (SAR; 10, 50 and 120) and constant electrical conductivity (EC; 2 dS $m^{-1}$), followed by application of distilled water (simulating rain water). The Ksat was measured for the different waters before and after applying the rain water to the soil. Soil samples were taken from different depths (1, 4 and 8 cm) for analysis of exchangeable cations. Soil horizontal cross-sections were taken from the first 2 cm of the soil cores after drying with acetone and impregnation with polyester resin mixed with green fluorescent dye catalyst and hardener. These sections were polished and visualized under a microscope to investigate the changes in the soil pore network. By increasing the SAR of the water applied from 0.11 (TW) to SAR 50 and 120, a significant reduction in Ksat was found, alongside a significant increase in the exchangeable sodium percentage (ESP) of the soil from 3 to 10 and 11, respectively. This was most evident near the soil surface. After applying rain water, the Ksat reduced significantly approaching 0 mm $h^{-1}$ where soil was treated with water of SAR 120. Visualisation of the soil pore network of the treated soils following the application of deionised water clearly showed a reduction in soil macroporosity where water quality of SAR ≥10 was applied, even where soils were non-sodic (ESP <6%). Where irrigation occurred with good quality, low SAR water this reduction was not evident.

## Introduction

Saline-sodic waters are increasingly being used for irrigation in arid and semi-arid areas of the world, especially in regions where coal seam gas (CSG) is extracted. Consequent to these irrigation practices, sodium accumulates within irrigated soil as a direct relation to the sodium adsorption ratio of the applied water. This commonly results in soil structure degradation, leading to low water infiltration and limited salt leaching (Singh et al. 1992). The main mechanisms for saturated hydraulic conductivity (Ksat) reduction are the processes of swelling and clay dispersion. Swelling is predominant in clay soils containing large quantities of smectitic clay minerals, while dispersion can occur regardless of clay mineral suite. Both of these processes lead to pore blockage, particularly when the soil is leached with low electrolyte water (Minhas et al. 1998). Abu-Sharar et al. (1987) observed a reduction in Ksat in soil leached with low electrolyte water followed by irrigation with saline-sodic water, which they attributed to a reduction in the percentage of macropores within the soil. Similarly, Sumner (1993) explains that a reduction in Ksat might be expected, even in well structured soils; due to leaching with low EC permeate causing a reduction in osmotic pressure and increasing the influence of repelling forces responsible for soil structural degradation. Quirk and Schofield (1955) have also shown that soil structural decline is governed by a critical EC threshold at a given SAR; this threshold being known as the threshold electrolyte concentration (TEC).

Therefore, with an increase in the incidence of saline-sodic solution irrigation practices, there are important management implications to be considered from the above research. It may well be possible to initially irrigate soils with high SAR irrigation solutions provided the EC is sufficiently high (above the TEC) without any apparent consequence to soil structure and Ksat. However, if these soils are subsequently leached with low EC water, such as rainfall, their potential to remain stable is severely reduced. In this paper we examine the impact on water with low EC and SAR on the soil pore network of a Red Ferrosol previously irrigated with saline-sodic water solutions.

## Methods

A sufficient quantity of a Red Ferrosol was collected from the top 15 cm of soil at the Agricultural Field Station Complex of the University of Southern Queensland, Toowoomba. This soil was air dried and crushed to pass a 2 mm sieve, then mixed with tap water up to 18 % gravimetric water content. The moist soil was then packed into PVC tubes (8 cm high and 5 cm internal diameter) at a bulk density of 1 g $cm^{-3}$. A total of 24 cores were packed to allow 6 replicates for each water quality. After packing, the lower ends of the cores were supported by cheesecloth and a 2 cm high (5 cm internal diameter) ring was attached to the top of each core to enable water head application. Selected soil properties are shown in Table 1.

Saline-sodic infiltration solutions were prepared using sodium chloride and calcium chloride to achieve the desired EC and SAR. These solutions included good quality water (TW) with an EC = 0.9 ± 0.1 dS $m^{-1}$ and SAR = 0.11± 0.04; a saline-sodic water of EC of 2 ± 0.1 dS $m^{-1}$ and SAR of 10 ± 0.3 (SAR10); and two high saline-sodic waters, similar to those produced during CSG extraction procedure, with a constant EC of 2 ± 0.3 dS $m^{-1}$ and SARs 50 ± 3 (SAR50) and 120 ± 5 (SAR120).

Cores were allowed to capillary wet (-6 cm) for 24h in the solution they would to be treated with. The cores were then supported in Buchner funnels on wooden stands above 400 $cm^3$ plastic containers. Plastic bottles of 1500 $cm^3$ were used to apply up to 10 pore volumes of each solution treatment using a constant hydraulic head. The leachate was collected and weighed to determine the Ksat of each treatment by use of Darcy's Equation.

**Table 1: Selected physical and chemical properties of the soil used.**

| *Properties* | Mean value ± SE |
|---|---|
| Clay % | 44.1 ± 0.5 |
| Silt % | 25.5 ± 1 |
| Sand % | 30.4 ± 0.7 |
| pH (soil-water ratio 1:5) | 5.8 ± 0.05 |
| EC (electrical conductivity, $dSm^{-1}$) | 0.35 ± 0.001 |
| Exchange sodium percentage (ESP; %) | 3 ± 0.5 |
| Cation Exchange Capacity (meq 100 $g^{-1}$ soil) | 26 ± 1 |

SE represent the standard errors of the mean, $n = 5$

After this treatment process 2 of the 6 replicates were left for three days to undergo free drainage. Exchangeable cations were measured from these cores at three depths (1, 4 and 8 cm). Distilled water (1500 $cm^3$) was applied directly to a further 3 replicates (constant hydraulic head, as per above) from each treatment in order to measure the change in K sat that might be expected during a rainfall event.

One replicate of each treatment before and after treating with distilled water was dried above a bath of acetone for 15 days, with the acetone replaced every 3 days. After drying, the cores were saturated from beneath with a mixture of polyester resin and catalyst with a percent mixture of 30 and 70 %, respectively. The cores were left to dry for 2 weeks under laboratory conditions. Once dry, the soil was taken out of the core and impregnated for 24 h under vacuum in a mixture of polyester resin, catalyst, green fluorescent dye and hardener at the following percent mixture: 40, 50, 5 and 5%, respectively. The soil was left to cure for 7 weeks and was then sectioned with a domain saw into 2 cm horizontal cross-sections. The surface (2 cm) was polished and placed under the microscope to visualise the changes in the soil pore network.

## Result and discussion

Fig. 1 shows the variation in Ksat of the Red Ferrosol treated with varying saline-sodic water (A) and the changes in Ksat after distilled water was applied (B). The changes in Ksat of the soil treated with good quality water (TW) were generally similar to those irrigated with SAR solutions of 10, 50 and 120 (Fig. 1 A). However, when the distilled water was applied, a significant reduction in Ksat was found in all treatments except that treated with TW (Fig. 1B), which maintained a Ksat similar to that in Fig.1A. After applying the distilled water to the soils treated with high saline sodic solutions (SARs 50 and 120) the Ksat decreased to 30 mm $h^{-1}$ and 5 mm $h^{-1}$, respectively, which likely resulted from the reduction of macropores within the soil pore network (Sumner 1993; Minhas *et al.* 1999).

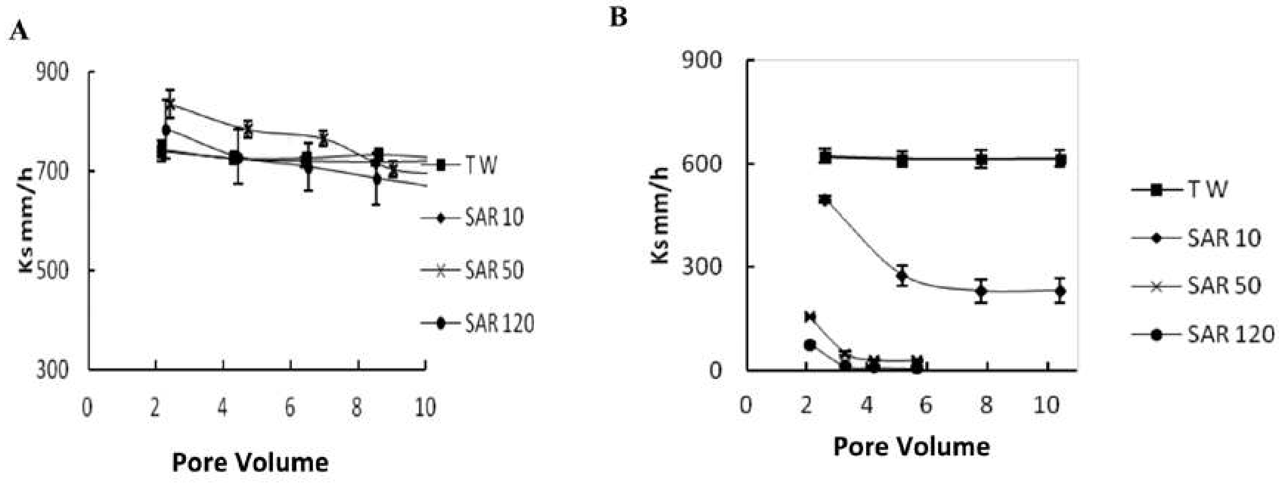


**Fig. 1: Changes in Ksat by pore volume of a Ferrosol soil when water of (A) varying SARs was applied (B) followed by application of distilled water. Intervals represent the standard error.**

The change in exchangeable sodium percentage (ESP) with soil depth for all treatments is shown in Table 2. Soil treated with TW showed a general decrease in ESP throughout the measured depths, which is consistent with application of water containing low ionic concentrations. Given the initial soil properties and the ionic composition of the TW treatment, the observed decrease in sodicity could be expected. Similarly, application of SAR 10 treatment generally resulted in non-sodic conditions, although a slight increase in sodicty is observed and the ESP at depth 4 cm is considered sodic by definition of Northcott and Skene (1972). On the other hand, where SAR 50 and 120 treatments were applied, significant increases in sodicity that would enhance the potential for clay dispersion were observed at all depths. Furthermore, increases were greatest in the surface depth, suggesting that this is initially the zone of greatest importance for soil permeability management.

**Table 2: Changes in exchangeable cation concentrations and ESP of the soil with depth after treatment with water with varying SAR.**

| Parameters | Initial exchangeable cation concentration | Water applied | Mean value of measured properties after treatment | | |
|---|---|---|---|---|---|
| | | | Depth | | |
| | | TW | 1 cm | 4 cm | 8 cm |
| Na (mg Kg-1) | 190 ± 10 | | 129 ± 8 | 184 ± 11 | 156 ± 10 |
| Ca (mg Kg-1) | 1420 ± 50 | | 1760 ± 78 | 1620 ± 70 | 1490 ± 12 |
| Mg (mg Kg-1) | 487 ± 28 | | 91 ± 5 | 215 ± 15 | 490 ± 9 |
| K (mg Kg-1) | 119 ± 23 | | 73 ± 6 | 105 ± 7 | 130 ± 5 |
| ESP | 3.12 ± 0.1 | | 2.11 ± 0.02 | 3.02 ± 0.01 | 2.56 ± 0.01 |
| | | *SAR 10* | *1 cm* | *4 cm* | *8 cm* |
| Na (mg Kg-1) | 190 ± 10 | | 241 ± 11 | 318 ± 14 | 229 ± 19 |
| Ca (mg Kg-1) | 1420 ± 50 | | 1700 ± 25 | 1530 ± 20 | 1360 ± 50 |
| Mg (mg Kg-1) | 487 ± 28 | | 172 ± 4 | 405 ± 9 | 440 ± 17 |
| K (mg Kg-1) | 119 ± 23 | | 50 ± 3 | 57 ± 5 | 92 ± 7 |
| ESP | 3.12 ± 0.1 | | 3.69 ± 0.04 | 5.22 ± 0.07 | 3.76 ± 0.17 |
| | | *SAR 50* | *1 cm* | *4 cm* | *8 cm* |
| Na (mg Kg-1) | 190 ± 10 | | 599 ± 20 | 414 ± 15 | 334 ± 7 |
| Ca (mg Kg-1) | 1420 ± 50 | | 1200 ± 71 | 1470 ± 95 | 1350 ± 73 |
| Mg (mg Kg-1) | 487 ± 28 | | 240 ± 11 | 420 ± 5 | 437 ± 13 |
| K (mg Kg-1) | 119 ± 23 | | 42 ± 7 | 89 ± 9 | 91 ± 6 |
| ESP | 3.12 ± 0.1 | | 9.84 ± 0.17 | 6.80 ± 0.14 | 5.48 ± 0.01 |
| | | *SAR 120* | *1 cm* | *4 cm* | *8 cm* |
| Na (mg Kg-1) | 190 ± 10 | | 679 ± 25 | 462 ± 33 | 384 ± 10 |
| Ca (mg Kg-1) | 1420 ± 50 | | 1000 ± 38 | 1260 ± 16 | 1286 ± 42 |
| Mg (mg Kg-1) | 487 ± 28 | | 277 ± 12 | 390 ± 21 | 420 ± 25 |
| K (mg Kg-1) | 119 ± 23 | | 46 ± 4 | 85 ± 8 | 92 ± 5 |
| ESP | 3.12 ± 0.1 | | 11.15 ± 0.2 | 7.59 ± 0.24 | 6.3 ± 0.1 |

SE represents the standard error of the mean

The soil horizontal cross-sections taken after treatment with SAR solutions and those following the application of distilled water are shown in Figure 2A and 2B, respectively. There is an apparent decrease in soil macroporosity with increasing solution SAR application (Fig. 2A). These results support those observed in Fig. 1A, while Ksat was not reduced severely by SAR 50 and 120 treatments, presumably due to an EC effect (Sumner 1993), a slight decline in Ksat was observed. This is attributable to a decrease in macroporosity (Fig. 2A). When distilled water was applied to cores previously treated with SAR solutions, there was an obvious decrease in all treatments, except for the TW treatment (Fig. 2B). Soils treated with high SAR solutions (SAR 50 and 120) exhibited a reduced soil pore network, which can be attributed to pore blockages from clay dispersion under sodic conditions (Table 2); hence, the observed Ksat reduction in Fig 2B. Importantly, the results for the SAR 10 treatment (Fig. 2A and B) show that an obvious reduction in the macroscopic soil pore network has occurred in the top 2 cm, even in the absence of sodic conditions (Table 2), which has resulted in a significant reduction in Ksat (Fig. 1B). This illustrates that a solution sufficiently low in EC is capable of causing soil dispersion and subsequent pore blockage in non-sodic soils (Sumner 1993) and highlights the importance of managing soil EC.

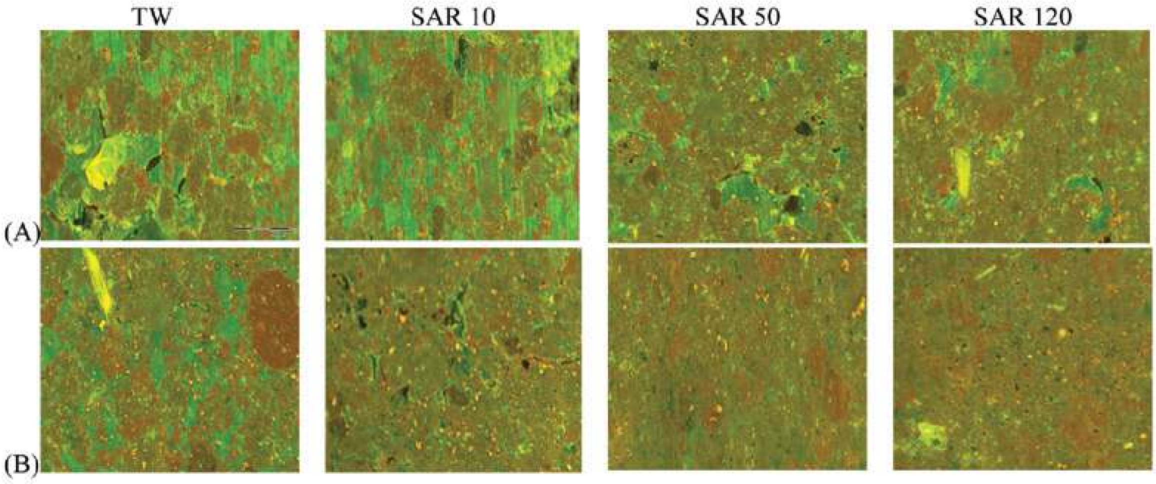


**Figure 2: Microscope images of the 2 cm horizontal cross-sections of the Red Ferrosol pore networks after treatment with (A) solutions of varying SAR; and (B) SAR solutions followed by distilled water. The green and yellow portions of the image represent florescent dyed resin filled pores; black portions represent pore space that was not impregnated; and red/brown portions are soil particles.**

## Conclusion

The results of our experiment on using varying saline-sodic water followed by rain water to irrigate a Red Ferrosol soil indicate that the soil pore network can be maintained in a reasonable condition even where solutions of high sodicity are applied, provided the solution EC is sufficiently high. However, the pore network is adversely affected by the SAR of water applied when low EC solution is allowed to percolate these soils, such as rainfall. This effect was greatest when the SAR of the applied water exceeded 10; but was significant even at SAR 10. Furthermore, visualisation of pore networks show that even soils with non-sodic conditions undergo substantial reduction in macroporosity when low EC solution is allowed to percolate through soil. This highlights the importance of managing soil EC in order to maintaini soil pore networks.

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# Sustainable landscape design for coal mine rehabilitation using runoff and erosion modeling

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### Abstract

Historically, the landform design component of mine site rehabilitation has been plagued by failure to recognise the need for improved planning. This study compared waste dump landform design outputs using existing landform guidelines with a design based on output from the Water Erosion Prediction Program (WEPP) runoff and erosion model. The landform design approaches were applied at two mine sites in the NSW Hunter coalfield. Material characterisation is essential in the Hunter environment, where most landforms rely heavily on establishment of vegetation to achieve erosion stability. Sodic soils are common, and make stabilisation of landforms difficult as they typically produce high rates of runoff and erosion, and are extremely difficult to vegetate. Importantly, the WEPP runoff and erosion modelling process had the advantage of providing explicit information on potential runoff and erosion rates and clear identification of priorities for the rehabilitation and landform stabilisation process. The modelling-based approach to landform design also included a comprehensive review of cover material characterisation.

### Key Words

Erosion, slope stability, WEPP, landform design.

### Introduction

Historically, the NSW Department of Primary Industries (DPI) has prescribed slope criteria for coal overburden landforms, with a targeted 10 degrees outer batter slope gradient. Gradients of up to 18 degrees have been accepted for ramps and low walls. There have been persistent problems with drainage stability, as waterways on steep slopes are expensive to construct and maintain and expensive to repair if they fail (DPI, 2008). There is also growing experience on mine sites across Australia that engineered drainage control systems are not self-sustaining; either requiring long-term maintenance or becoming major causes of landform instability because they concentrate overland flows.

Traditional rehabilitation design elements also include:

- berms to either capture and hold water or convey it to drainage points,
- infiltration basins, terrace benches, and
- drains stabilised with rock-filled gabions, concrete linings or rock.

Rehabilitated landforms built using these methods depart considerably from the stable natural landforms surrounding them, both in performance and appearance. They are effectively flow-concentrating, and observations of such landforms (both constructed and natural) across Australia leads to the conclusion that they are predisposed to gully erosion.

This study, as part of a larger project (C18024) commissioned by the Australian Coal Association Research Program (ACARP), investigated two mine sites in the NSW Hunter coalfields. The work compared landform designs developed using existing landform guidelines (traditional designs) and on the basis of runoff and erosion model output using WEPP and SIBERIA.

### Methods

Simulations of runoff and erosion of a range of slopes were carried out using the Windows version of the WEPP model (Flanagan and Livingston 1995). WEPP is a simulation model with a daily input time step, but internal calculations can use shorter time steps. For every day, plant and soil characteristics important to erosion processes are updated. When rainfall occurs, those plant and soil characteristics are considered in determining whether runoff occurs. If runoff is predicted to occur, the model computes sediment detachment, transport, and deposition at points along the slope profile, and, depending on the version used, in channels and reservoirs.

Assessment of the use of WEPP to predict interrill and rill erosion under mine site conditions has shown that its use by Landloch has produced excellent agreement between measured and predicted erosion rates (Howard and Roddy 2012), providing considerable confidence that WEPP predictions are an effective tool for the types of landform investigations reported here.

The WEPP runoff/erosion model was run to establish slope profiles for the two waste dumps that satisfied landform constraints and minimised erosion. Erodibility parameters for the WEPP model were determined using measurements of erosion and runoff from flumes where topsoil and overburden samples were exposed to a range of overland flow conditions to measure critical shear ($\tau_c$) and rill erodibility ($K_R$); and plots exposed to simulated rainfall to measure interrill erodibility ($K_i$) and infiltration rates for derivation of an effective hydraulic conductivity parameter.

Simulations did not consider gully erosion, which is beyond the scope of the WEPP model.

**Table 1: Infiltration and erodibility parameters**

| Property | Ashton | | Glendell | |
|---|---|---|---|---|
| | Subsoil | Overburden | Topsoil | Overburden |
| Interrill erodibility ($K_i$) (kg.s/m$^4$) | 5,120,010 | ND* | 2,668,385 | 2,368,006 |
| Steady Infiltration rate (mm/h) | 3.7 | >103 | 4.3 | 41.1 |
| Effective hydraulic conductivity (mm/h) | 0.75 | 41 | 0.4 | 12 |
| Rill erodibility ($K_R$) (s/m) | 0.002726 | 0.000408 | 0.003975 | 0.002413 |
| Critical shear ($\tau_c$ ) (Pa) | 3.61 | 19.46 | 21.36 | 9.59 |

*ND (No Data): No measurable erosion occurred due to absence of runoff

**Results**

Initial simulations demonstrated that the Glendell topsoil has the greatest erosion potential of all four materials (Figure 1), with the specific patterns of predicted erosion being a function of both detachment rates and transport capacity. The Glendell topsoil's high critical shear meant that rill detachment was slower to develop for this material than for all others, but its relatively fine particle size meant that predicted capacity to transport this material in runoff was much higher than for the other materials.

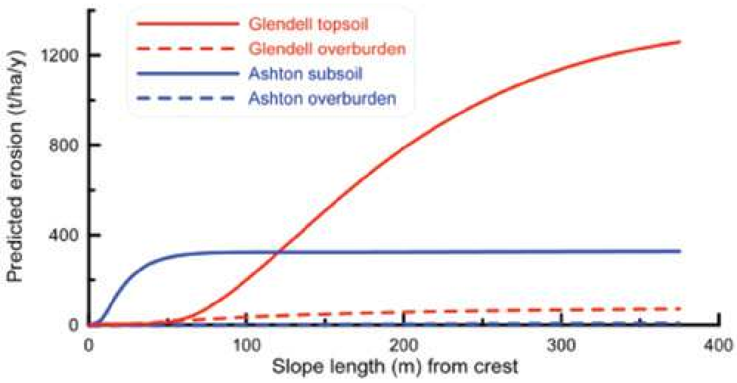


**Figure 1: Predicted erosion for the four materials investigated along a 60 m high batter slope with a 16% gradient, bare surface and 5 m rill spacing.**

A series of model runs were then carried out to identify the material properties and surface conditions necessary to achieve stable slopes of Glendell topsoil.

Assessment of the impact of effective hydraulic conductivity of the Glendell topsoil on predicted annual average erosion showed a rapid exponential decline in predicted annual erosion as soil hydraulic conductivity increases beyond 6 mm/h (Fig. 2).

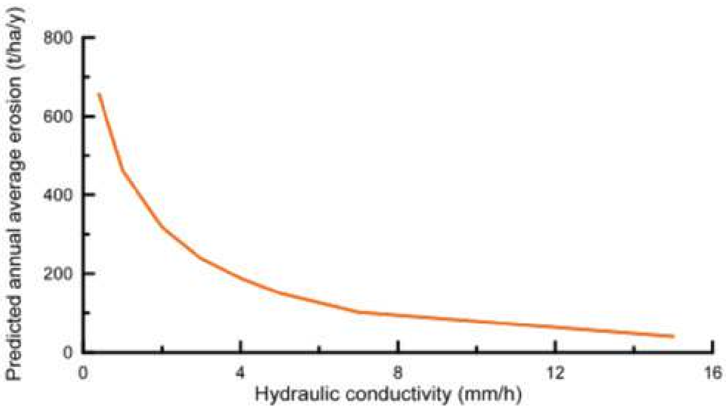


**Figure 2. Predicted impact of effective soil hydraulic conductivity on annual erosion from 60 m high slope, bare surface, 5m rill spacing, assuming all other factors are unchanged.**

*Effect of cover on slope stability*

Factors describing the direct impact of cover on erosion (C factor) were derived from the Revised Universal Soil Loss Equation (RUSLE) (Renard *et al*. 1993), with care taken to eliminate effects of soil consolidation and root growth from the C-factors derived. When the impacts of cover on runoff rates, rill spacing, and sediment detachment and transport are combined, the resultant predicted impacts of cover on annual erosion show that 50% grass cover on the Glendell topsoil will give an average soil loss rate of <2 t/ha/y, with a peak erosion rate of >6.5 t/ha/y. These results are consistent with recommended cover levels identified in the literature. Contact cover of 50% has been reported as being the minimum necessary to effectively reduce erosion losses under natural rainfall in Queensland (McIvor *et al*. 1995; Carroll *et al*. 2000; Carroll and Tucker 2000), with Grigg *et al.* (2001) recommending projected cover levels of 70% to ensure maximum surface stability but also stating that significant movement can still occur for cover > 70% on slopes exceeding 12-15%.

*Concave profile developed using WEPP*

The WEPP model was run on bare topsoil to develop a concave slope profile that minimised predicted erosion. The concave profile developed was composed of a number of linear segments of varying gradient (Table 1). The waste dump footprint was constrained and slope length couldn't be increased, the resultant slope -when bare of vegetation - was predicted to erode at rates greatly in excess of "tolerable" levels (Fig.4).

**Table 1. Concave profile developed for Glendell topsoil using WEPP**

| Horizontal distance (m) from crest of slope | Segment gradient |
|---|---|
| 0-97 | 25% |
| 97-127 | 20% |
| 127-170 | 16% |
| 170-270 | 12% |
| 270-375 | 10% |

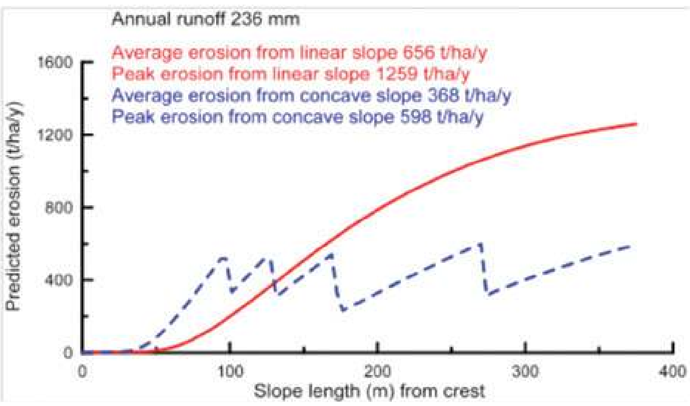


**Figure 3. Comparison of predicted erosion rates from the linear (blue - dash) and concave (red - solid) profiles of bare Glendell topsoil, 60 m high and 16% gradient.**

The main concern for the concave slope profile developed was whether the levels of vegetative cover, likely to be developed on rehabilitated areas, would be sufficient to further reduce erosion (Fig. 3) to target levels (<2 t/ha/y average erosion). To test the effectiveness of the concave profile, infiltration, rill spacing, and erosion factors consistent with 50% surface vegetative cover were used to model runoff and erosion. The results (Fig. 4) show that with 50% cover, the concave profile is predicted to produce levels of erosion that are lower than the identified stability targets.

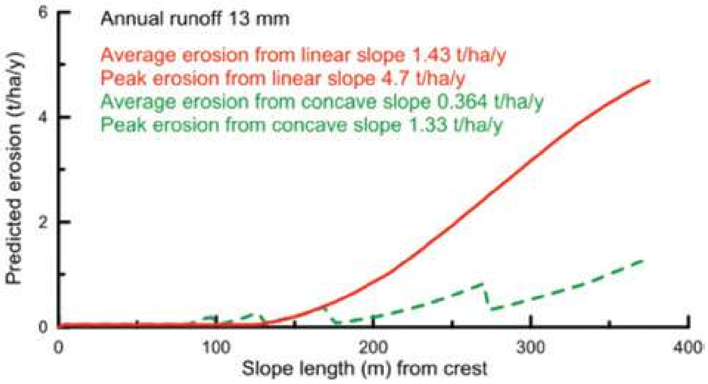


**Figure 4. Comparison of predicted erosion rates from linear (red) and concave (dashed green) profiles of Glendell topsoil with 50% vegetative cover, 60 m high and 16% gradient.**

## Conclusions

This study illustrates the importance of a comprehensive approach to rehabilitation planning, with landform design being an essential component of planning, rather than a standalone activity. Applying caution to the conditions and limitations, the WEPP runoff/erosion model established a slope profile that satisfied landform constraints and minimised erosion. In practice, landform design should be influenced by material characterisation, the requirements for vegetation establishment, and equally, vegetation strategies should take account of the requirements for landform stabilisation and efficient hydrologic function.

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# Comparative effects of sodium and potassium in irrigation water on chemical properties of a vineyard soil of the Barossa Valley

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Although winery wastewater is increasingly used for irrigation in areas with limited irrigation water supplies, high concentrations of salts can eventually degrade the soil structure and reduce water availability to grape vines. The effects of sodium on soil properties are well understood but those of potassium are contradictory. In this study, a paired-site comparison was conducted on a texture-contrast soil irrigated for eight years with high quality water from the Barossa Infrastructure Ltd (BIL) having low SAR and low Potassium Adsorption Ratio, PAR, and winery wastewater from the North Para Environmental Committee (NPEC) having a high SAR and PAR. Soil samples were taken at various distances from the irrigation drippers at the end of the irrigation seasons (April-May) and after the winter rains (November) to determine solution cations and pH in saturation paste extracts and to calculate EC, SAR and PAR.

Although the non-irrigated control sites for the BIL-irrigated soils were (naturally) higher in salt than those for the NPEC-irrigated soils, the composition and concentration of salts under the drippers reflected those in the source waters. For example, the SAR and PAR in the saturation paste extracts of the soil irrigated with NPEC water (high concentrations of Na and K) were higher than those irrigated with BIL water. At this stage sodicity hazard is very high in both BIL (natural sodicity) and NPEC (induced sodicity) irrigated blocks whereas as K is not of serious concern in either of them. Laboratory-based column-leaching experiments are currently underway to evaluate the longer term effects.

# SoilWatch – a practical approach to monitoring soil health and the impact of land management change in NSW

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SoilWatch is a performance monitoring tool which allows public funding bodies to quantify the impact of investment by measuring on ground changes in soil health over time. It is a partnership program which provides support to NSW CMAs through training, field days and program development. Since 2008, over 250 site pairs (each containing a 'treatment' and a 'control') have been established in NSW.

An 18 page report is generated from the OEH laboratory database for each site pair. It includes the soil test results, photos of the site and groundcover and 10 pages of soil test information. This evidenced based framework allows managers with limited soil assessment experience to provide site specific soil health and land management advice with confidence.

SoilWatch protocols are consistent with both NSW and National soil monitoring standards at 0-5cm and 5-10 cm. An experiment comparing SoilWatch sampling methodology to those of the National and NSW soil monitoring protocols has been conducted. Initial results indicate that the methods are comparable, which builds confidence in our approach.

Future directions include incorporation of the eight class NSW land and soil capability assessment from test results and location data, which is then linked with a spatial database. This powerful tool will identify land being managed either within or beyond capability. Time series MODIS satellite data will be explored to track cumulative biomass and sheet erosion differences between control and intervention sites, and may be useful for developing remote monitoring of soil carbon.

# Response of microbial activity and biomass to increasing salinity depends on the final salinity, not the original salinity

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To test if microbes in saline soil are more tolerant to increasing salinity than microbes from non-saline soil, an incubation experiment with five soils ($EC_e$ ranging from 1 to 50 dS $m^{-1}$) was conducted, the EC was increased to 37 $EC_e$ levels (from 3 to 119 dS $m^{-1}$) by adding NaCl. Pea straw was added at 20 g $kg^{-1}$ to provide a nutrient source, the soils were incubated at optimal water content for 15 days, soil respiration was measured continuously and microbial biomass C was determined every three days. Cumulative respiration at a given adjusted EC was not affected by the original soil EC. Microbial biomass in all soils increased from day 0 to day 3, then decreased. The relative increase was greater in soils which had a lower microbial biomass on day 0 (which were more saline). Therefore the relative increase in microbial biomass appears to be a function of the biomass on day 0 rather than the EC. The results suggest that microbes from originally saline soils are not more tolerant to increases in salinity than those from originally non-saline soils. The strong increase in microbial biomass upon pea straw addition suggests that there is a subset of microbes in all soils that can respond to increased substrate availability even in highly saline environments.

# Tuesday 4 December

## THE LIVING SOIL

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# Nitrogen mineralization and nitrification in Australian soils treated with organic amendments

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### Abstract

Nitrogen (N) dynamics were studied in 3 different soils with varying pH (Adelaide Hills soil, AH- 4.89; Sunny Vale soil, SV- 8.02 and Mawson Lakes soil, ML- 9.15) under laboratory conditions for 80 days. Soils were treated with urea, chicken manure, green waste compost and biosolids at 900 kgN/ha and exposed to two different temperatures (18°C and 37°C). Net N ammonification rate at 37 °C was higher compared to that at 18 °C. NH4- N was found to be the dominant form of mineral N. There was no greater net N mineralization in the soils treated with organic N sources than in those treated with mineral N. Since there was no appreciable increase of NO3- N in AH and SV soils, a short term nitrification assay (SNA) study was performed to investigate the nitrification potential. The low SNA found in these two soils indicates either a low initial population of nitrifying bacteria or that the bacteria are in a dormant state. Our results indicate that soil pH, temperature and indigenous soil microbial community play a vital role in determining the fate of N transformations in soils irrespective of the N sources.

### Key Words

Ammonification, fertilizer, nitrification, nitrogen, organic amendments.

### Introduction

Application of organic wastes such as animal manures, sewage sludge and municipal green wastes to soil is increasing in farming practices primarily for maintaining soil organic matter thereby enhancing crop productivity and also as an ecofriendly waste management technique (Bolan *et al.* 2010; Rahn *et al*. 2009). These wastes are rich in nitrogen (N) which provide high agricultural value, especially in organic farming systems, and also stimulate the soil microbial activity.

Nitrogen transformations from soil and applied organic amendments are regulated by factors such as soil pH, texture, substrate availability (C:N) and environmental factors such as temperature and rainfall. Soil microbial populations also constitute a critical element in the N dynamics of soil controlling the amounts of bioavailable N (NH4- N and NO3- N) (Robertson & Groffman 2007).Organic amendments vary greatly in their composition, degree of stabilization or decomposition and rate of release of nutrients. For managing nutrient cycling from organic amendments, it is therefore necessary to study the amendments' decomposition rates in different soils at varying temperatures. In this study we compared the N dynamics between inorganic (urea) and organic (green waste compost, biosolids and chicken manure) N sources using three soils with varying pH and at two temperatures.

### Methods

*Soils, N sources and laboratory incubation*

The soils used in this study were collected from 3 sites (0-15 cm) located at a range of locations in South Australia. Four different N sources, urea, chicken manure, biosolids and green waste compost were used in this study. Soil samples and amendments were air dried, subsequently sieved (2 mm mesh) and stored in a dry place at room temperature. Subsamples were used to analyse the physicochemical properties of the soils and amendments (Table 1). The short term nitrification assay (SNA) of soils was measured by a modified method of Hart *et al*. (1994). Soil samples (225 g) were mixed with amendments at a rate of 900 mg N/kg soil. Soil without any amendments was used as a control. Soils were then incubated aerobically at 18°C and 37°C constant temperature rooms. Each treatment was replicated three times to give a total of 45 experimental units (3 soils*4 amendments*3 replicates + 3 control soils*3 replicates) at each temperature. Soil water content was adjusted to field capacity and N transformation was monitored in the incubated soil samples for 80 days.

**Table 1: Soil and amendment properties**

| Soils / Amendments | pH | Total C (g/kg) | Total N (g/kg) | SNA (mg/kg/hr)* |
|---|---|---|---|---|
| Sunny Vale soil (SV) | 8.02 | 9.2 | 0.42 | 0.035 |
| Adelaide Hills soil (AH) | 4.89 | 27.6 | 1.06 | 0.105 |
| Mawson Lakes soil (ML) | 9.15 | 19.1 | 1.18 | 1.020 |
| Chicken manure (CM) | 7.62 | 172 | 31.2 | |
| Green waste compost (GWC) | 8.82 | 171 | 14.3 | |
| Biosolids (BS) | 6.86 | 260 | 41.7 | |
| Urea (U) | 7.35 | 204 | 465 | |

*SNA = Short term nitrification assay

*Soil extraction and analysis*

Soil (3 g) was sampled at various intervals and extracted by shaking with 2M KCl (soil: KCl = 1:10) for 1 hour in an end-over-end shaker (Apthorp *et al*. 1987), centrifuged and filtered. The soil extract was analysed for the NH4-N and NO3-N by spectrophotometric analysis.

## Results

*Ammonification*

Urea treated soils showed a rapid increase in NH4-N (i.e. ammonification) at 18°C with the ML soil showing a maximum NH4-N concentration compared to other two soils (Table 2). Similar to 18°C, urea treated soils at 37°C also showed an immediate increase in soil NH4- N during the initial days of incubation (data not shown). At both incubation temperatures, urea-treated soils showed a higher NH4-N concentration compared to soil treated with organic amendments. Irrespective of the soil type and incubation temperature, NH4-N concentration varied in the following order: urea > CM > BS > GWC >control. For example, peak NH4-N in urea-treated SV soil at 18°C was 1490 mg N/kg and in BS treated SV soil is 221 mg N/kg (Table 2 and Figure 1). Similarly, NH4-N concentration varied among organic amendment treatments. For instance, at 18°C incubation study, GWC treated ML soil showed a lower NH4-N concentration (10 mg/kg) when compared to that in BS treated ML soil (261 mg/kg). However, compared to control soils, there was a significant increase in NH4-N concentration in CM and BS treated soils at both the temperatures. In addition, time taken to achieve peak NH4-N concentration (i.e. peak ammonification) varied among treatments (Table 2) and temperatures (data not shown) indicating that the rate of ammonification varied between amendments.

*Nitrification*

The pattern of NH4-N decrease and nitrate build-up (i.e. nitrification) varied among the soils and temperatures regardless of the N sources. For example, at 18°C, ML and SV soils showed clear evidence for nitrification of urea with nitrate build-up starting from day 14 whereas there was no evidence for nitrification in urea-treated AH soil until day 49. This can be related to the low SNA value in this soil (Table 1).

In contrast to 18°C samples, nitrate was produced in very low amounts in all the soils at 37°C (data not shown). For example, urea-treated ML soil showed a maximum 1513 mg NO3-N/kg at 18°C whereas urea-treated ML soil at 37°C showed a maximum of 60 mg NO3-N/kg. Nitrate build-up varied among urea and organic amendment treatments at both the temperatures. For instance, at 18°C study, SV soil + urea produced a maximum of 625 mg NO3-N/kg whereas SV soil + BS treatment produced 279 mg NO3-N/kg (Figure 1). Similarly, nitrate build-up varied among organic amendment treatments with CM and BS giving higher nitrate concentrations than GWC at 18°C.

Table 2: **Ammonification and nitrification reactions as indicated by peak NH4-N and NO3- N concentrations and the time taken to achieve the peak concentrations at 18°C. Values presented are a mean of three replicates.**

| Soil | Amendments | Ammonification | | Nitrification | |
|---|---|---|---|---|---|
| | | Peak $NH_4$-N (mg/kg) | Day of peak value | Peak $NO_3$-N (mg/kg) | Day of peak value |
| Mawson Lakes (ML) soil | Control | 18 | 4 | 15 | 49 |
| | Urea | 1534 | 2 | 1513 | 42 |
| | Green waste compost | 10 | 3 | 53 | 77 |
| | Chicken manure | 284 | 1 | 542 | 77 |
| | Biosolids | 261 | 2 | 364 | 49 |
| Adelaide Hills (AH) soil | Control | 21 | 21 | 18 | 35 |
| | Urea | 1343 | 6 | 48 | 77 |
| | Green waste compost | 34 | 14 | 121 | 77 |
| | Chicken manure | 647 | 28 | 243 | 77 |
| | Biosolids | 292 | 6 | 227 | 77 |
| Sunny Vale (SV) soil | Control | 121 | 3 | 43 | 77 |
| | Urea | 1490 | 6 | 625 | 49 |
| | Green waste compost | 18 | 3 | 65 | 42 |
| | Chicken manure | 301 | 4 | 502 | 49 |
| | Biosolids | 221 | 5 | 279 | 77 |

To sum up, compared to the NH4-N build-up in all the treatments, there was no remarkable build-up of NO3-N in the treatments at 37°C nor in some of the 18°C treatments (SV soil+ urea; AH soil+ urea) (Table 2). In addition, compared to 18°C treatments, nitrate was produced in very low amounts in all the treatments at 37°C.

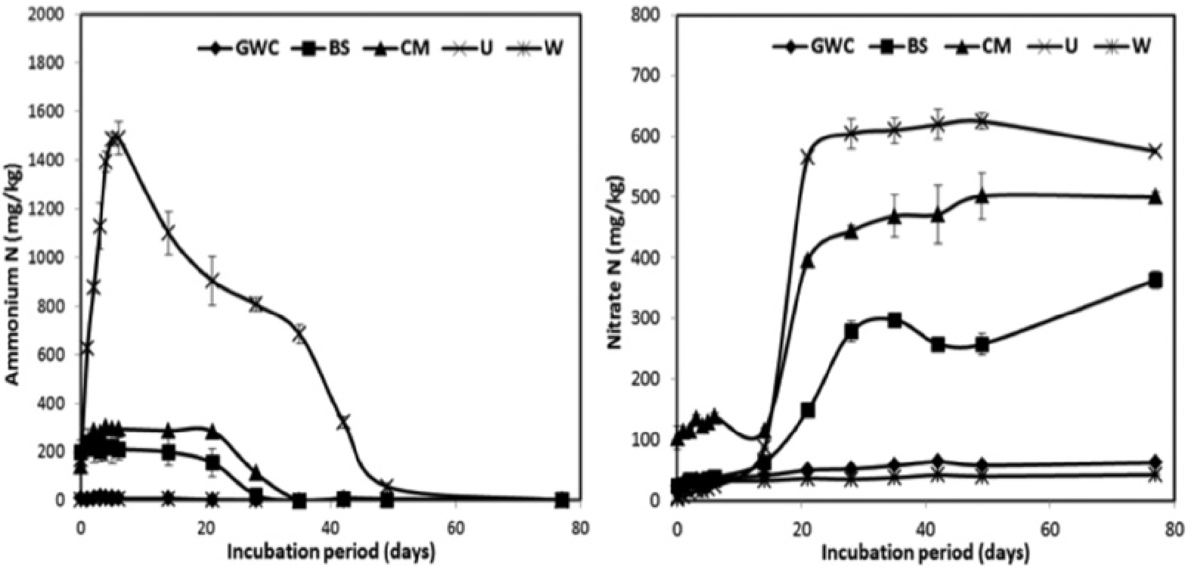


**Figure 1: The change in ammonium N and nitrate N concentrations during 18°C incubation of Sunny Vale soil (SV) treated with green waste compost (GWC), biosolids (BS), chicken manure (CM) or urea (U) or water/ control (W) at 900 mgN/kg soil. Values presented are a mean of three replicates with standard error bars shown.**

**Discussion and Conclusion**

Results show that ammonification rate increased with an increase in temperature in all the soils regardless of the soil pH which is consistent with findings of Myers (1975). Ammonification rate was higher in mineral fertilizer treated soils than in soils amended with organic N. Similar results were obtained by Eneji *et al*. (2002) when they compared N dynamics in soils treated with urea and organic N sources such as chicken manure, swine manure and cattle manure. The slow increase of $NO_3^-$ N in urea-treated AH soil (acid soil) when compared to urea-treated ML and SV soils at 18°C can be related to low SNA activity in this soil. Since there was no decrease in $NH_4$-N concentration, the loss of $NH_4$-N as $NH_3$ was considered highly unlikely. Similar results were obtained in studies by Khalil *et al*. (2005) in which the acidic soils used in their study predominantly accumulated $NH_4$-N, with nitrification either small or stable.

In general, nitrification rate in the soils at 18°C followed: AH soil (acid soil) < SV soil (alkaline soil) <ML soil (highly alkaline soil). This supports the results obtained from SNA study in which there was no nitrate build-up in AH soil. Khalil *et al*. (2005) also stated that nitrification was faster in the high pH soils than in the two acidic soils. In general, nitrate concentration was higher in urea-treated soils than in soils treated with organic amendments which can be compared with the findings of Xie and MacKenzie (1986) in which they stated that about 1 to 5 kg manure-N was found to be equivalent to 1 kg of urea-N in terms of increasing soil nitrate concentrations. Etinger-Tulczynska (1969) showed that the temperature 37-40°C had an injurious effect on soil nitrifiers and hence production of nitrate from ammonium was inhibited at 37°C when compared to 28°C. These findings are consistent with our results where nitrification was lower at 37°C than at 18°C.

These findings will have important implications in determining N dynamics in agricultural soils when applying various N sources. The strategy of applying N fertilizers to the soils should be carefully analysed and particular attention should be paid to soil indigenous microbial community actively involved in nitrogen transformations and their response to change in soil pH. In the absence of active nitrifying bacteria, nitrogen can be wasted, adding cost but no benefit to agriculture.

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# Impacts of tillage on earthworm abundance and distribution in a silt loam soil

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## Abstract

The effects of different types of tillage on lumbricid earthworm populations, in a silt loam soil, were assessed over an 11 year period in a field trial site. A rotation of spring sown crops was grown using tillage systems that were either: (i) intensive tillage (I), (ii) minimum tillage (M), or (iii) no tillage (NT). The first seven years included 'with' and 'without' winter cover crop (forage brassica) treatments (+/- CC). A permanent pasture (PP) control and a permanent fallow (PF) were also incorporated into the trial design. Results from 2000-2011 for earthworm abundance, biomass, community composition, and total soil organic carbon concentration at three depth increments (0-25 cm) are discussed in this paper. Throughout the experiment by far the highest abundance of earthworms was found under PP. A decline in earthworm abundance was observed in the tillage treatments, but it did not vary in relation to type of tillage used, with populations of similar size to PF. Neither was there a strong effect of winter cover crops on earthworm abundance. After 11 years, there were no net changes in SOC overall under the different tillage systems, although a greater concentration of total soil organic carbon was found near the soil surface in NT compared to at depth (15-25 cm). Earthworm populations followed a similar pattern, with a greater proportion of earthworms populating the top 0-7.5 cm in NT. There were no differences in earthworm biomass between the tillage treatments and PF, Intensive tillage resulted in a greater decline in species diversity.

**Key words:** Earthworms; lumbricids; soil organic carbon; permanent fallow; permanent pasture; intensive tillage; minimum tillage; no tillage; winter cover crop

## Introduction

The importance for soil fertility of ingestion and mixing of crop residues and soil by earthworms has long been acknowledged (Russell 1910; Feller *et al*. 2003). Earthworms play an important role in ecosystem functioning by increasing microbial activity (Ernst *et al.* 2008) and enhancing turnover of organic residues and the associated mineralisation (Bohlen *et al*. 1997). Tillage operations and food availability are generally considered the most important factors controlling the abundance and diversity of earthworms in agro-ecosystems (Chan 2001). Many studies concerning the impact of conventional tillage practice on earthworm populations document increases in earthworm populations and diversity with a reduction in tillage (Chan 2001; Lagerlof *et al*. 2012). However, not all studies show a tillage effect (Ernst and Emmerling 2009; Umiker *et al*. 2009). There is growing evidence that different earthworm responses to tillage can be expected in different soil types, with the most negative effects of ploughing found in heavier clay soils and smaller differences in the coarser textured, lighter soils (Joschko *et al*. 2009; Umiker *et al*. 2009). Regardless of whether tillage affects earthworm abundance it is anticipated that there will be differences in earthworm community composition with different types of soil tillage (Umiker *et al*. 2009). It is also expected that the vertical distribution SOC will be affected by tillage practices. Ernst and Emmerling (2009) demonstrated that soil tillage modifies the vertical gradient of SOC and that there is more in the topsoil under reduced tillage systems. However, overall profile C may not change as SOC concentrations deeper in the soil may remain the same or decrease (Umiker *et al*. 2009). Given the functional importance of earthworms, the effect tillage has on their abundance and depth distribution is of particular interest.

## Material and Methods

The trial was initiated in spring 2000 on a Wakanui silt loam (Udic Dystocrept) at Lincoln, Canterbury, New Zealand (43°40'S latitude, 172°28'E longitude; mean annual air temperature 11.4°C, mean annual rainfall 867 mm (Cornforth 1998)). Prior to trial establishment, the site had been under permanent, irrigated, sheep-grazed pasture that had not been cultivated for 14 years. The trial includes six cultivation treatments comprising different combinations of spring and autumn tillage types. The cultivation treatments analysed in this paper consist of the three main tillage treatments where the same tillage procedure that was carried out in spring was carried out in autumn, i.e., (i) Intensive (I): mouldboard plough down to 200 mm depth followed by secondary cultivation (one pass with a spring tined implement followed by harrowing and rolling twice); (ii) Minimum tillage (M): the top 100 mm was disced, followed by secondary cultivation (harrowing and rolling twice) or (iii) No tillage (N): direct sowing without any soil tillage. The experiment had a split plot design with tillage type (I, M and N) as main-plot treatment and winter cover crop (+/- winter forage crops) as sub-plot treatment (denoted +/- CC). The main plot size was 28 m x 9 m. From 2008 the -CC plots were changed to include autumn sown cereal crops, but +CC plots remained unchanged. Plots representing the original ryegrass-clover pasture (PP) were maintained within the trial as a control, alongside a chemically-induced permanent fallow (PF) where the dominant herbicides used were glyphosate and tribenuron methyl. The trial, which contained three replicates of each treatment, was laid out in an incomplete Latin square (7 rows of 3 plots), with a complete replicate in each column of seven plots, giving a total of 21 main plots and 42 sub-plots. Trial management and arable cropping rotation details can be found in Fraser *et al*. (2010).

*Earthworm extraction*

Earthworm abundance was monitored on an annual basis from 2000-2011 (except in 2007 when no sampling occurred) in late winter/ early spring - which is when the size of New Zealand earthworm populations peak on an annual basis (Springett 1992; Fraser *et al*. 2012). Two large soil samples (25 cm x 25 cm x 25 cm) were collected from each plot, with each sample stratified into separate depths corresponding to 0-7.5 cm, 7.5-15 cm and 15-25 cm. Earthworms were hand-sorted (Gilyarov 1975) and species were weighed (following gut voidance in water overnight) and identified according to the key of Sims and Gerard (1985) in all years except 2003 and 2007 (in 2003 only total numbers were recorded).

*SOC sampling and analyses*

Soil samples collected in spring from each plot (seven replicate soil cores, 30 cm deep x 5 cm wide) from 0-7.5 cm, 7.5-15 cm and 15-25 cm depths. Subsamples from each depth were sieved (2mm), air dried, ground and oven-dried overnight at 60˚C. Samples were analysed for total organic C by the Dumas combustion method using a TruSpec CN analyser (Leco Australia Pty., NSW, Australia) operating at 950 degrees C.

*Statistics*

Total profile earthworm count data from 2000-07 under -CC were modelled by a generalised linear mixed model (GLMM) with a Poisson distribution and a log link following Schall (1991). Correlations between measurements in consecutive years were best fitted by an AR model of order 1 with variances varying with year. The remaining variance components were bound at zero. Mean yearly earthworm count and biomass data from 2000-11 under + CC were modelled by GLMM as above and Analysis of Variance (ANOVA) respectively. Correlations between treatments across random strata were assumed to be independent and consistent. Mean yearly earthworm counts and Soil Carbon % at increasing depths from 2002-07 under +/- CC were modelled by GLMM as above and normal distributions respectively. Correlations between measurements at increasing depths were held at identity due to problems with convergence. The fixed terms in all analyses were partitioned to allow required comparison of treatments across random strata as described in Fraser *et al*. (2010). All statistical analyses were carried out in GenStat version 14. Backtransformed estimated means from count data are presented with 95% confidence intervals. Variation associated with remaining estimated means are provided by the Least Significant Difference (LSD), $\alpha$=0.05.

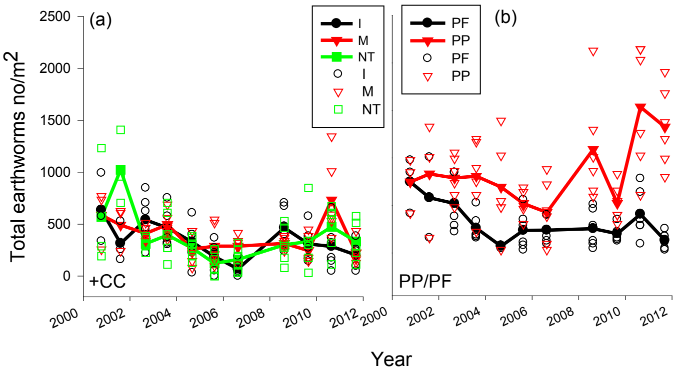
## Results and Discussion

*Earthworm abundance, biomass and community composition over time*

Although there are major changes in earthworm abundance over time ($p<0.001$), there are no clear differences in trends of earthworm abundance between the tillage treatments ($p=0.994$; Figure 1a) or cover crops ($p=0.345$). There is strong evidence that both the mean abundance ($p<0.001$) and mean biomass ($p<0.001$) of earthworms is far higher in PP than PF (Figures 1b, 2a and 2b) but it appears that abundance in PF is similar to that found in all three tilled treatments (Figure 2a). This indicates that in this soil type in the absence of a continual crop e.g. continuous pasture, earthworm numbers may be negatively impacted to a constant 'equilibrium ' level, regardless of the presence (using I, M or NT tillage) or absence (using chemically induced fallow) of annual crops. These results also suggest that the dominant herbicides used in the PF (glyphosate and tribenuron methyl) were no more deleterious for the earthworms than cultivation. Our findings are somewhat at variance with a review by Chan (2001) who found that in general conventional tillage decreases earthworm abundance compared to NT. However, Joschko *et al*. (2009) suggested that there is evidence to suggest that such ploughing associated declines tend to be more evident in heavier clay soils. The results from this study are similar to those found in the Palouse region of USA, where reductions in tillage did not significantly influence earthworm densities on silt loam soils (Umiker *et al*. 2009).

Similar to earthworm abundance, earthworm biomass (g/m$^2$) is consistent for the tillage and PF treatments (Figure 2b). Biomass in the PP is considerably higher (P<0.001) as would be expected given the significantly greater abundance of earthworms. At the beginning of the trial (in 2000) the dominant earthworm species in PP was *Aporrectodea (A.) caliginosa* (Savigny, 1826) (62%). Over the ensuing 11 years, species diversity declined, with *A. caliginosa* representing 82% of the PP population by 2011. In 2011 the proportion of *A. caliginosa* in I was 89% compared to 75% in NT. A notable difference in community composition was that there were less *Lumbricus (L.) rubellus* (Hoffmeister, 1843), an epigeic species that lives close to the surface, in I compared to NT (P=0.009). As *L. rubellus* play an important role in incorporation of organic matter into soil it is not surprising that they would be more dominant under NT than I. In agricultural ecosystems the loss in earthworm biodiversity might negatively affect soil functioning.

**Figure 1. Total earthworm populations (2000-2011) under (a) tillage treatments with cover crops and (b) permanent pasture (PP) and permanent fallow (PF). Lines join modelled estimates for each year (closed symbols). Scatter points represent observed data (open symbols).**




Organic matter input from long-term pasture is of the sort favourable for earthworm populations in terms of quantity, nutritional quality and also continuity throughout the year (Lagerlof *et al*. 2012). It is not surprising that initiating a cropping phase after a long-term pasture phase has resulted in a rapid decline to earthworm populations (Figure 2) and substantial losses of C. After the first 7 years of the trial there was an average loss 8.6 Mg C/ha across the tillage treatments compared to a loss of 17 Mg C/ha from the PF. It is difficult to explain the similarity in earthworm abundance between the PF and the tillage treatments, particularly given the large differences in C loss. It is possible that earthworm populations have declined to an equilibrium level which can be maintained despite further losses of C.

*Distribution of Earthworms and total C throughout the profile*

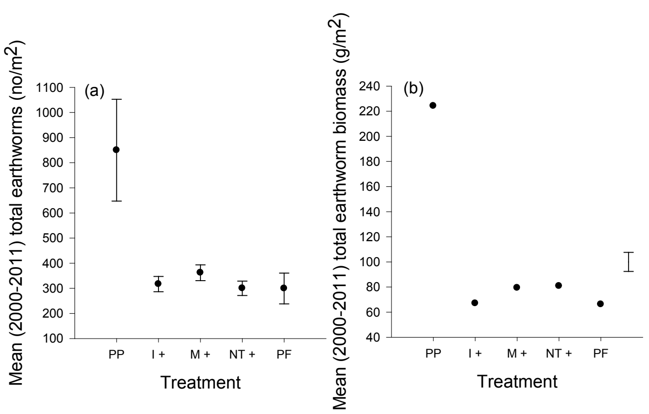



**Figure 2. Mean effect (2000-11) of tillage under + (CC), PF and PP on (a) total earthworm abundance, (bars represent back transformed 95% CI about estimated means), and (b) total earthworm biomass (bar represents 5% LSD with 8 df).**

Although there does not appear to be overall differences in earthworm abundance between tillage, CC and PF treatments, there is an indication that the distribution of earthworms throughout the profile is not consistent. This may be partially attributed to the distribution of total C throughout the profile. In general, there is a greater concentration of total C in the top (0-7.5cm) layer of the soil, and this decreases with depth (Figure 3b). The pattern is similar for PF, PG, M and NT. However, as might be expected under I, total C has been redistributed more uniformly throughout the profile. The pattern of earthworm distribution appears to closely follow this distribution of total

C, with one important exception - under the presence of +CC, i.e. a winter crop is sown after autumn tillage, the earthworms have returned to the surface, even though the soil carbon has remained distributed through the profile.

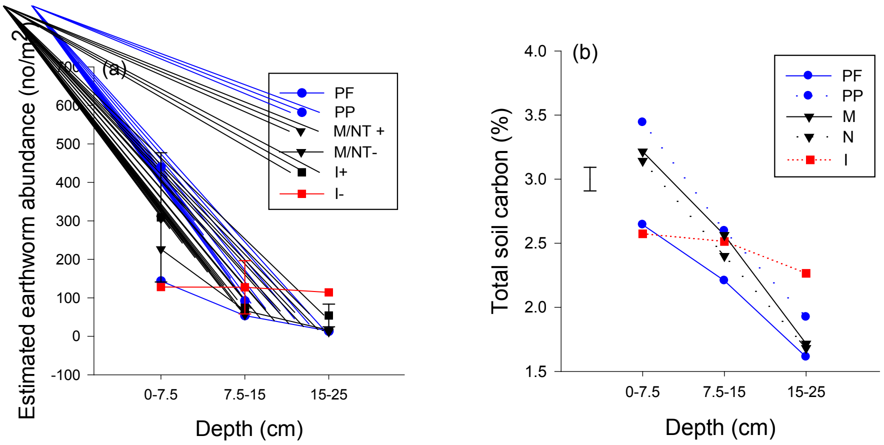



**Figure 3. Mean results (2002–07) by soil depth for (a) Earthworm population size (no/m$^2$) (bars represent back-transformed 95% confidence intervals on selected points), and (b) Total % soil organic carbon, (bar represents approximate average 5% LSD).**

It is important to observe that although there is greater accumulation of SOC near the soil surface under reduced and no tillage, similar to earthworm population trends, the overall storage of carbon remains unchanged. This is due to the mixing actions of ploughing and the greater accumulation of SOC at depth with intensive tillage. The uneven distribution of earthworms and SOC may have implications for enzymatic activities and microbial activity (Umiker, Johnson-Maynard *et al.* 2009). Further planned work investigating the C pools will determine how C informs earthworm distribution and help to better understand the functional implication of different tillage induced C *vs* earthworm distributions down the profile.

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# Microbial and chemical properties of soil changes along a soil development chronosequence near Lake Wellman, Southern Victoria Land, Antarctica

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The total ice-free area of Antarctica comprises about 0.32% of the continent. The soils are located on the continental coastline, particularly on the Antarctic Peninsula and along the Transantarctic Mountains. Little is known of soils distant from scientific bases. The objective of this study was to report on properties of soils from near Lake Wellman in the Darwin Mountains (S79º55'16.2" and E156º55'30.7") 370 km south west of Scott Base. Four pedons on each of four drift sheets were sampled for chemical and microbial analyses. The four drifts, Hatherton, Britannia, Danum, and Isca, ranged from early Holocene (10 ky) to mid-Quaternary (ca 900 ky). The landforms contain primarily high-centered polygons with windblown snow in the troughs. The soils are dominantly complexes of Typic Haplorthels and Typic Haploturbels. Soil properties of weathering stage, salt stage, depths of staining, and coherence increased with drift age. The soils were dry and alkaline with low levels of organic carbon, nitrogen, and phosphorus. Electrical conductivity was high. Soil microbial biomass was low with highest levels detected in Hatherton soils. Soil DNA was extracted and 16S rRNA gene clone libraries prepared from samples from below the desert pavement. Bacterial clones were assigned to the phyla *Deinococcus-Thermus, Actinobacteria,* and *Bacteroidetes. Deinococcus-Thermus* clones were identified as belonging to the genus *Truepera*. The actinobacterial clones were more diverse; however, 60% of the clones from the Hatherton soil belonged to the genus *Arthrobacter*. Culturable bacteria, including some that clustered with soil clones (e.g. *Arthrobacter* and *Adhaeribacter*), belonged to *Actinobacteria* and *Bacteroidetes*.

# The effect of temperature and organic carbon availability on the relative rates of microbial nitrogen immobilisation and nitrification in a semi-arid soil

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Conversion of nitrogen (N) to nitrate in agricultural soil exposes N to gaseous or leaching losses. Preventing these N losses may be achieved by managing soil N processes to increase microbial immobilisation and storage of N, and/or to decrease nitrification. An index to measure the relative importance of these two processes is the *N/I* (nitrification to immobilisation) ratio. This ratio has been used to describe N saturation in forests, and is correlated with leaching losses in temperate agricultural soils. Most crop production in Western Australia is carried out in the state's semi-arid region, characterised by cool, wet winters and very hot, dry summers. Soils in this region generally have coarse textures and very low organic matter contents. Using soil from a long-term cropping trial in the WA semi-arid region, we investigated how the *N/I* ratio is affected by organic carbon availability and soil temperatures from 5 to 50°C. We used field treatments with differing C availabilities (no till, burnt stubble, tilled soil and tilled soil with added organic matter), and a laboratory treatment of ± artificial root exudates as an additional C source. We used the $^{15}N$ isotopic pool dilution technique to measure gross N cycling rates.We hypothesised that with increasing C availability, heterotrophic immobilisers would be more active, competing more successfully for ammonium against nitrifiers and decreasing the *N/I* ratio, while with increasing temperature, heterotrophic microbes would become increasingly constrained by a lack of available C due to their increasingly rapid utilisation of energy sources, increasing the *N/I* ratio.

# Role of anecic earthworms on carbon incorporation to depth

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Anecic earthworms have the ability to incorporate carbon (C) from the surface to depth in the soil. These deep burrowing earthworms have a limited distribution throughout New Zealand, thus have potential to spread their populations to enhance C incorporation. This study aims to quantify the amount of C stored by the activities of anecics in the long-term.

In two soils where *Aporrectodea longa* was introduced in the1980's differences in soil C stocks were observed in 2011. In a Pallic soil, total C stored in the top 0.3 m was lower with anecic introduction (7.89 vs. 9.84 kg C/m2). In contrast, there was no significant influence of anecics on soil C in an Allophanic soil (13.67 vs. 14.28 kg C/m2).

The differing influence of anecics on soil C may be explained by differences in the earthworm community and their influence on the physical characteristics of the soil. In both soils anecic abundance was approximately 350 ind/m2 where introduced. In the Pallic soil, endogeic earthworms were abundant and there was little difference in total earthworm abundance (1500 vs. 1800 ind/m2). In the Allophanic soil, total earthworm abundance was much higher where anecics were introduced (640 vs. 60 ind/m2). The interaction between endogeic and anecic earthworms may be important in influencing soil C stocks, and this is being explored further in mesocosm studies. Earthworms may have had a greater influence on the soil physical structure in the denser Pallic soil in comparison to the Allophanic soil, contributing to a decline in C storage.

# Tuesday 4 December

## THE LIVING SOIL

*Presented Posters*

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# Laboratory and field evaluation of tillage disturbance effects on soil organic matter decomposition and associated physical properties

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## Abstract

Physical disturbance (e.g., by cultivation) may result in decreased soil organic matter (SOM) levels if mineralisation increases, e.g., because SOM in disrupted aggregates becomes more bio-available or because environmental constraints to mineralisation are alleviated. We compared mineralisation in intact soil cores (5 cm diam; 0-15 cm depth) and in soils that were physically disturbed by screening through a 4-mm sieve. In a 100-d study using soils from experimental plots (on a silt loam site at Lincoln, Canterbury, New Zealand) representing different management histories (arable cropping using either intensive cultivation, minimum tillage, or no tillage; long-term pasture; and chemical fallow) mineralisation (at 20$^{o}$C, -10 kPa) of C and N was strongly influenced by management history, but there was no difference ($P > 0.05$) between intact and disturbed cores. In another experiment, comparisons of mineralisation in intact and disturbed (sieved < 4 mm) cores taken from 14 arable and pasture fields with either silt loam or clay loam soils also showed no effect of disturbance. Field monitoring at the Lincoln site indicated that differences in key environmental regulators of mineralisation (soil temperature, moisture) between intensive tillage and no tillage plots were relatively small and unlikely to be of biological significance. Even though soil $O_2$ concentrations tended to be slightly lower under no tillage (higher near-surface bulk density may have restricted gas exchange with the atmosphere), $O_2$ availability is unlikely to have restricted mineralisation in this treatment.

## Introduction

Physical disturbance of soil during tillage operations may promote decomposition of SOM and lead to a decline in soil C and N. A commonly invoked mechanism for increased decomposition is the exposure to microbial degradation of SOM that was protected within aggregates prior to disturbance (Beare *et al*. 1994; Balesdent 2000). Changes in the soil environment, particularly alteration of aeration, temperature and moisture status, may also play a role. For example, oxygen deficient zones within aggregates may limit decomposition and the influx of $O_2$ when large aggregates are disturbed may stimulate decomposition.

To understand the impact of physical disturbance on C and N mineralisation, it is important to separate the direct effects of disturbance (increased organic matter availability/accessibility) from indirect effects due to changes in environmental regulators of mineralisation (e.g., aeration, moisture, temperature). The objectives of this study were to quantify the direct effect of disturbance by measuring C and N mineralisation under moisture- and temperature-controlled conditions in the laboratory; and (2) under field conditions, assess the influence of tillage disturbance on the environmental regulators of SOM mineralisation.

## Materials and Methods

### *Laboratory assessments*

Intact soil cores were collected (2005) from five treatments in a field trial that was established (in 2000) on a Wakanui silt loam at Lincoln, Canterbury, New Zealand to examine effects of tillage intensity on SOM following cultivation of long term pasture (crops grown in 2000-2005 period were: barley-wheat-pea-barley-pea; all irrigated). The cores represented three tillage treatments:

(1) Intensive tillage: mouldboard ploughed to ~20 cm, followed by secondary cultivation;

(2) Minimum tillage: top 10 cm cultivated using a spring tined implement, followed by secondary cultivation;

(3) No-tillage: seeds direct drilled.

Cores were also taken from plots representing the original ryegrass-clover pasture and from plots that were maintained fallow (plant-free using herbicides, i.e., not disturbed physically) since the start of the trial. In all, 15 plots (5 treatments x 3 replicates) were sampled.

The cores were extracted using a 5-cm diameter stainless steel corer and carefully transferred to 20-cm long PVC sleeves (each sleeve was cut lengthwise so that it could be expanded to avoid any disturbance when transferring the core to the sleeve). The sleeves were secured around the cores using cable ties and material deeper than 15 cm was trimmed off using a sharp blade. In the laboratory, the cores were randomly separated into two groups. One set remained intact while the other set was physically disturbed by passing the soil through a 4-mm sieve. A small sub-sample of the sieved soil (~ 30 g moist soil per core) was retained for determination of moisture content, mineral N, and total N and C. The remainder of the soil, including any material collected on the sieve, was returned to the cores, which were repacked to the original bulk density. Deionised water was added to adjust soil water potential to –10 kPa.

The soils were incubated in 5.5 L air-tight plastic containers (3 cores per container) at 20°C. Samples of headspace air were periodically removed (total of 27 samplings during the 100-day incubation) with a syringe and analysed for $CO_2$ using an infra-red gas analyser (LI-COR, Lincoln, Nebraska). At the end of the incubation (day 100), the soils were sieved (<4 mm), and subsamples were extracted with 2 *M* KCl for determination of ammonium- and nitrate-N (Keeney and Bremner 1982). Net N mineralised over the 100 day period was estimated by subtracting initial mineral N (measured on the samples taken from the disturbed cores just before the incubation) from that measured at the end of the incubation.

In a second laboratory study, using the disturbance procedure described above, we compared mineralisation in intact and disturbed cores (5 cm diam; 0-15 cm depth) from sites representing different textural classes (Mayfield silt loam vs Waterton clay loam) and land uses (no tillage arable cropping vs permanent pasture). A total of 14 fields were sampled (i.e., 3 arable and 4 pasture fields on each of two soil types) on commercial farms. The Mayfield sites were on adjacent arable and pastoral (sheep/beef) farms near Darfield, Canterbury while the Waterton sites were on adjacent arable and pastoral farms near Leeston, Canterbury. Carbon and N mineralisation was measured in a 68 d incubation at 20°C and -10 kPa water potential.

*Field measurements at Lincoln trial site*

At the Lincoln trial site, high-resolution measurements of soil water, $O_2$ concentration, and temperature were made over the course of a growing season (2011-12) in adjacent plots with contrasting levels of tillage disturbance (i.e., intensive tillage and no tillage plots). Sensors [i.e., thermocouples (6 per plot), time domain reflectometry probes (3 per plot), and galvanic $O_2$ sensors (3 per plot) (KE-25, Figaro, Japan] were placed at a depth of 6 cm depth. When the crop was present, measurements were made in interrow positions. Measurement frequency was 60 sec between consecutive readings (hourly average values presented here).

**Results and Discussion**

In the Lincoln tillage trial, after five years of arable cropping (2000-05), there was a significant decrease in soil C concentration (0-15 cm layer) compared with pasture soil; mean C concentration was 25 g kg$^{-1}$ in the arable systems compared with 30 g kg$^{-1}$ under pasture (Table 1). Differences between the three tillage treatments were relatively small. The chemical fallow, which was not physically disturbed, had lowest soil C concentration (though the differences between fallow and either intensive or no-tillage were not significant). The total C stock to 15 cm was about 10 t ha$^{-1}$ less in the minimum and no-tillage systems than in the pasture (Table 1). The intensively cultivated system has the smallest quantity of C in the top 15 cm; however, as intensive cultivation mixes soil to a depth of ~20 cm, the 15-20 cm layer of this treatment will have higher concentrations of C than the other tillage treatments (Fraser *et al*. 2010).

The physical disturbance treatment imposed in the laboratory (< 4 mm sieving) did not affect the amount of C mineralized in 100 d (Table 2). Effects of exposure of protected organic matter (due to aggregate disruption) on C mineralisation would be expected to be most apparent in the immediate aftermath of disturbance. Overall, disturbance did not significantly affect $CO_2$ production in the 14 d period after applying the disturbance treatment (Table 2). There was evidence of a disturbance x management history interaction (significant at $P = 0.075$) for C mineralisation in the first 14 d of incubation. This arose because, in minimum tillage soil, there was higher mineralisation (by 20%) in the disturbed treatment, whereas, in the no tillage soil, mineralisation was higher (by 12%) in undisturbed cores (Table 2). This tillage x management interaction was only apparent early in the incubation and had disappeared by day 30.

**Table 1: Soil C and N levels (0-15 cm) under different treatments in Lincoln Tillage Trial in 2005**

| Management history | Total C | Total N | Bulk density | C stock | N stock |
|---|---|---|---|---|---|
| | g kg$^{-1}$ | | g cm$^{-3}$ | t ha$^{-1}$ | |
| Pasture | 30.3 | 2.7 | 1.33 | 60.3 | 5.43 |
| Fallow | 23.4 | 2.2 | 1.48 | 52.0 | 4.96 |
| Intensive tillage | 24.1 | 2.2 | 1.28 | 46.1 | 4.21 |
| Minimum tillage | 26.4 | 2.4 | 1.28 | 50.7 | 4.61 |
| No tillage | 24.9 | 2.3 | 1.34 | 50.1 | 4.59 |
| LSD (5%) 8 df | 1.8 | 0.2 | 0.04 | 3.1 | 0.35 |
| F-pr | <0.001 | 0.002 | <0.001 | <0.001 | <0.001 |

**Table 2: Effects of management history and laboratory disturbance on C and N mineralisation**

| Management history | C mineralised, days 0-14 | | C mineralised in 100 d | | N mineralised in 100 d | |
|---|---|---|---|---|---|---|
| | | | kg ha$^{-1}$ | | | |
| | Intact | Disturbed | Intact | Disturbed | Intact | Disturbed |
| Pasture | 736 | 740 | 3037 | 3191 | 200 | 186 |
| Fallow | 138 | 153 | 861 | 849 | 72 | 71 |
| Intensive tillage | 212 | 197 | 1048 | 1053 | 87 | 79 |
| Minimum tillage | 224 | 269 | 1133 | 1292 | 89 | 99 |
| No tillage | 244 | 219 | 1227 | 1119 | 107 | 90 |
| LSR (5%) | 1.27 | | 1.21 | | 1.36 | |

LSR = Least significant ratio. This is the smallest ratio between two means (larger mean/smaller mean) for the means to be significantly different at the 5% probability level (8 df).

In the second study, where we imposed the laboratory disturbance treatment on soils with different land use histories and textures, there was no difference ($P > 0.05$) in C or N mineralisation between the intact and disturbed treatments (Figure 1). However, C mineralisation was influenced by soil type (higher in clay vs silt loams) and prior land use (pasture > arable soil).

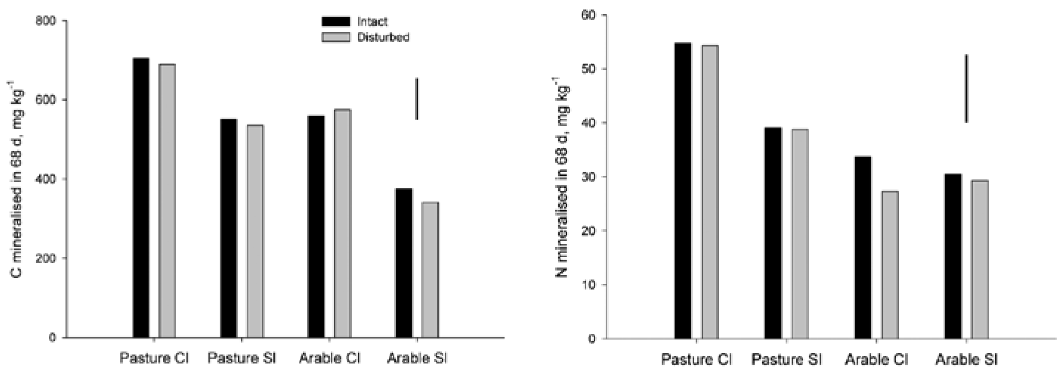


**Figure 1. Effect of physical disturbance on mineralisation of C and N in soils representing different textures ( cl = clay loam; sl = silt loam) and land uses (pasture or arable cropping with no tillage). Error bars are LSDs (5%).**

Field monitoring at the Lincoln site showed that differences in soil temperature and moisture between intensive tillage and no tillage plots were relatively small (Figure 2). Soil $O_2$ concentrations showed a strong diurnal pattern, associated mainly with changes in temperature, i.e., the minimum $O_2$ concentration occurred close to the daily temperature maximum. This association between soil $O_2$ and temperature can presumably be explained by increases in biological activity and associated $O_2$ consumption at higher temperatures. Concentrations of $O_2$ were slightly lower under no tillage, possibly due to decreased gas exchange with the atmosphere associated with higher near-surface bulk density in no-tillage vs. intensive tillage (Fraser *et al*. 2010). The effect of reduced diffusion on soil $O_2$ was most apparent when temporary water saturation (due to rainfall event following an irrigation in late December 2011) resulted in a rapid, but brief, decline in $O_2$ under no tillage (Figure 2). Overall, at the concentrations recorded in this study (mean of 0.185 m$^3$ m$^{-3}$), $O_2$ availability is unlikely to have restricted SOM mineralisation under no tillage.

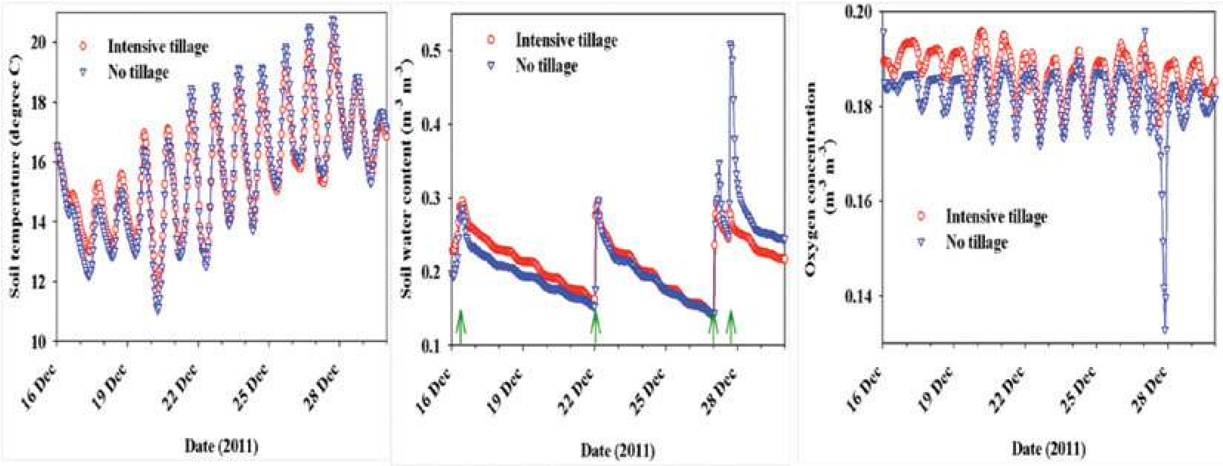


**Figure 2. Temporal patterns of soil temperature, water content, and $O_2$ concentration in intensive and no tillage plots during a selected measurement interval in late December 2011. Arrows indicate water inputs (rainfall or irrigation)**

## Conclusions

The disturbance treatment imposed in the laboratory study (< 4 mm sieving) would be regarded as severe compared to that caused by field cultivation practices. Even so, there was no enhancement of C or N mineralisation when soils were disturbed in this way, suggesting that quite severe physical disturbance would be needed to impair the protective action of aggregates for SOM. The small differences in soil temperature, water content, and $O_2$ availability between intensive tillage and no tillage plots would have minimal biological effect. The decline in SOM following cultivation of pasture may be due to decreased inputs of organic matter (roots and above-ground residues) rather than to enhanced decomposition resulting from tillage disturbance.

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## Can *Eucalyptus obliqua* seedlings grow well in forest soils not subjected to fire?

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Eucalypt seedling establishment and growth in native temperate forest soils is vigorous after high intensity burns which remove the organic soil layer, leaving an ash bed. Beneficial effects of burning on the mineral soil including fertilization, structural changes, and inhibition of deleterious microorganisms might explain this "ash bed effect." Notwithstanding good seedling performance in ash beds, ecosystem effects (e.g. carbon loss, smoke pollution) of fire may be undesirable. So, we investigated the growth of *Eucalyptus obliqua* seedlings in a pot-experiment using temperate eucalypt forest organic and mineral soils that were air-dried but not burnt. The organic soil was fumigated with methyl-bromide gas or not to investigate the effects of soil biota including mycorrhizas, and was fertilised or not with chelated iron followed by soluble phosphorus. Chelated iron was intended to preempt "Mundulla Yellows" symptoms, but unexpectedly exacerbated phophorus deficiency. The mineral soil was neither fumigated nor fertilised. Although fumigation of organic soil diminished ectomycorrhizas at 4.5 months, after eleven months mycorrhiza frequencies did not differ among organic soil treatments. Ectomycorrhizas were more frequent, however, in mineral than in organic soil (95% verus 71% root tips colonized). Nevertheless, mean seedling aboveground dry weight was 2.5-fold greater in organic than in mineral soil. The fertilisation regime improved aboveground dry weight 1.5-fold in ambient, organic soil, but diminished it by 0.25 in fumigated, organic soil. We conclude that in the absence of competing vegetation, *E. obliqua* seedlings that form abundant ectomycorrhizas in ambient organic soils substantially can outperform those in non-burnt mineral soil.

# Carbon to nitrogen ratio of organic matter inputs influences soil microbial population dynamics in relation to microbial release of fixed phosphorus

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Australian grain producers apply $1 billion worth of phosphorus (P) fertilisers each year, but only 50% is taken up by plants. Much of the remaining fertiliser P becomes fixed in soil and the P 'bank' in Australian arable soils is estimated to be worth $10 billion, or 100 kg P/ha of arable land. This experiment aimed to evaluate the potential of carbon (C) and nitrogen (N) availability to influence microbial release of fixed phosphorus in soil. We tested the hypotheses that (1) organic matter inputs with low C:N increased the microbial biomass until it exhausted the supply of easily available phosphorus forcing it to access fixed phosphorus in soil and (2) organic matter inputs with high C:N supported a smaller but more diverse microbial community with more strategies to access fixed phosphorus. To test our hypotheses we added simple organic compounds with different C:N ratios to model soils that were representative of both P-fixing and non P-fixing soil rhizosphere conditions. We also incubated with and without rock phosphate as a strategy to force the microbial populations to assess P that might not be readily available. In addition to measurements of P pools and P enzymatic activities we assessed microbial population size of both bacteria and fungi as well as components of microbial diversity to assess the effect of variation in C:N on the soil microbial populations.

# The effect of soil physical properties on $N_2O$ emissions from a dairy grazed pasture in South Otago, New Zealand

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New Zealand employs a country-specific emission factor (EF) of 1% for nitrous oxide ($N_2O$) emissions from excreta N deposited by grazing animals onto pasture (EF3). We measured N2O emissions from dairy cow urine patches on a poorly drained soil (mottled pallic silt loam) for 5 - 6 months for 3 consecutive years (2009 - 2011) within the same paddock (2.17 ha) in South Otago. Calculated EF3 for each trial year were 1.2, 3.7 and 5.5% in 2009, 2010 and 2011, respectively. Our hypothesis was that increased earthworm activity in the 2011 trial area resulted in significantly higher ($P < 0.001$) $N_2O$ emissions in 2011. We measured earthworm, climatic, soil physical and chemical factors in each trial area. An ANOVA found no differences between all factors in the 2009 and 2010 sites. However the trial area used in 2011 had been excluded from grazing for 1 yr and showed significant signs of soil recovery following cultivation in 2009; microporosity and earthworm population were significantly higher ($P < 0.01$) compared to the 2009 and 2010 results. Rainfall was intermediate in 2011 compared to 2009 and 2010, however changes in soil structure likely extended the period of optimum conditions for denitrification activity. We therefore conclude that soil recovery following cultivation of a pallic silt loam and increased earthworm activity due to stock/traffic exclusion in 2011 increased $N_2O$ emissions on a poorly drained dairy pasture in South Otago, NZ.

Keywords

Nitrous oxide emissions, soil structure, pore size distribution, earthworms, soil recovery, Pallic soil, New Zealand

# Temporal variation and effect of disturbance on microbial community structure in a pasture soil

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Microbial fingerprinting techniques are increasingly being used to characterise microbial communities.

Ordination and multivariate distance measures of these data have often been used to test for treatment effects or spatial variation in microbial community structure, but fewer studies have investigated temporal variation on a seasonal or longer basis. Microbial community structure in well-drained pasture topsoil ($n$ = 5) where soil had been tilled to a 15 cm depth and replanted, was compared with that in undisturbed pasture plots in a randomised complete block design. Microbial community composition (by T-RFLP) was measured quarterly over a 2 yr period. Fungal, archaeal, and particularly bacterial communities exhibited temporal shifts in structure over the study though the degree of variation by date and/or season differed substantially. Date of sampling accounted for about 46% of variance in the bacterial community ordination but only 18% in the fungal and 11% in the archaeal communities. The effect of the disturbance differed across communities. Over all dates there was a highly significant ($P = 0.002$) change in the fungal community, marginally significant change in the archaea ($P = 0.09$), and no detectable change in bacteria ($P = 0.44$) from disturbance. Even in the fungal community, however, the variation explained by treatment (3.5%) was smaller than that explained by sampling date. This study illustrates that temporal variation in ordination of microbial community data can be significant, particularly for the bacterial community where temporal variation was much greater than the fungal and archaeal communities.

# Tuesday 4 December

## SOIL CARBON AND CLIMATE CHANGE (SEQUESTRATION – ACCOUNTING)

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# Variations in the measurement of carbon sequestration due to core diameter and carbon density calculation: Do they make a difference?

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## Abstract

Carbon sequestration, reported as gains in carbon stocks, requires measurement of carbon concentration and bulk density of soil. Nineteen perennial pasture sites were sampled in the Boorowa region, south-eastern NSW to compare: i) the influence of core diameter size (155 mm *vs* 75 mm *vs* 40 mm) on bulk density and ii) carbon density (Mg C $ha^{-1}$ to 0.30 m) calculation (fixed depth *vs* equivalent soil mass) on total soil carbon stock. Bulk density was significantly different ($P$ <0.05) with core diameter size for all soil layers to 0.30 m with the exception of the 0.05 to 0.10 m soil layer. However, due to the variability in carbon concentration in soil there was no significant difference in carbon stocks calculated using either the fixed depth or equivalent soil mass carbon density values regardless of core diameter size. The mean carbon stock (Mg C $ha^{-1}$ to 0.30 m) calculated for the fixed depth and equivalent soil mass carbon density values using the 155 mm, 75 mm, 40 mm diameter cores was 50.9 (7.4 sd) *vs* 49.9 (7.3 sd) 53.8 (8.3 sd) *vs* 53.1 (8.2 sd), and 50.9 (8.4 sd) and 49.9 (8.0 sd), respectively. Based on these findings, the diameter of the cores used for bulk density measurements for carbon stock calculation should be selected based on operational ease and sampling efficiency rather than notions of precision of carbon stock reporting.

**Key words**
Bulk density, carbon stock, fixed depth, equivalent mass, carbon concentration

## Introduction

In Australia, trading carbon (C) that is stored in soil is increasingly being promoted as a new source of revenue for landholders. The mass of C in soil per hectare, also known as the "carbon stock", is a product of the C concentration, bulk density (BD) and the depth of soil sampled (Lal *et al.* 2001). To reliably account for changes in C stock in soil and to minimise error in C stock calculations it is essential to have accurate measurements of both C concentration and bulk density (Lee *et al.* 2009, Wilson *et al.* 2010).

There are two main ways to calculate the C stock in soil: 1) fixed depth carbon density (FDCD) where the C concentration is multiplied by the bulk density to a fixed depth and 2) equivalent soil mass carbon density (ESMCD) where carbon stock calculations are made using carbon concentration and bulk density over a varying soil thickness equivalent to a mass of soil from a base or reference site (Holmes *et al.* 2012, Lee *et al.* 2009, Wilson *et al.* 2010). The FDCD calculation is the most common calculation however, it can over- or under-estimate changes in carbon stocks which may result from changes in land use and management (Lee *et al.* 2009, Wilson *et al.* 2010). For example, management such as the establishment of perennial pasture may increase the concentration of C in soil thereby decreasing the density of soil (Malamoud *et al.* 2009, Rawls *et al.* 2003). Therefore the FDCD calculation may show no overall increase in carbon stock, where in reality the carbon stock has increased, but so to has the relative depth of soil. The ESMCD calculation accounts for changes and variations in bulk density associated with C gains and provides a more accurate measurement of changes in C stock (Lee *et al.* 2009, Schrumpf *et al.* 2011).

In Australia, the current convention for monitoring C in soil recommends collecting a minimum of three bulk density measurements using a minimum core diameter of 40 mm from for each soil layer (0-0.10, 0.10-0.20 and 0.20-0.30 m) per site (Sanderman *et al.* 2011). This paper compares the effect of three core diameter sizes on bulk density measurements. We explore the impact of bulk density core size on carbon stock calculations using the FDCD and ESMCD methods.

## Methods

*Site location and sampling*

Nineteen sites were sampled in the Boorowa region, south-eastern NSW. Sites included native and introduced perennial pastures. Native pastures typically contained danthonia (*Austrodanthonia*), microlaena (*Microlaena stipoides*) and stipa (*Austrostipa scabra*). Introduced pastures were typically comprised of phalaris (*Phalaris aquatica L.*) and cocksfoot (*Dactylis glomerata L.).* Both pasture types included exotic annual species such as subterranean clover (*Trifolium subterraneum*).

The soil profiles of sampled sites site were classified as either Kurosols or Chromosols (Isbell 2002). The ranges of selected chemical properties of sampled sites are reported in Table 1.

Sites were sampled in autumn 2010 to 0.30 m at 0-0.05, 0.05-0.10, 0.10-0.20, and 0.20-0.30 m depth intervals according to SCRP protocols (Sanderman *et al.* 2011). Cores were collected to at least 0.50 m to ensure a good quality core. Soil was sampled using 3 core diameters; 155 mm ($n$ = 4), 75 mm ($n$ = 4) and 40 mm ($n$ = 4). Cores were located independently across the paddock area. A Proline soil auger was used to collect the 155 mm diameter cores while a hydraulic soil corer was used for the 75 and 40 mm diameter cores.

**Table 1: Soil chemical properties (0 to 0.30 m): The range for pH (CaCl2), Colwell phosphorus (P), sulfur (S, KCl 40), CEC (cmol+/kg), Total Carbon (TC, g/100g) and Total Nitrogen (TN, g/100g). Phosphorus and Sulfur measured on the surface 0.20 m only.**

| Depth (m) | pH (1:5 $CaCl_2$) | P (mg/kg) | S (mg/kg) | CEC (cmol+/kg) | TC (g/100g) | TN (g/100g) |
|---|---|---|---|---|---|---|
| 0 - 0.05 | 4.23 - 6.17 | 9 - 61 | 13 - 48 | 4.27 - 13.15 | 1.54 - 3.72 | 0.11 - 0.31 |
| 0.05 - 0.10 | 4.07 - 5.37 | 6 - 21 | 10 - 26 | 2.59 - 6.71 | 0.89 - 1.67 | 0.06 - 0.14 |
| 0.10 - 0.20 | 4.08 - 5.45 | 4 - 12 | 7 - 25 | 1.91 - 5.87 | 0.46 - 0.89 | 0.02 - 0.07 |
| 0.20 - 0.30 | 4.17 - 5.43 | … | … | 2.19 - 7.80 | 0.25 - 0.64 | 0.01 - 0.06 |

*Analytical methods*

Bulk density was determined for each depth interval of the twelve cores collected at each site as described by Dane and Topp (2002). Four measurements per diameter were averaged according to the Australian soil monitoring protocol (McKenzie *et al.* 2002). Results were calculated as BD in $Mg/m^3$ (equivalent to $g/cm^3$) on an oven-dry basis to the nearest 0.01 $Mg/m^3$.

Chemical analyses were conducted on composite samples for each depth increment Soil samples were prepared for chemical analysis as described by Rayment and Higginson (1992; Method 1B1). All samples were tested for carbonates using HCl and observing the degree of effervescence (Rayment and Lyons, 2011; Method 19D1). No samples required pre-treatment for inorganic C. Total Carbon (g/100g) was determined on all samples using a LECO (CNS 2000) combustion furnace (Merry and Spouncer, 1988, Rayment and Higginson, 1992; Method 6B3).

Results for this paper are reported as C stock in Mg C $ha^{-1}$ calculated by:

1. Fixed depth carbon density
   FDCD (Mg C $ha^{-1}$) to 0.30 m = carbon concentration (g/100g) x bulk density ($g/cm^3$) x depth (cm)
2. Equivalent soil mass carbon density (Baldock 2012)
   ESMCD (Mg C $ha^{-1}$) to 0.30 m = $a + (((b - c) \div d) \text{ x } e)$

Where; $a$= FDCD (Mg C $ha^{-1}$) to 0.20 m, $b$= $10^{th}$ percentile of soil mass per hectare 0-0.30 m of all sites in the comparison (155 mm core = 4818 Mg $ha^{-1}$, 75 mm core = 5278 Mg $ha^{-1}$ and 40 mm = 4987 Mg $ha^{-1}$), $c$= soil mass per hectare 0-0.20 m, $d$= soil mass 0.20-0.30 m, and $e$= FDCD (Mg C $ha^{-1}$) 0.20-0.30 m.

*Statistical analysis*

Statistical analyses were performed using GENSTAT v.8 (VSN International Ltd, UK) software. Differences at $P$ = 0.05 were assessed using ANOVA for bulk density using the three different core diameter sizes and similarly for 0-0.30 m (all soil layers) for FDCD and ESMCD values. Differences between sites are interpreted as random effects and considered to be blocking elements. Core diameter was considered a fixed effect and therefore differences in bulk density between cores of different diameter are interpreted as being due to the choice of the core size. Interaction of sites (random) and diameter (fixed) is a random effect and forms the denominator for testing the F statistic for the presence of diameter effects.

**Results and Discussion**

Core diameter had a significant effect on bulk density for all soil layers with the exception of the 0.05-0.10 m soil layer (Table 2 and 3). In the 0-0.05 m soil layer, there was a difference between the 40 and 75 mm diameter core. In the 0.10-0.20 m soil layer, all three diameters are different from each other (155 < 40 < 75 mm). In the 0.20-0.30 m soil layer, the 40 and 155 mm diameter core are different from one another. There was also a significant interaction between diameter and site indicating that the effect of core diameter on bulk density varied with site and soil layer (Table 3).

The variance in bulk density for each core diameter and soil layer at each site was calculated (data not presented). Samples collected with the 155 mm diameter core displayed the greatest variance in all except the 0.10-0.20 m soil layer. This may reflect the difficulty in collecting bulk density samples using large diameter cores with the Proline soil auger, particularly in surface soil and where there is an abrupt soil textural change in the profile (i.e. 0.20-0.30 m). Samples collected with the 40 mm core displayed the least variance for all except the 0.10-0.20 m soil layer. We suggest that this was due to the impacts of the hydraulic corer on the poorly structured A2 horizon with such a small (40 mm) diameter core.

Collection of 40 mm and 75 mm diameter bulk density samples using a hydraulic soil corer is more cost effective, time efficient, and with the evidence provided in this study, more reproducible than wider diameter cores collected using the Proline soil auger. Further research is required to measure the impacts of soil moisture at the time of sampling on bulk density measurements using different core sizes.

**Table 2: Mean Total Carbon (g/100g) and bulk density (g/cm$^3$) for soil layers from all sites. Values in brackets are standard deviation. Probability (F pr.) and least significant difference (l.s.d.) presented.**

| | | Core Diameter (mm) | | | | |
|---|---|---|---|---|---|---|
| Soil depth (m) | TC (g/100g) | 155 BD (g/cm$^3$) | 75 BD(g/cm$^3$) | 40 BD(g/cm$^3$) | F pr. | l.s.d (5%) |
| 0-0.05 | 2.57 (0.53) | 1.13 | 1.16 | 1.08 | 0.07 | 0.07 |
| 0.05-0.10 | 1.17 (0.21) | 1.26 | 1.33 | 1.30 | 0.10 | 0.07 |
| 0.10-0.20 | 0.59 (0.09) | 1.33 | 1.47 | 1.41 | <0.001 | 0.06 |
| 0.20-0.30 | 0.39 (0.10) | 1.41 | 1.53 | 1.46 | 0.04 | 0.05 |

**Table 3. ANOVA of core diameter and site interactions on BD (g/cm$^3$) for soil layers with degrees of freedom (d.f.), sum of squares (s.s.), mean square (m.s.), variance ratio (v.r.) and probability (F pr.).**

| Depth (m) | Source of variation | d.f. | s.s. | m.s. | v.r. | F pr. |
|---|---|---|---|---|---|---|
| 0-0.05 | Site | 18 | 2.04 | 0.11 | 2.42 | |
| | Site.Diameter stratum - Diameter | 2 | 0.27 | 0.14 | 2.92 | 0.067 |
| | Residual | 36 | 1.69 | 0.05 | 2.01 | |
| 0.05-0.10 | Site stratum | 18 | 0.66 | 0.04 | 0.89 | |
| | Site.Diameter stratum - Diameter | 2 | 0.20 | 0.10 | 2.43 | 0.102 |
| | Residual | 36 | 1.49 | 0.04 | 2.05 | |
| 0.10-0.20 | Site stratum | 18 | 0.75 | 0.04 | 1.27 | |
| | Site.Diameter stratum - Diameter | 2 | 0.72 | 0.36 | 10.86 | <.001 |
| | Residual | 36 | 1.19 | 0.03 | 1.75 | |
| 0.20-0.30 | Site stratum | 18 | 1.90 | 0.11 | 4.68 | |
| | Site.Diameter stratum - Diameter | 2 | 0.16 | 0.08 | 3.46 | 0.042 |
| | Residual | 36 | 0.81 | 0.02 | 1.02 | |

There was no significant difference in C stocks calculated using either the FDCD or ESMCD values regardless of core size (Table 4). This finding reflects the greater variability in C concentration than bulk density measurement (Table 2). This agrees with the Holmes et al (2012) study. All sites sampled in this study were perennial pastures on Chromosols and Kurosols. It could be expected that greater differences would have been found on soil types exhibiting larger differences in profile properties and soils under more intensive management. It is also anticipated that the ESMCD calculation would perform more accurately and reliably when quantifying temporal variations in C stock under land management systems where bulk density may significantly change. Further research is required to determine the sensitivity of core diameter used to determine bulk density and/or the method used to determine C stock calculation (FDCD *vs* ESMCD) under different land use, for example cropping, or soils that are heavier in texture, for example Ferrosols.

**Table 4: Mean carbon density (Mg C ha-1 0-0.30 m) using FDCD and ESMCD values for different core diameter sizes with standard deviation in brackets. The difference in mean CD values is reported.**

| Carbon density calculation | Core Diameter (mm) | | |
|---|---|---|---|
| | 155 | 75 | 40 |
| FDCD | 50.9 (7.4) | 53.8 (8.3) | 50.9 (8.4) |
| EMCD | 49.9 (7.3) | 53.1 (8.2) | 49.9 (8.0) |
| Difference between FDCD and EMCD | + 1.00 | + 0.71 | +0.96 |

## Conclusion

The diameter of the core used to take soil samples for the determination of soil C stock does influence the variability of bulk density measurements and this effect changed with soil depth. Despite these observations, the reported influence of core diameter on variability in bulk density measurements did not lead to significant differences in the C stocks of the soils calculated as either FDCD or ESMCD. Therefore it is apparent that the variability of soil C has a greater influence on the reported measures of C stocks than the diameter of the core used to take the bulk density samples. Based on these findings the diameter of the cores used for bulk density measurements in perennial pastures for carbon stock calculations should be selected based on operational ease and sampling efficiency rather than notions of precision of carbon stock reporting.

## Acknowledgements

This research is part of a PhD through the EH Graham Centre of Agricultural Innovation and the Future Farm Industries CRC. Funding for this research was provided by NSW DPI. The authors would like to acknowledge Robert Smith, NSW DPI for technical assistance and Terry Koen, Office of Environment and Heritage for assistance in statistical analysis.

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# Australian carbon dust emission: A carbon accounting omission?

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Erosion preferentially removes the finest carbon- and nutrient-rich soil fractions, and consequently its role may be significant within terrestrial carbon (C) cycles. However, the impacts of wind erosion on soil organic carbon redistribution are not considered in most SOC models, or within the Australian national carbon accounting scheme. Although SOC can be redistributed locally by water and wind erosion, dust emission can remove surface SOC from vast areas of inland and agricultural areas of Australia and transport it quickly offshore; representing a net loss of SOC from terrestrial systems. Estimates of the carbon dust emission magnitude require information on the spatial and temporal variation of SOC enrichment in dust emissions (*P*). We developed a process-based approximation of *P* within the Computational Environmental Management System (CEMSYS v5) national wind erosion model. It enabled the prediction of carbon dust emissions at a 50 km spatial resolution across Australia every month from 2000–2010. Carbon dust emissions were summed for all months in the study period and across Australia to generate a time series of national carbon dust emissions. The magnitude, frequency and spatial variation of carbon dust emissions and their implications for Australian national carbon accounts will be discussed.

# Modelling soil carbon sequestration potential in Australian wheat region

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### Abstract

In Australia, conversion of native land for agricultural production has led to significant decrease in soil organic carbon (SOC), and reversing the trend of SOC decline represents a great potential for agricultural soils to sequester carbon. This paper demonstrates a modelling approach to estimation of carbon sequestration potential in agricultural soils. We used the APSIM farming systems model, together with soil profile data from 557 sites, to simulate the SOC sequestration potential for an annual wheat system. The point simulation results were used to develop a meta-model to estimate regional SOC potential across the wheat growing region. The results show that an annual winter cropping system has the potential to raise SOC if it could be managed properly. Alternative rotation systems (e.g. with pastures) may lead to higher SOC sequestration potential. However, building extra carbon in soil needs extra nitrogen input. Economically viable farming and management systems need to be developed to increase both SOC and farm profitability.

**Key Words**

Soil organic carbon, decomposition, modelling, wheat, APSIM.

### Introduction

In Australia, conversion of native land for agricultural production has led to decreases in soil organic carbon (SOC) stocks in the order of 42~59% (Guo and Gifford, 2002). Due to the large amount of SOC, reversing or stabilizing the trend of SOC decline represents a great potential for agricultural soils to sequester carbon in mitigating greenhouse gas emissions. Adoption of conservation agricultural practices including enhanced rotation, fertilizer application and stubble retention generally resulted in a relative gain in SOC as compared to conventional management (Luo *et al*, 2010; Sanderman *et al*, 2010). However, in many cases, this relative gain may reflect a slowing rate of SOC loss, rather than additional SOC sequestration, because many soils are still in the SOC declining phase and have not reached a new equilibrium state (Sanderman and Baldock, 2010). The limited available experimental data present a quantitative assessment of the soil carbon sequestration potential and the measures and time required to achieve that potential.

The dynamics of soil organic carbon are controlled by the carbon input from primary production and output governed by SOC decomposition; both are controlled by climate, soil and management conditions. Under a given environment and management regime, a new SOC equilibrium is reached if the carbon input into soil balances the carbon loss by decomposition. This can take several decades to occur (Luo *et al.* 2010). Management strategies that are economically viable and result in a higher equilibrium SOC need to be developed for SOC sequestration purposes. However, evaluation of alternative strategies using an experimental approach is impractical due to the time and investment needed. A modelling approach is best suited for such purposes.

Previous studies showed that the APSIM farming systems model was able to capture the SOC dynamics as affected by management and environmental conditions in typical Australian dry land cropping systems (Luo *et al.* 2010; Huth *et al.* 2009). This paper aims to use APSIM to explore the SOC sequestration potential in the wheat growing region of Australia. We estimated the potential SOC level that could be achieved under the best possible management regime in an annual wheat system, and the time needed to achieve this SOC potential across the wheat regions.

### Methods

*Site simulations*

The APSIM model (Keating *et al*, 2003) version 7.3 was used, together with long-term daily weather records and soil profile data to simulate SOC dynamics. In total, profile data for 577 soils distributed across the Australian cereal-growing regions were obtained from the Agricultural Production Systems Research Unit (APSRU) reference sites (http://www.asris.csiro.au/themes/model.html#). Daily weather data (including daily rainfall, maximum and minimum temperature and radiation) at the nearest climate station to each of the 577 sites were obtained from the SILO Patched Point Dataset (http://www.longpaddock.qld.gov.au/silo/).

Wheat is the most dominant crop in Australia. To simplify the modelling, we attempt to assess the potential of C sequestration under a typical annual wheat system with the best possible management practices including optimal control of pest and diseases, optimal application of fertilizers (no nutrient deficiency) and 100% residue retention. APSIM was run at the 577 reference sites for 122 years from 1889 to 2010 to simulate the SOC change. A wheat crop was assumed to be sown every year depending on rainfall and soil water content, which varied for different regions. Different wheat cultivars were assigned according to sowing date and location: the earlier the sowing date, the later the maturity type of the wheat cultivar used. Crop residues (stem plus leaf) after harvest were retained in the system, and we assumed that 10% were left on the soil surface and the rest incorporated into the top 20 cm of soil.

*Regional simulations*

To get a regional map of SOC potential, a simple meta-model was developed based on the APSIM simulation results at all reference sites to calculate the SOC potential and change spatially. The meta-model links the final SOC content in soil to growing season rainfall (*P*), temperature (*T*), radiation (*R*), inert C content ($C_{inert}$) and plant available water capacity of soil (*PAWC*), and explained 84% of the variation in APSIM simulated SOC as influenced by spatial climate and soil variations. It has the form:

$$SOC = 70.44 + 1.22C_{inert} + 0.028P + 0.024PAWC - 2.21T - 0.0092R + \varepsilon.$$

Based on soil texture of the top soil (A horizon), we divided the 577 soils into five groups: sands (111 soils), sandy loams (73 soils), loams (115 soils), clay loams (148 soils) and clays (130 soils). For each soil group, a probability density function (PDF) was constructed for each of the soil properties included in the meta-model (i.e., *PAWC*, $C_{inert}$) using the soil data from the 577 reference sites.

For the whole wheat region, spatial soil texture data was collected from the Australian Soil Resource Information System (ASRIS) (http://www.asris.csiro.au/) at a resolution of 0.01°×0.01° (latitude × longitude). For each ASRIS pixel, soil texture was used to sample the constructed PDF of the corresponding soil group 2000 times to derive an ensemble of each of the soil variables included in the meta-model. Daily climate data from 1889 to 2010 at a spatial resolution of 0.05° were collected from the Bureau of Meteorology, Australia (http://www.bom.gov.au). The nearest climate data point (totally 7845 points) to each ASRIS pixel was used to calculate the climate variables (*P, T, R*) included in the meta-model. The meta-model, together with the climate data and the 2000 sets of soil variables, were used to predict the SOC in each pixel. The average SOC content and the 95% confidence intervals (CI) of the meta-modelled SOC were calculated. A 95% CI of SOC change was also estimated.

## Results

*SOC potential across the wheat region*

The initial SOC contents (0-30cm) used in the modelling varied significantly spatially (Fig. 1A). Assuming 122 years of continuous wheat farming with optimal management, the simulated mean SOC contents by the meta-model ranged from 17.6 to 109.2 t ha$^{-1}$ with a mean of 44.7 t ha$^{-1}$ (Fig. 1B). This mean SOC content showed apparent regional pattern – a decreasing trend from southwest to northeast in Western Australia and from southeast to northwest in eastern Australia. It tended to increase with rainfall, but to decrease with increasing temperature. It is emphasized that, there were great uncertainties in the spatial simulation results induced by uncertainties in soil properties, particularly for regions with lower SOC content. For most of the study area, the uncertainty of simulated SOC content ranged from 20% to 60% (Fig. 1D).

*SOC change from start to end of simulation*

Under the assumed wheat system and initial SOC contents used in the modelling, after 122 years, SOC in the surface 30 cm soil layer increased on average by 0 to 20 t ha$^{-1}$ in most of the study area, except for northern-most parts of Queensland and WA, where SOC showed decreases by 0–10 t $ha^{-1}$ (Fig. 1C). The greatest increases of > 20 t $ha^{-1}$ occurred at the southeast edge of the study area. The greatest uncertainty of SOC change was estimated in the northern-most parts of the study area in Queensland and WA. In those regions, soil could be either a C sink or source depending on soil properties. In most of the southern part of the study area soil was always a C sink i.e. sequestering C under the continuous wheat system. Among the 577 sites, SOC increased at 418 sites and decreased at 159 sites, with a regional average increase of 11.3 t $ha^{-1}$, representing around 20% increase as compared to the initial SOC level. (Fig. 2).

*Time needed to reach a new equilibrium SOC*

The time needed to reach the new equilibrium of SOC is related to both the initial carbon content and environmental condition. Among the 577 sites, 45 sites (7.8%) did not reach equilibrium at the end of the 122-year simulation. Two hundred and sixty sites (45.1%) needed more than 100 years and 194 sites (33.6%) needed more than 50 years to reach equilibrium (Fig 2). Sixty sites (10.4%) needed more than 10 years to reach equilibrium, while 18 sites (3.1%) reached equilibrium in the first decade of the simulation. Multi-regression analysis indicated that temperature and precipitation and their interaction had significant effects on the time needed to reach the new equilibrium, but temperature and precipitation together only explained 8% of the variation of time needed.

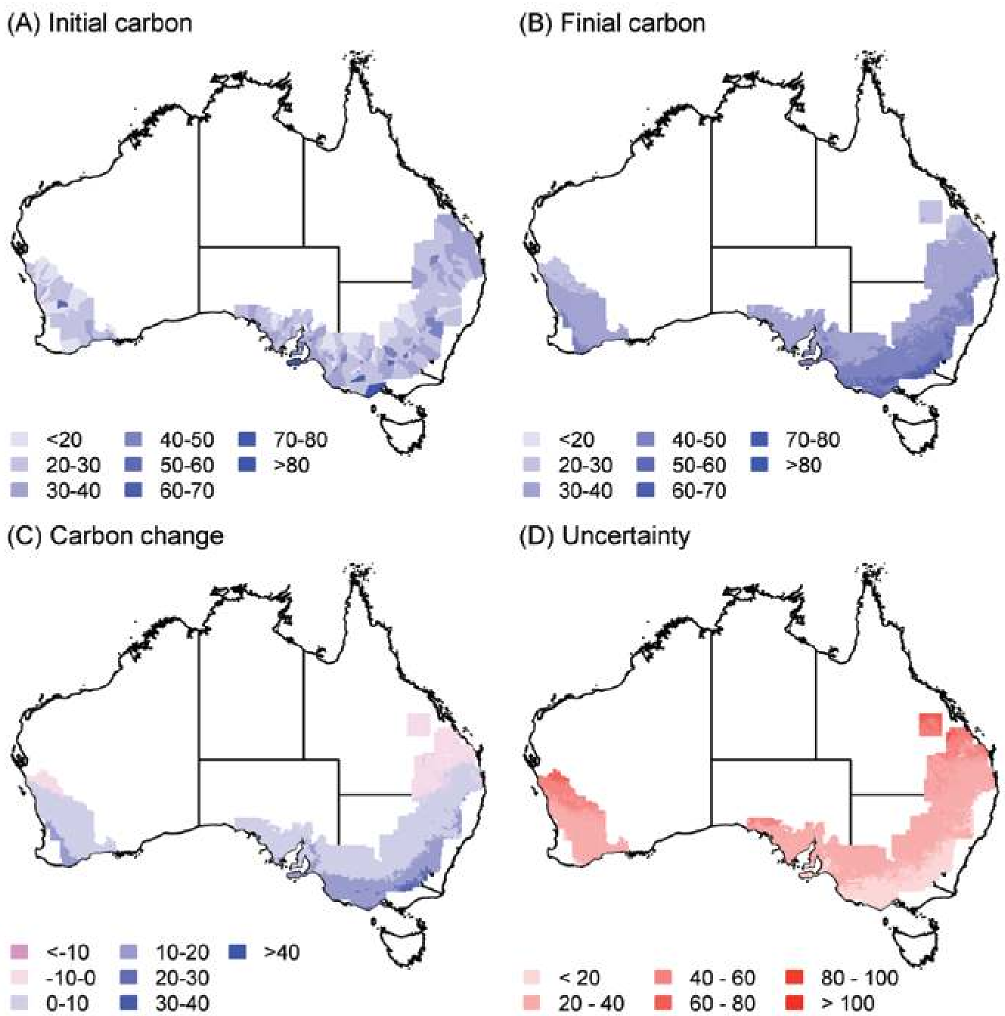


**Figure 1: The initial soil organic carbon contents in the 0-30cm soil layer (t ha-1) used in the modeling (A), the simulated contents of soil organic carbon (t/ha) at the end of 122 years of simulation (B), the absolute change of SOC in 122 years (C) and the uncertainty in the simulation results (%) (D) under a continuous wheat system**

## Discussion and Conclusion

This paper demonstrates a modelling approach to estimation of carbon sequestration potential in agricultural soils. The results presented are only preliminary simulation results, thus with significant uncertainties, particularly associated with the uncertainties in spatial soil information. In addition, the continuous wheat system and the optimal management strategies used in the simulation are simplifications of the farming systems in Australia. In reality, the farming systems are more complex, opportunity summer cropping in Queensland and crop-pasture rotations in other parts of the study region are common practices. For economic reasons, farmers normally do not target maximum yield, and may not adopt full residue retention as assumed in the simulations. These are the factors that need to be considered in future modelling. Nevertheless, the preliminary results allow exploration of some important features of carbon sequestration in agricultural soils.

The simulated SOC increase in most of the annual cropping area implies that an annual winter cropping system has the potential to raise SOC if it can be managed properly. Alternative rotation systems (e.g. with pastures) may lead to higher SOC sequestration potential. However, the long time needed for SOC to reach a new equilibrium implies that SOC sequestration is a slow process, and it may take 50 years to reach the SOC potential. Furthermore, due to the constrained C:N ratios of the biggest SOC pool of humus, building extra carbon in soil means extra nitrogen needs to be supplied as well. Based on a C:N ratio of 10 for humus, sequestering 10-20 t C/ha would need ~1 t/ha of nitrogen to be put into soil. This will have implications on both economic viability and future soil fertility.

It can be concluded that systems modelling is an effective approach to SOC sequestration research. There is potential to sequester carbon in Australian agricultural soils, but economically viable farming and management systems need to be developed to increase both SOC and farm profitability.

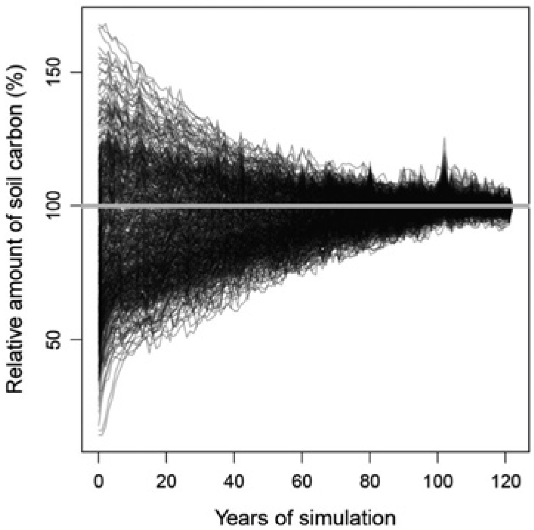


**Figure 2: The relative change of soil organic carbon in the top 30 cm soil in each year to that at the end of the simulation from 1889 to 2010 based on APSIM simulations at 577 sites in Australia**

## Acknowledgments

Founding supports from the Grain Research and Development Corporation (GRDC) and the Department of Agriculture, Fisheries and Forestry (DAFF) are gratefully acknowledged.

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# Simulation of soil organic carbon across Cox's Creek catchment in northern New South Wales using RothC: A first approximation

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## Abstract

The aim of this study is to test the prediction quality of RothC model when it is run spatially across a catchment using readily available spatial data. The study area is the Cox's Creek catchment in northern New South Wales. Soil carbon data was available for the 2000-2002 time period. Initial carbon pools were determined after running the RothC in equilibrium mode for 10,000 years under constant environmental conditions and interactively changing residual inputs considering three soil types under native vegetation. Then using these initial pools, land management information and observed weather data, simulation of soil organic carbon (SOC) was carried out across the catchment from 1970 to 2000. The greatest mean predicted SOC for year 2000 was recorded for pasture on Tenosols while the smallest was recorded for forest and pasture on Vertosols. Model validation was done based on the actual measurement done in 2000-2002. Overall predictions were poor (RMSE of 0.86 %). In addition Lin's concordance correlation coefficient was low between predicted SOC *vs.* measured SOC ($r_L$ = 0.18). While the results are poor they are important as they do illustrate the perils of running RothC and similar models with naive inputs. Further work is aimed at calibration of the model with the use of inverse modelling techniques and to identify sources of error.

## Keywords

Soil Carbon, Modelling, RothC, Spatial

## Introduction

Modelling and managing the variation of soil organic carbon (SOC) over the time and space is important due to its link to ecosystem health and as a way to offset green house gasses. However, collecting soil data over the time for monitoring purpose across the landscape is expensive in terms of time and cost. Therefore, scientific approaches have been developed to simulate the SOC changes over the time using mechanistic models. Currently there are more than 250 models used to simulate SOC worldwide. Manzoni and Porporato (2009) and Batlle-Aguilar *et al.,* (2011) classified these models based on their internal structure into four main classes *i.e.* (i) process oriented (multi) compartment models, (ii) organism oriented (food web) models, (iii) cohort models describing decomposition as a continuum, (iv) a combination of model type (i) and (ii).

One of the most widely used model is the Rothamsted carbon model (RothC) which is a process oriented (multi) compartment model (Coleman and Jenkinson, 1996). RothC is used to model the turnover of the SOC in non-waterlogged environment for the topsoil. The model is based on temperature, soil moisture content and plant inputs of carbon. Within the model SOC is split into four active fractions and one small inert organic matter (IOM) fraction. The active fractions are decomposable plant material (DPM), resistant plant material (RPM), microbial biomass (BIO), and humified organic matter (HUM). Each fraction decomposes by a first-order decay process with its own characteristic rate. The IOM fraction is considered to be resistant to decomposition. RothC generally runs on a monthly time step.

There are many applications of the RothC model in the spatial context. It has been embedded in some of the national level carbon accounting programmes in relation to United Nations Framework Convention on Climate Change (UNFCCC) and Kyoto Protocol. In Australia it has been incorporated into the FullCAM model as part of a national carbon accounting system (Richards and Evans, 2004). Falloon *et al.*, (2006) applied RothC in spatial context at a resolution of 1 km in the United Kingdom to assess changes in SOC with land use change. In many of these cases the model is run based on readily available soil, weather and land use data with minimal calibration. This is a concern as the model predictions can form the basis for government policy. While RothC has been extensively validated at smaller scales such as on plot or in farm scale (Skjemstad *et al.*, 2004), little published work has considered its accuracy when run spatially at a larger extent such as in catchment level.

Therefore, the aim of this research is to examine the prediction quality of RothC at the catchment scale when readily available spatial data is used as inputs and standard model parameters are used without calibration.

**Methods**

*Study area*

The study area, the Cox's Creek catchment is located in northern New South Wales (NSW) of Australia and covers approximately 1445 $km^2$. The elevation varies between 240 m and 635 m above sea level. The catchment has a mixture of land uses comprising irrigated agriculture (4%), dry land cropping (35%), pasture (38%) and forest (20%). The soils of the catchment vary from heavy-textured Vertosols along the Cox's Creek and its tributaries to sandy Tenosols towards western boundary of the catchment.

*Soil Data*

SOC data were obtained for 95 sites, out of which 81 sites were collected based on soil landscape mapping by the NSW government collected between April, 2000 and September, 2002. The remaining 14 sites were taken from the University of Sydney Cotton Catchment database collected in 2000. The NSW government SOC data, which were measured using Walkley and Black method, were corrected for total organic carbon using a conversion factor of 1.13 (Skjemstad *et al*., 2000). In this study the data for 0 - 0.3 m depth interval was considered. As the NSW government legacy data were based on horizon sampling, equal area splines were fitted to extract SOC, clay and sand fractions data at the pre-determined 0 - 0.3 m depth interval (Bishop *et al*., 1999). The bulk density was estimated using a pedo-transfer function (Tranter *et al*., 2007). The digital soil map of the study area produced by Nelson and Odeh (2009) was generalized into three soil groups as Vertosols, Tenosols and other soil types (Other)

*Climate data*

Interpolated monthly climatic grids for rainfall, monthly evaporation (Class A pan), maximum temperature, and minimum temperature were acquired from the SILO database (source: Queensland Climate Change Centre of Excellence) covering the study area from 1970 to 2000 at a spatial resolution of 5 km. The data were stacked and processed as required to include in the equilibrium and spatial mode of the RothC model.

*Customized version of the RothC model*

Two versions of RothC were coded in R statistical programming language (R Development Core Team, 2008). The first program was used to calculate optimum inputs to reach the target SOC level interactively changing residue inputs (equilibrium mode). Second was used to carry out simulations spatially from 1970 to 2000, the time of sampling.

*Determination of the carbon components to initialize the model*

Seventeen sites which were under native forest were selected and it was hypothesised that they were at equilibrium at the time of sampling and had been at equilibrium since before clearing of vegetation for agriculture. These sites were split under the three soil groupings (Vertotols, Tenosols, Other) and the average SOC values for each was calculated. Long term average weather data for each site was calculated from the gridded weather data from 1970 to 2000. RothC was repeatedly run for 10,000 years with different rates of residue inputs and the input value that resulted in total carbon being equal to observed carbon was used as the starting point for the simulations. Once the optimum input residue levels were determined, RothC was run in equilibrium mode for 10,000 years. From this, the means of the different pools of carbon for the past 200 years were used as the starting point for all sites in the study area. Therefore, there were three different starting points based on the three soil groupings. The IOM was assumed to be 3.8 t C $ha^{-1}$(Coleman and Jenkinson, 1999). It was assumed that all the land was under native vegetation before the clearing for the agricultural use. RothC model parameters were kept as default except for the RPM rate constant which was changed from 0.3 to 0.15 (Skjemstad *et al*., 2004)

*Simulation of the carbon from 1970 to 2000 in relation to land use*

The input data for simulation included land use, soil information and spatial grid referenced to stacked monthly interpolated weather data. For the purpose of the simulation, it was assumed that the all land uses under agricultural production were cleared in 1970. Management files for the four land use classes were created *i.e.* native vegetation, pasture, dry land cropping and irrigated agriculture. For the native vegetation, residue inputs were used as determined by the equilibrium runs. In the case of pasture, annual residue inputs of 2.18 t C ha$^{-1}$ were used (Liu *et al*., 2011). For dryland agriculture a single rotation was adopted with winter crop wheat and winter rotation as chickpea while for the irrigated area, the land management file was prepared keeping summer crop as cotton and winter crop as chickpea. Residue inputs were calculated as described by Skjemstad *et al.*, (2004) taking the average yield of the study area (Australian Greenhouse Office, 2002). For irrigated cotton land use 97 mm were added to measured rainfall data for cotton growing months (Australian Cotton Cooperative Research Centre, 2000). All residues were assumed to have a carbon content of 45% (Skjemstad *et al.*, 2004).

*Model validation*

Two indices were calculated to evaluate the model prediction *i.e.*

*a) Mean Square Error (MSE)*

$$MSE = \frac{1}{n}\sum_{i=1}^{n}\left(y_i - \hat{y}_i\right)^2$$

*b) Root Mean Square Error (RMSE)*

$$RMSE = \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left(y_i - \hat{y}_i\right)^2}$$

Where, $y_i$ is the observed SOC content and $\hat{y}_i$ is the predicted SOC after simulations. The Lin's concordance correlation coefficient was calculated between measured SOC *vs*. predicted SOC (Lin, 1989).

**Results and Discussion**

*Determination of the carbon components to initialize the model*

For the equilibrium run of the RothC model average carbon values of the forest were 1.01 %, 2.04 % and 1.57 % for Vertosols, Tenosols and other respectively. The optimum annual inputs to reach the target carbon levels under the constant environment conditions were 1.30 t C ha$^{-1}$, 2.87 t C ha$^{-1}$ and 2.19 t C ha$^{-1}$ respectively for Vertosols, Tenosols and Others.

*Simulation of the carbon from 1970 to 2000*

Table 1 summarizes the mean predicted SOC for land use and soil types for year 2000. The largest mean predicted SOC is recorded for Pasture on Tenosols while the smallest was recorded for forest and pasture on Vertosols.

**Table 1: Summary of the predicted SOC with respect to land use and soil type for year 2000**

| Land use | Soil type | Mean Predicted SOC (%) | CV of the Prediction |
|---|---|---|---|
| Forest | Vertosols | 1.04 | 5.49 |
| Forest | Tenosols | 2.08 | 2.06 |
| Forest | Other | 1.60 | 2.39 |
| Dry land agriculture | Vertosols | 1.09 | 4.82 |
| Dry land agriculture | Tenosols | 2.11 | 0.62 |
| Dry land agriculture | Other | 1.74 | 5.84 |
| Irrigated agriculture | Vertosols | 1.12 | 5.59 |
| Pasture | Vertosols | 1.04 | 4.39 |
| Pasture | Tenosols | 2.16 | 3.55 |
| Pasture | Other | 1.67 | 4.39 |

Note – No irrigated agriculture sites were located within Tenosols and other soil types

The overall MSE was 0.74 % while RMSE was 0.86 %. Results of the Lin's concordance correlation ($r_L$) revealed the low correlation between predicted SOC *vs.* measured SOC ($r_L$ = 0.18). However, most of the applications of the RothC model in the spatial context are not validated with independent datasets which is an essential component of any modelling process. Low correlation was recorded due to a number of assumptions for the model simulation *i.e.* that the soil carbon under native vegetation was at equilibrium at the time of sampling, all land were under forest pre-clearing, clearance for agricultural production was done in 1970, uncertainty associated with splines, use of fixed rotations, and the use of pedo-transfer functions to predict the bulk density. Furthermore, the use of uniform starting condition (equilibrium SOC) and the use of relatively constant carbon input values with fixed rotations gave similar output after 30 years of simulations. This last point is shown by the small changes in the mean carbon predictions for each soil type from the initial carbon values (Table 1). This highlights the importance of incorporation of accurate land use and land use change information in running such models spatially. As this is an ongoing study we are working to improve the results by incorporating time series of aerial/satellite images to acquired land use change information/ land clearance information, incorporation of pre-European vegetation map of the area, determination of carbon pools with MIR and NMR techniques to initialize the model in landscape.

## Conclusions

The model predictions were poor but also important as they illustrate the perils of running RothC and similar models with naive inputs. Also it highlighted the importance of incorporation of accurate land use and land use change information when it is run spatially. Further work is aimed at calibration of the model and to identify contribution of the inputs and model parameters towards model error.

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# Wildfire effects on soil carbon and water repellency: A case study from the Blue Mountains, NSW, Australia

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## Abstract

Soil carbon and water repellency can be considerably modified as a consequence of a wildfire. However, very few studies have attempted to assess the recovery of carbon and water repellency and determine if there is a relationship between the two. This study examines the impacts of a wildfire that occurred in November 2009 which took place near Wentworth Falls in the Blue Mountains, NSW, Australia. The wildfire was classed into five different severity levels ranging from unburnt to high severity. Within each study area 15 ten metre transects were selected (three in each fire severity class). Along each transect ten sample points were selected at one metre intervals and samples were collected at each point at depths of 0-2 cm (surface) and 2-5 cm (subsurface). Samples were taken at six months and one year post-wildfire. The Water Drop Penetration Time (WDPT) method was used to measure the water repellency for each soil sample, while the combustion of soil was used to measure total carbon for samples bulked across each transect. A two-way analysis of variance (ANOVA) was used to analyse the results, where severity and time were the factors. A statistically significant difference in both carbon and water repellency was seen to occur in the surface soil as the burn severity increased. Carbon levels in the subsurface soil were seen to significantly increase over time. However, there was no significant interaction between time and burn severity for either carbon or water repellency.

## Introduction

Wildfire severity is spatially heterogeneous and can have severe impacts on soil properties (Bryant *et al.* 2005; Cerdá and Doerr 2005; Massman *et al.* 2008). The heat generated during a wildfire can cause considerable changes to soil total carbon and soil water repellency (Doerr *et al.* 2004). Surface soil undergoes the most significant changes due to the surface soil being in direct contact with the wildfire causing temperatures to be the highest at the soil surface (Neary *et al.* 2005). Temperatures at 5 cm in mineral soils rarely exceed 15°C and often no heating occurs below 20–30 cm (DeBano *et al.* 1998). Duration is perhaps the component of fire severity that results in the greatest below-ground damage. In fact, intense but fast moving wildfires at well fuelled sites do not transfer much heat down to more than few centimetres below the surface. After wildfires, soil temperatures can remain high for a few minutes to several days (Certini 2005). Although these soil changes have been identified throughout literature, there is a major gap in regards to the relationship between soil carbon and water repellency recovery post-wildfire. Therefore, the aim of this paper is to determine if there is a relationship between soil carbon and water repellency recovery in a fire affected area.

## Study area and methods

Wentworth Falls is situated in the Blue Mountains, approximately 80km west of Sydney, Australia. The underlying geology of Wentworth Falls consists of an ancient Triassic sandstone plateau with Narrabeen mudstone embedded throughout it. The soils within the region are dominated by coarse sands. Wentworth Falls was affected by wildfire in November 2009 and was previously burnt in January 2002. Both sites are dominated by resprouting *Eucalyptus* sp. with a dense shrubby understorey.

Landsat 5 satellite imagery was obtained to assess pre- and post- wildfire vegetation. The differenced Normalised Burn Ratio (dNBR) was computed to determine the difference in NBR values between the pre- and post- wildfire periods across the study area. The dNBR was then classified into five classes to quantify the relative degree of fire severity that occurred across the study area. In this instance, 5 severity classes were used ranging from unburnt to very high burn severity (Fig.1; see Chafer 2008).

For each severity class, three sites were selected at random (Fig. 1). At each site a ten metre transect was marked out and soil samples were collected at one metre intervals along the transect. At each sample site, the 0-2 cm and 2-5cm depth (total n = 300) intervals were sampled. The random selection of sites was restricted to ensure are all sites were at least 20 m from any human disturbance, i.e. roads. Leaf litter and organic matter was gently brushed away from the surface at each sampling site. All samples were left to air dry at 27°C. Samples were sieved through a 2 mm sieve and the fine earth fraction retained for analysis. The same locations were sampled both 6 and 12 months post-wildfire.

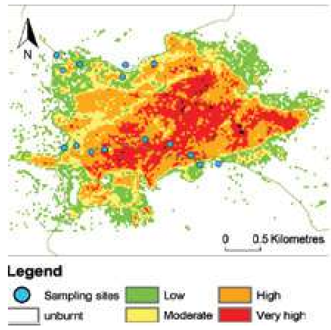


**Figure 1. Burn severity map for Wentworth Falls and sample sites.**

Water repellency was analysed in the laboratory through the Water Drop Penetration Time (WDPT) technique (Bisdom 1993). Each sieved soil sample was placed into a petri dish at least 1 cm deep and then was slightly compacted using a weight. All samples were left to equilibrate over night in controlled atmospheric conditions (20°C). Using a standard eye dropper, 3 drops of distilled water were placed on the surface of each soil sample and the time for each droplet to penetrate into the soil was recorded. Each recording was able to then be compared to the WDPT time intervals used by Bisdom (1993). Samples which infiltrated <5 seconds were classed as wettable, 5-60 seconds classed as slightly wettable, 60-600 seconds strong repellency, and 600-3600 seconds severe repellency. All sample recordings were terminated once the time reached 3600 seconds.

The average total carbon per transect was analysed using a LECO combustion analyser. Approximately 1 g of soil from each sample site along each transect were combined, for the 0-2 cm and 2-5 cm depth intervals. Each of the bulked transect samples was then ground until the entire sample passed through a 53 micron sieve. Approximately 0.1 g of soil was used to measure the total carbon.

As the transects rather than the sites within each, were selected randomly, the mean repellency for each transect was calculated and used in the subsequent statistical analysis. Therefore, the average total carbon and repellency time for each transect was recorded for each depth interval, the top 0-2 cm and bottom 2-5 cm. From this, the results for each transect were then categorised under the appropriate severity class and time period in which they were sampled (6 months or 1 year post-wildfire). Normality of data was assessed and if not normal it was transformed and logged. An analysis of variance (ANOVA) was used to determine if there was a significant difference ($P < 0.05$) between:

a) the water repellency and total carbon across the different burn severity classes;

b) the two time periods that were sampled; and

c) the interaction between burn severity and time.

**Results and discussion**

No significant difference in carbon or water repellency was found in the interaction between burn severity and time (Table 1). This is seen as all p-values were > 0.05.

Carbon levels in the unburnt soil had an average of 11-12 % in the surface soil and 6-9% in the subsurface soil (Fig. 2). This percentage of carbon is normal in Eucalypt forest as Rodríguez-Alleres *et al*. (2007) found organic carbon to be greater in Eucalyptus forest (~11.1%), in comparison to Pinus sp. (~9.7%), grassland (~5.1%) and maize (3.9%). These carbon levels are further supported by a study conducted by Atanassova and Doerr (2011) in Australia assessing three eucalypt-forest soils which found total organic carbon levels to range from 7-13.6% in sandy soils to 10% in a sandy loam soil.

Unburnt sites at Wentworth Falls have severely repellent soils (Fig. 3), which could be a result of vegetation species. Doerr *et al*. (2006) and Howell *et al*. (2006) established that soil repellency is common in pre-fire drought conditions in Australia. However, this could also be a reflection of the coarse sandy soil particles found within Wentworth Falls. Water repellency in sandy soils is known to develop as a consequence of sand particles being coated with organic substances produced by fungal activity (Chan 1992).

**Table 1. P-values for carbon and water repellency**

| | Time | Severity | Time.Severity |
|---|---|---|---|
| Carbon percentage (0-2 cm) | 0.123 | 0.002* | 0.599 |
| Carbon percentage (2-5 cm) | 0.003* | 0.065 | 0.948 |
| Water repellency (0-2 cm) | 0.144 | 0.043* | 0.799 |
| Water repellency (2-5 cm) | 0.109 | 0.057 | 0.934 |

* Significant at a 0.05 level

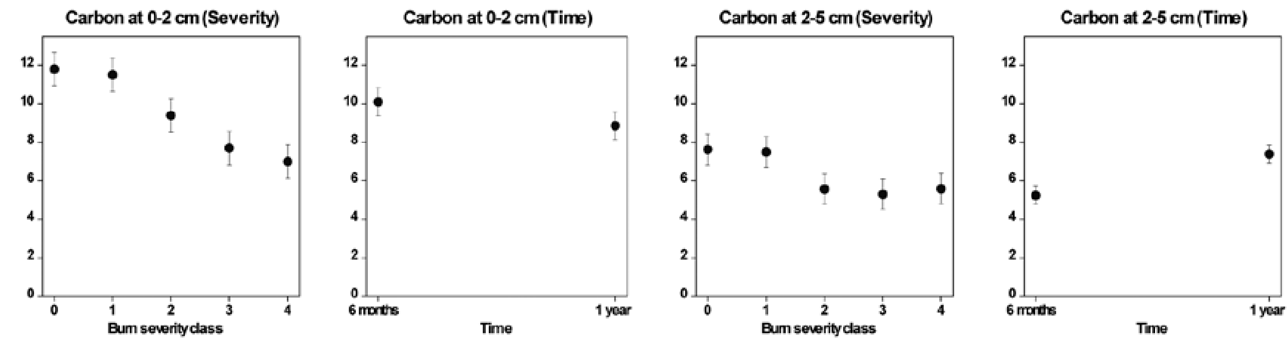


**Figure 2 . Mean carbon percentage for the surface and subsurface soil for each burn severity class and each time period sampled.**

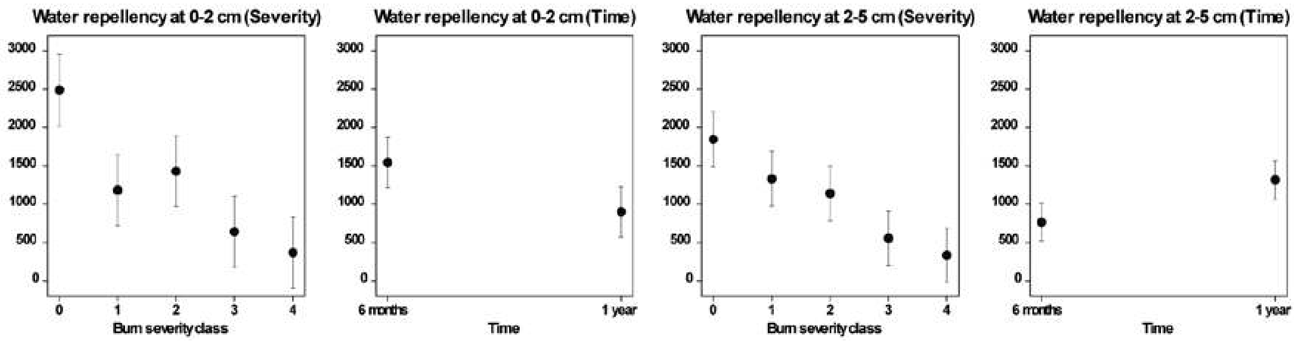


**Figure 3. Mean water repellency for the surface and subsurface soil for each burn severity class and each time period sampled.**

Wentworth Falls predominantly experienced a gradual decline in carbon and water repellency in both the surface and subsurface soil as burn severity increased (Fig. 2). Study sites affected by low burn severity became noticeably less repellent than the unburnt sites, whilst carbon levels experienced a slight decrease in sites classed as low burn severity. Carbon percentage was seen to further decline in both the surface and subsurface soils. The removal of vegetation and organic matter during the wildfire would have released carbon into the atmosphere, therefore reducing the amount of carbon in the soil, causing this decline as more vegetation was destroyed in higher burn severities (Davidson and Ackerman, 1993). Water repellency was seen to increase in areas influenced by moderate burn severity, followed by a decline in high and very high burn severities. The water repellency in the subsurface continued to decline as the burn severities became higher. This general decline in water repellency with increase in burn severity is a result of the destruction of water repellency which occurs as temperatures in the soil reach between 280 and 400°C (DeBano and Krammes, 1966). Overall, a significant difference in carbon was identified at 0-2 cm with a *P*-value of 0.002 (Table 1). This difference is also identified in Fig. 2 by assessing the confidence intervals (CI) around each mean. Where the CI overlaps between severity it suggests there is no significant difference, however the CI does not overlap between low and moderate burn severities, suggesting a significant difference in carbon occurs. This significance also occurs at 0-2 cm for water repellency ($P$ = 0.043) where CI does not overlap on the graph between unburnt and low burn severities.

When assessing time as an independent variable it can be seen that both surface carbon and water repellency slightly declined between 6 months and 1 year post-wildfire in the surface soil. The results of the subsurface soil was opposite as both carbon and water repellency increased between 6 months and 1 year post-wildfire. This is possibly the result of vegetation growth and increased organic matter availability supplying higher levels of carbon into the soil. Vegetation recovery also consists of the production of more roots, especially around 5 cm below the soil surface, allowing for an increase in carbon stock (Mendham *et al.*, 2003). However, the only significant difference was for carbon levels at the subsurface (Table 1; Fig. 2). Nevertheless, this study shows that both carbon and water repellency have similar responses to wildfire. Further work should consider the change in the form of carbon (labile or stable) post wildfire and with different severity levels.

## Conclusion

This study indicates that there is a strong relationship between the recovery of carbon and water repellency post-wildfire. It is clear that with an increase in burn severity both carbon and water repellency are reduced. A significant difference in carbon and water repellency occurred in the surface soil as burn severity increased. It was also evident that both the carbon and water repellency decreased in the surface soil 1 year post-wildfire, while an increase occurred in the subsurface soil. However, only the carbon in the subsurface was seen to significantly change over time. No significant interaction between time and burn severity occurred. This linkage between carbon and water repellency in post-wildfire recovery could help contribute to catchment management when determining the impacts wildfire has on the local Sydney Basin catchments.

## Acknowledgments

The authors would like to thank everyone who helped with the sample collection including Francesca Cole, Senani Karunaratne, Jason Lessels, Charles Moszkowicz, Nirmala Liyanage and Richard Muita.

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# Tuesday 4 December

## SOIL FERTILITY AND SOIL CONTAMINANTS (PHOSPHORUS)

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# A new technique to visualise diffusion of phosphorus and zinc from fertiliser

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Nutrients applied to soil as fertiliser move away from the point of application mainly through diffusion. Sorption and precipitation reactions reduce the mobility of nutrients such as phosphorus (P) and zinc (Zn). The extent of diffusion of P and Zn in soil is usually assessed by laborious and destructive methods (sampling of soils in small sections) or through the use of radioisotopes (autoradiography) or synchrotron-based X-ray fluorescence microscopy. We developed a novel cheap method to visualise P and Zn diffusion from fertilisers in a simple and non-destructive way. Filter papers are impregnated with Fe oxide (for P) or calcium carbonate (for Zn). The paper is placed on top of the soil in which a fertiliser granule or fluid fertiliser injection has been incubated for a given period. After a sufficient deployment period, the paper is removed and the P or Zn captured on the filter paper is visualised colorimetrically, using malachite-green for P or dithizone for Zn. We used this method to assess the effects of soil type, moisture content, fertiliser formulation and fertiliser additives on P and Zn diffusion from the fertiliser source. As the method is non-destructive, additional chemical measurements can be performed on the same samples. This method allows easy comparison of fertiliser sources and enables a better understanding of the physicochemical processes affecting fertiliser P and Zn behaviour at the soil:fertiliser interface.

# Phosphorus Distribution in Steep North Island hill country in New Zealand: Preliminary results

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Helicopters equipped with GPS now have the capability to apply fertiliser to areas as small as 50 m x 50 m, creating an opportunity for varying fertiliser regimes in hill country regions. The overall objective of our study was to determine whether precision agricultural methods could be applied in hill and steep-land areas to maximise P fertiliser use efficiency and minimise P loss to waterways. ArcGIS was used to delineate six soil-landscape units on a typical Central North Island hill country farm using aspect, slope and elevation. The soil comprised Allophanic Soil (Andisols) on the flat to low slopes with Brown Soil (Inceptisols) on the steeper hills. Forty-one landscape-units were identified within the research area that fell within six soil-landscape groups; (North Medium (slope of 13-25 ˚), South Medium (slope of 13-25 ˚), North Steep (slope of >25 ˚), South Steep (slope of >25 ˚), High Ridge (slope of 1-12 ˚, >400 m elevation), and Low Valley (slope of 1-12 ˚, < 150 m elevation)). Three replicate units from each of the six soil-landscape groups were randomly selected, giving 18 study-units. Samples from 1 transect and 5 grids were collected for each study-unit. Olsen P, Anion Storage Capacity, and pH were determined for each sample. Preliminary results show large between-sample variability, and no strong relationship between Olsen P and soil landscape units.

# Decrease in environmental and agronomic phosphorus concentrations when fertiliser is reduced or omitted from a range of pasture soils with varying phosphorus status

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Concentrations of soil phosphorus (P) above agronomic optimum provide no production benefits but can result in unnecessary losses of P that can adversely impact water bodies. Reducing these excessive P concentrations in pasture soils can be achieved through applying P at less than maintenance requirements. A field study of decreases in P was undertaken on soils with a range of initial P concentrations and on-going P fertiliser application rates. Decreases in $CaCl_2$-P (so-called environmentally available P) were proportionally greater than those for Olsen- and Colwell-P (so-called agronomic P). The decline in all P measures was exponential, with larger decreases occurring at higher initial soil P concentrations, and when less on-going P fertiliser was applied. A model was developed to assist policy makers and primary producers in setting realistic timeframes to return excessively concentrated soils to agronomic and/or environmental optimum. A study of changes between P pools revealed that the majority of the decreases in soil P were due to movement to pools not directly plant-available. Phosphorus not accounted for in the soil or in exports in farm products was assumed to have been lost to the surrounding environment. A further study identified an extremely low P-fixing soil as susceptible to large P losses, with the majority of P applied to it not recoverable to a depth of 1 m. Further work on the influence of soil properties on decreases in soil P is required.

# Predicting the changes in environmentally and agronomically significant phosphorus forms following cessation of phosphorus fertiliser applications to grassland

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## Abstract

Phosphorus (P) loss from soil can impair surface water quality. One possible strategy to decrease P loss is to stop applying P fertilisers. Agronomic measures such as Olsen P may not predict the potential for P loss, hence we examined the changes in both agronomic (Olsen P) and environmental (water-extractable P - WEP) P tests following a halt to P fertiliser application to three long-term grassland field trials. Exponential decreases in Olsen P and WEP over time were observed at all of the trial sites. Rates of decreases in Olsen P and WEP concentration were best correlated with the initial WEP concentration (WEPi) and the quotient of Olsen Pi/P retention, respectively. Equations were generated to predict the time taken for Olsen P and WEP concentrations at each site to decrease to a target value of 25 mg/kg and 0.02 mg/L, respectively. The estimated time for WEP concentrations to reach the target ranged from 23 to 44 years, but 0 to 7 years for the agronomic target. This demonstrates the potential large lag times between halting P fertiliser application and observed improvements in stream water quality, and highlights the disparity between agronomic and environmental optimum soil P values. Isotope exchange kinetics (IEK) is considered a more accurate method to determine plant available P and may provide an improved estimate of the potential for P loss. IEK was carried out for a selection of soils spanning the length of each trial and the replacement of Olsen P/PR with E1 min/PR improved the prediction of the rate of decline in WEP concentration.

## Introduction

Long-term application of P fertilisers in excess of the crop requirements has led to a build-up of P in many well developed agricultural soils. High P soils have an increased potential for P loss to surface water via surface or subsurface run-off (Sharpley *et al.* 1994). Coupled with an increasing world population and concerns over future phosphorus availability (Gilbert 2009), clearly more sustainable and P efficient farming practices are required. One method to improve P efficiency by decreasing P loss would be to stop applying P fertilisers and decrease soil test P (STP) concentrations.

The rate of decline in STP concentration following cessation of P fertilisers has been shown to be highly variable and site specific. Factors such as soil mineralogy, farm management and initial STP concentration have all be shown to be influential (Lookman *et al.* 1995). Furthermore, the decline in STP concentration may not directly translate to a decrease in the potential for P loss to water. The relationship between STP and the potential for P loss may vary according to soil type, land-use practices and run-off pathways (Sharpley *et al.* 1994). McDowell & Sharpley (2001) found that a deionised water extract of soil (water extractable P; WEP) could be used to estimate the P loss to surface run-off. An alternative technique to assess soil P concentrations uses isotope exchange kinetics (IEK). E values describe the soil P pools exchangeable over 1 min, 24 h and 3 months ($E_{1\ min}$, $E_{1min-24\ h}$, $E_{24\ h\ -\ 3\ months}$) and have been related to plant uptake over different time frames (Frossard & Sinaj, 1998) while McDowell *et al.* (2001) demonstrated that the potential for P loss to overland flow was closely related to $E_{1min}$. We argue that the measurement of IEK provides a fuller picture of P availability and the potential loss to water than single extraction methods.

This study utilised archived soils from three long-term trials and aimed to (1) investigate the decline in the concentration of agronomic (Olsen P) and environmental (WEP) soil tests following the cessation of P fertiliser; (2) derive an equation to predict the decline in Olsen P and WEP concentration from readily measureable soil parameters; (3) determine whether the use of IEK can improve the prediction of WEP.

## Methods

*Study sites*

Archived samples of air-dried, sieved (<2 mm) surface soil (0-75 mm depth) were sourced from long-term trials across New Zealand. The trials were initially established to investigate the effect of halting fertiliser inputs on farm productivity. They spanned 16 – 26 years following a halt to P inputs and included contrasting soil types and a range of initial STP concentrations (Table 1).

**Table 1. Summary details of the long-term fertiliser trials utilised in this study**

| Site location | New Zealand soil classification | Management | Previous fertiliser rate (kg P $ha^{-1}$ $yr^{-1}$) | Years since P fertiliser halted | Number of replicate plots |
|---|---|---|---|---|---|
| Whatawhata | Typic Oxidic Granular soil | Sheep grazed pasture | 10<br>50<br>100 | 26 | 2 per treatment |
| Winchmore | Pallic Firm Brown soil | Irrigated sheep grazed pasture | 34<br>51 | 21 | 4 per treatment |
| Lincoln | Mottled Immature Pallic soil | Ungrazed pasture | 0 | 16 | 4 per treatment |

*Soil analysis*

Olsen-P and WEP were determined on each soil. WEP was determined via extraction with deionised water, at a 1:300 soil-to-solution ratio with 45 minutes end-on-end shaking as outlined by McDowell & Condron (2004). Phosphorus retention (PR) was determined for the separate plots from each trial in triplicate from three bulked sampling dates following the procedure of Saunders (1965). In brief, soil samples were equilibrated by shaking soil for 16 hours end-over-end with the P retention solution, containing 1 mg P ml-1` at a 1:5 soil to solution ratio and buffered at pH 4.6. Samples were centrifuged and the P remaining in the supernatant determined colorimetrically via reaction with nitric vanadomolybdate acid (Kitson & Mellon, 1944). Archived data was available for pH (water; 1:2.5 soil-to-solution ratio), and total organic carbon (TOC by loss on ignition).

*Isotope Exchange Kinetics*

Isotopic exchange kinetics were studied in all treatments and replicate plots for a selection of sampling dates (every 4-6 years depending on the availability of archived samples). The IEK method has been described in detail in Frossard & Sinaj (1998). For this study, soil was shaken in deionised water for 16 hours to reach equilibrium in a 1:300 soil-to-solution ratio to mimic the extraction of WEP. An aliquot of each soil sample was removed, filtered (0.45 µm) and the P concentration determined - providing the intensity factor (Cp). Following the addition of 0.01 – 0.05 Mbq carrier free $^{33}$P-labelled phosphate ions in 1 ml $H_3{}^{33}PO4$, aliquots of the soil-solution were taken at 1, 10, 30 and 60 minutes and immediately filtered through a 0.45µm filter. The reactivity of the $^{33}$P in each aliquot was determined using a liquid scintillation counter.

The rate of decline in radioactivity with time (r/R) was calculated from Eq 1.

$$r/R = r_1/R \; x \; (t + r_1/R_1{}^{/n})^{-n} \qquad [1]$$

where R/r1is the ratio of total introduced radioactivity (R) to the radioactivity remaining in solution after 1 minute (r1), and n is a parameter to estimate the decrease in radioactivity with time (t) as determined from the slope of a plot of log (r(t)/R against log(t).

Values of E were then calculated for 4 time points, 1 minute ($E_{1\,min}$), 30 minutes ($E_{30\,mins}$), 24 hours ($E_{24hrs}$) and 3 months ($E_{3\,mo}$) from Eq 2.

$$E = a \; x \; C_p/(r/R) \qquad [2]$$

where a is the soil to solution ratio (i.e. 1:300) and Cp is the intensity factor.

**Results and Discussion**

Olsen P and WEP showed a significant exponential decline over time following a halt to P fertiliser at almost all trial sites and previous P application rates. An exception was Whatawhata, where the lowest intial P application rate of 10 kg P/ha/yr did not show a significant decrease in WEP concentration with time. The decrease in soil P concentration could be described by Eq 2.

$$y = \alpha \text{ x } e^{-\beta t} \qquad [2]$$

The value of β describes the rate of decrease in P concentration over time. A correlation analysis was performed to determine those soil parameters that best predict the rate of decline in Olsen P and WEP concentration. The strength of correlation between β values for Olsen P decline and soil properties was greatest for the initial WEP concentration (WEPi), while for WEP decline the correlation was greatest when PR was included in the quotient (initial Olsen P (OPi)/PR) illustrating the importance of soil mineralogy (PR also estimates Al- and Fe-oxide concentration) in controlling P loss to surface run-off.

*Predicting soil P decline*

One of the main objectives of this study was to derive an equation to predict the decline in environmentally and agronomically significant P fractions from easily measured soil parameters. By substituting WEPi for α Eq 2 can be rearranged to give time for WEP to reach a target WEP concentration WEPt (Eq 3).

$$t = -\,^{\beta-1}[(\ln \text{WEPt}) - (\ln \text{WEPi})] \qquad [3]$$

Regression analysis showed that 79 % of the variation in β values can be described by the quotient of the initial Olsen P concentration dived by PR (Eq 4).

$$\beta = 0.035 \ln \text{OPi/PR} - 0.0455 \qquad [4]$$

Equations 3 and 4 were combined to predict the time it would take for WEP to decrease to a target WEP concentration following a cessation of P fertiliser application (Eq 5).

$$t = 1/(-0.035 \text{ x } \ln \text{OPi/PR} - 0.0455) \text{ x } (\ln \text{WEPt} - \ln \text{WEPi}) \qquad [5]$$

The target WEP concentration was based ANZECC guideline for DRP to show increased likelihood of adverse effects in lowland streams and rivers, but diluted by the ratio of annual surface runoff to total flow and set at 0.02 mg P/L.

Using a similar approach an equation was generated to predict the decline in Olsen P concentration to an agronomic optimum. Regression analysis suggested that 78% of the variation in β values can be described by the initial WEP concentration in the soil (Eq 6).

$$\beta = 0.032 - 2.81 \text{ x WEPi} \qquad [6]$$

Substituting the initial and target WEP concentrations in Eq 3 with equivalent Olsen P concentrations and the β value from Eq 6, the time for the initial Olsen P to reach the target concentration, set at 25 mg P/kg, was determined from Eq 7.

$$t = 1/(0.032 - 2.81 \text{ x WEPi}) \text{ x } (\ln \text{OPi} - \ln \text{OPt}) \qquad [7]$$

Figure 1 shows the time it would take for WEP and Olsen P concentrations at the start of each trial to decrease to the target concentrations as calculated from Eqs 5 and 7. For WEP, this ranged from 23 to 44 years and for Olsen P, 0 to 7 years. This demonstrates the potential large lag times between halting P fertiliser applications to high P soils and observed improvements in-stream water quality and highlights the disparity between agronomic and environmental optimum soil P values, indicating that even at the agronomic optimum, pastoral soils can lose environmentally significant P concentrations.

*Prediction of soil P response using IEK data*

The four exchangeable soil P pools declined exponentially with time following a halt to fertiliser inputs. The E values were combined with the Olsen P and WEP data in a stepwise regression. The Mallow Cp parameter indicated that only one variable, the quotient E1min /PR, was required to predict the rate of decline in WEP concentration and Eq 8 describes 85% of the variation compared to 75% from Eq 7. Thus, if the relationship between E values and Olsen P could be determined for a wide range of soil types, this may allow a more accurate prediction of the potential lag time between cessation of fertiliser and improvements in water quality.

$$\beta = 0.307 \text{ x } (\text{E1min/PR}) - 0.0474 \qquad [8]$$

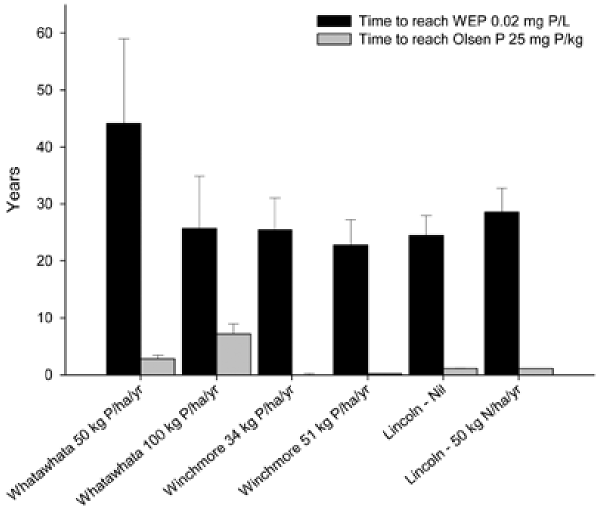


**Figure 1: Estimated time to reach 0.02 mg WEP/L and 25 mg Olsen P/kg for each site.**

## Conclusions

Withholding P fertilisers on a wide range of New Zealand pastoral soils led to an exponential decline in Olsen P and WEP. The rates of decrease in WEP and Olsen P concentrations were best described by the quotient of Olsen P over PR and the initial WEP concentration respectively. The relationship between WEP decline and soil properties can be further improved by the replacement of OPi/PR with E1 min/PR. Prediction of the time it would take for a soil to reach a target Olsen P or WEP concentration from these relationships highlights the disparity between environmental and agronomic optimums. This suggests that attempts to decrease P loss through halting fertiliser P applications may lead to significant losses in farm productivity. Further research is required to try and address this issue, for instance technical advances to manipulate the plant availability of the residual soil P pool.

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# Labile P pools in root-zone soils under cereal, legume and oilseed break crops at several sites across southern Australia

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Two important labile P pools in soil are the plant-available and microbial P pools. A balance between these pools is required to enable plant P requirements to be met whilst simultaneously maintaining an optimal microbial biomass activity that will convert stable organic P into more labile forms. Further, microbial biomass itself can act as a source of available P through the decomposition of dead microbial cells. These labile P pools in soil are influenced differently by cereals and break crops during their growth cycle because they fundamentally vary in the effects that their roots have on rhizosphere P dynamics. Lupins may solubilise P via exudates, canola may increase the release of microbial P via exudation of biofumigant glucosinolates, legumes may enhance microbial P via excretion of labile N compounds, and cereals may do the same by excreting large amounts of labile C. Overall, the microbial community of the rhizosphere will differ in the range of P solubilising organisms present. Furthermore, each of these crops has different root architecture and will therefore differ in P uptake from the labile pools. Whilst much has been reported for break crop species and their influence on P dynamics in controlled environment studies, there are few reports from field studies, and information is scant for southern Australia. This study will report measurements of microbial P and P availability for root zone soils under break crops and wheat from five locations in southern Australia, covering a range of climates and soil types. The aim was to determine whether the effects that these break crops have on labile P pools in soil (especially the microbial pool) is short term (i.e within the life cycle of the break crop), or if the effects persist in the longer term and can potentially influence P availability to subsequent crops.

# Tuesday 4 December

## SOIL AND LAND DEGRADATION (13A)

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# Controlled traffic as the basis of sustainable soil management for intensive vegetable production

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## Abstract

Intensive vegetable production relies on diverse crop rotations, frequent cropping schedules and intensive use of machinery for incorporation of crop residue, seedbed preparation and harvest. Intensive random traffic, as is used in vegetable production, requires excessive tillage in an effort to remediate soil compaction. By permanently isolating traffic to defined wheel tracks, controlled traffic farming (CTF) provides a number of farming system benefits including improved energy efficiency, soil health, crop yield, timeliness and economics. The adoption of controlled traffic in the Tasmanian vegetable industry is challenged by a wide diversity of machinery, and topography ranging from flat to steeply undulating.

Research in the vegetable industry has shown improvements in soil physical conditions can be achieved in a short time with the use of controlled traffic. The implementation of controlled traffic leads to a change in tillage management, resulting in fewer, less energy-intense, operations. The role of tillage becomes largely one of managing residue to provide seeding and subsequent harvest conditions appropriate to the crops grown. The need to remediate soil compaction largely disappears, apart from some remedial deep tillage at the interface of the wheel track and the crop bed to prevent excessive encroachment of wheel track compaction into the bed.

**Key Words**
Controlled traffic, vegetables, soil structure, tillage

## Introduction

Using controlled traffic farming (CTF) to separate compacted traffic zones from soil used for growing crops provides a wide range of benefits for crop production. Benefits observed in research and commercial practice in the grain and sugar industries include reduced soil erosion, more efficient energy, water and fertiliser use, improved soil structure, organic matter and timeliness, and increased productivity (Bowman, 2008; McPhee *et al*., 1995; Stewart *et al*., 1997).

Although the adoption of CTF in the Australian grain and sugar industries has increased in recent years, uptake in the intensive vegetable industry is very limited for a number of reasons, including diversity and incompatibility of current equipment, and often, a diversity of ownership arrangements (e.g. private, contractor and company-based machines) requiring industry-wide involvement for effective change. This is particularly the case for the Tasmanian vegetable industry, which encompasses processing and fresh sectors, with many second tier agri-businesses (e.g. processors, fresh market packers) also engaged in the contracting of services such as sowing and harvest.

Recent research in vegetables in Tasmania has shown that the isolation of traffic can improve soil conditions in the crop growth zone between the wheel tracks. While these improvements offer significant potential advantages for crop production, a number of issues need to be addressed for the practical adoption of controlled traffic in the vegetable industry, including tracking stability on compacted wheel tracks and side slopes, and implement working and track width compatibility.

## Materials and Methods

A replicated experimental site was established at the TIA Vegetable Research Facility (41$^o$ 12.34’ S, 146$^o$ 15.84’ E), near Devonport, Tasmania. The site has Red Ferrosol soil and undulating topography, representative of prime vegetable production areas in Tasmania, and a winter dominant annual rainfall of 1000 mm. Irrigated summer production of temperate vegetables is the main enterprise of the region, although rain-fed winter crops are also grown.

Two treatments were imposed – controlled traffic based on 2.0 m wheel tracks, and conventional practices using random traffic and wheel track configurations appropriate to normal industry practice for the crop being grown. Both treatments were cultivated as required, although the type, number and intensity of tillage operations varied with the treatment and seedbed requirements.

The site was established from pasture in June 2008. The controlled traffic treatment was deep ripped using a tractor with RTK[1] satellite guidance and auto-steer. The ripper tynes in the wheel track line were removed. The conventional area was ripped with a standard ripper (tynes left to operate in the wheel tracks) and guidance was not used.

Crops grown represented commercial production for the local region, with one important exception. Only crops that could be established and harvested using equipment on 2 m wheel tracks were grown. This was to ensure, as much as possible, that the soil in the controlled traffic beds would remain untrafficked for consecutive seasons. Crops grown over 3½ years were potatoes, onions, broccoli, beans, processing carrots and a number of green manure crops (rye grass and BQ mulch).

Measurements taken mid-season of each cash crop were soil bulk density, and related derived parameters, and soil resistance. The cores used for soil sampling were 70 mm diameter and 50 mm deep. Bulk density cores were taken at depths of 0–50, 125–175 and 275–325 mm, which, with the 50 mm depth of the core are identified as surface, 150 mm and 300 mm, respectively. The core samples were weighed before and after oven drying, and the following parameters calculated: soil bulk density, porosity, gravimetric and volumetric water content, water filled pore space and the ratio of soil:water:air in the sample.

Soil resistance data were collected with a recording cone penetrometer. Insertions were made at 100 mm intervals across a 3 m transect, to a depth of 600 mm, with resistance force automatically recorded at 15 mm increments. The 3 m transect allowed inclusion of a full crop bed and adjacent wheel tracks in the recorded data. Infiltration tests, using a Cornell Sprinkle Infiltrometer, were conducted on only one occasion, during the growth of a winter broccoli crop.

## Results

### *Porosity*

Although soil cores were taken at three depths, only data from 150 mm are presented. This is often the approximate depth of final seedbed tillage operations, and is therefore most likely to capture the effects of soil compaction from seedbed preparation traffic that has been relieved only to the depth of final tillage. The data chosen for presentation relates to porosity. Data for other depths reflect similar trends, although the magnitude of differences varies.

Porosity is derived from bulk density samples. Figure 1 shows that porosity under controlled traffic is consistently higher than under conventional management, with the exception of the final cropping season. This reversal of trend is thought to be due to a reduction in tillage depth, and an additional tillage operation that was required to address issues arising from the previous green manure crop. The conventional area was mouldboard ploughed to incorporate the green manure, but mouldboard ploughing is inconsistent with the objective of retaining permanent wheel tracks in controlled traffic. Further, while there was a higher porosity in the controlled traffic bean crop, it is thought that the overall decline in porosity from the previous season was also due to tillage practices. The summer bean crop followed a winter broccoli crop, and it was necessary to revert to rotary tillage to manage the broccoli residue. The shallow rotary tillage in the controlled traffic treatment was just enough to manage the residue, rather than a deep operation as in the conventional treatment. This may explain the overall decline while still maintaining a relative advantage. Increases in porosity have been reported for other cropping systems which do not rely on tillage, such as zero-till grain production, under controlled traffic systems (McHugh *et al*., 2009). The increase in porosity under controlled traffic has implications for water holding capacity, drainage, aeration and root growth, all of which can be beneficial for plant growth (Boone, 1994).

[1] Real Time Kinematic – satellite guidance system that obtains a correction signal from a nearby ground-based reference station in order to provide ±2 cm accuracy

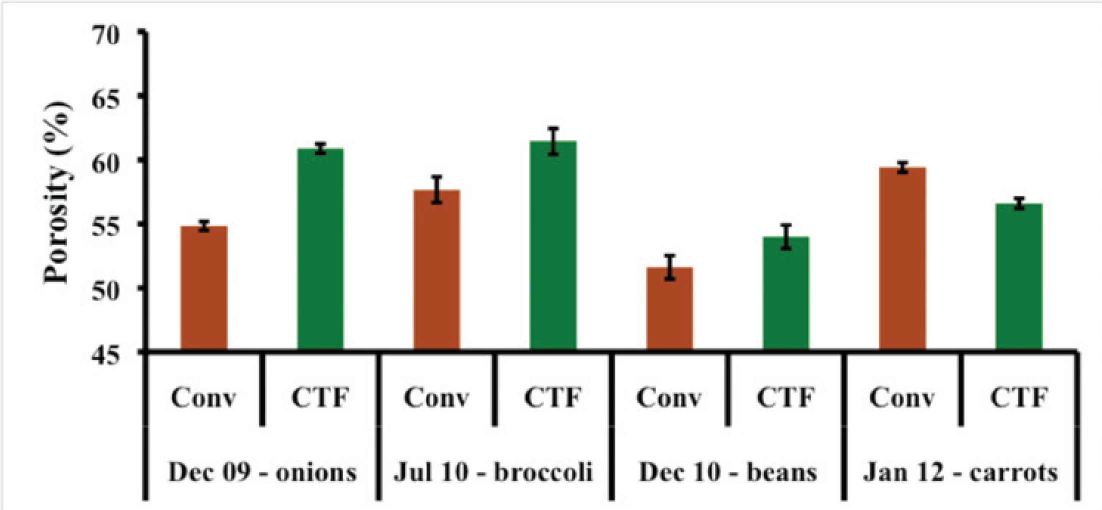


**Figure 1: Soil porosity (%) at 150 mm depth from conventional and controlled traffic treatments. Error bars indicate S.E. of means. All differences between treatments are significant (P<0.05) with the exception of the data for Jul 10.**

*Soil resistance*

Soil resistance, as measured by a cone penetrometer, is not considered to be a highly reliable and repeatable scientific measure, given the influence of soil water content. However, it is a useful illustrative tool for growers, particularly when the data is presented as a transect across beds and wheel tracks. Representative transect profiles are shown in Figure 2, taken after potato harvest and part way into the subsequent onion growing season. These illustrate that the low strength conditions within the CTF beds were maintained through potato harvest, whereas the conventional area needed substantial tillage to re-establish soil conditions suitable for onion growth. The wheel track locations in the controlled traffic transect are approximately centered around the 500 and 2500 mm locations on the x-axis.

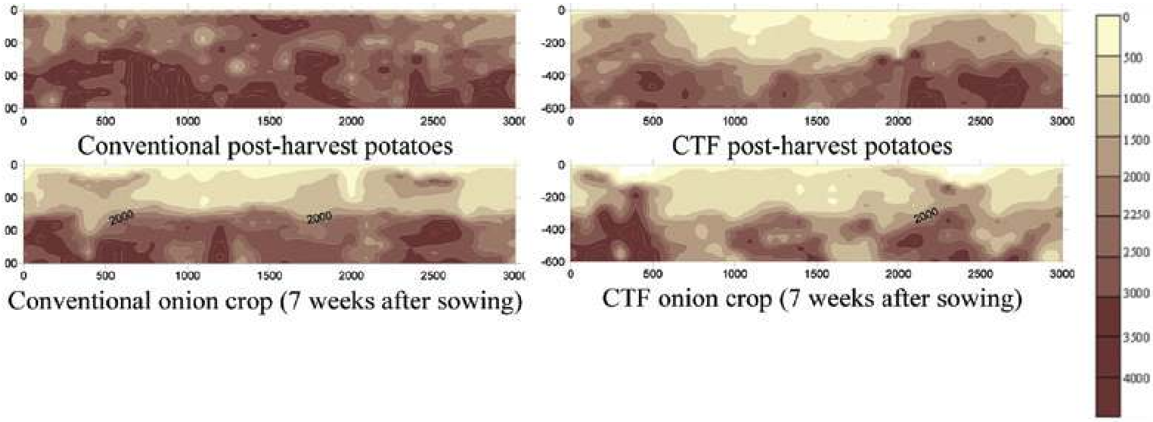


**Figure 2. Cone resistance profiles from transects taken from conventional (left) and CTF (right) treatments after potato harvest (top) and after seedbed preparation and sowing of the subsequent onion crop (bottom). Legend appears at right, resistance values in kPa,**

*Infiltration*

Infiltration tests were conducted only once, mid-season during a winter broccoli crop. A Cornell Sprinkle Infiltrometer (Ogden, 1997) was mounted on a ring 200 mm above the soil surface. The data show a substantial difference in infiltration capacity between the two treatments (Table 1), and within the error of measurement, the run off from the conventional area was equal to the rainfall. The differences in the data have important implications for reduced run off and erosion from the crop beds, and for the capture and storage of rainfall and irrigation water. Significant increases in infiltration and hydraulic conductivity under controlled traffic have been reported in the literature for a range of environments and cropping systems (Bai *et al*., 2009; Braunack and McGarry, 2006; Lamers *et al*., 1986; Tullberg, 2010).

**Table 1: Infiltration test data from conventional and controlled traffic treatments (July 2010)**

| | Conventional | Controlled traffic |
|---|---|---|
| Duration of test (min) | 30 | 90 |
| Average application rate (mm/h) | 199 | 183 |
| Average time to run-off (min) | 4 | Not reached |
| Average run-off rate (mm/h) | 202 | 0 |

## Conclusion

The soil physical changes that occur as a result of controlled traffic have impacts across many aspects of crop production. Although current tillage practices in the vegetable industry seek to remediate traffic-induced soil compaction, soil which is not subject to traffic in the first place provides a more resilient growing environment and requires significantly less effort for seedbed preparation. The findings from this work show clear benefits for soil management in intensive vegetable cropping through the adoption of controlled traffic. During the developmental phase of the system, the soil is subject to more tillage than is likely to be the case in a more highly refined system. Therefore, these benefits would likely consolidate over a longer period of controlled traffic management.

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# Impacts of land re-contouring on soil properties in Marlborough, NZ

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Landscape recontouring has been occurring over the last 10 years on the rolling and hilly lands of the wine region of Marlborough, New Zealand, and involves the movement of large volumes of soil and bedrock. Modification is intended to increase ease of vineyard establishment and management. Anthropogenic soils range between reconstituted profiles comprised of original A and B horizon material relayered over in situ bedrock (with a sharp contact) on crests and shoulder slopes, to profiles comprising mixed A horizon material over mixed B horizon and bedrock material over bedrock fill in footslope and toeslope positions.

This study aims to characterise the soils created by recountouring and evaluate the effects of this process on long term land use sustainability. Three recontoured vineyard sites in the Awatere Valley, Marlborough, were studied. Virgin and anthropogenic soils on these sites were described and sampled for pH, electrical conductivity (EC), and C.E.C. Intact cores were also taken for soil water holding capacity characterisation. Initial results indicate differences in the water holding capacity of recontoured and virgin soils, with recontoured soils holding a greater amount of water in the plant-available zone. EC measurements suggest that salts are being remobilised from saline bedrock incorporated into anthropogenic soils. At one recontoured site there has been surface salt precipitation and vine dieback around a seep, and an irrigation pond fed by subsurface and overland flow shows moderate levels of salinity. Release of sodium to the environment is a concern also for erosion and slope stability because of the potential for clay dispersion.

# Roadside survey technique to assess the trends in land use, land management and soil condition of geo-located sites in the wheatbelt of Western Australia

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A WA project assessing soil erosion through roadside survey (RSS) techniques was started in 2007 with the aim of monitoring soil condition (by reporting the occurrence of wind and water erosion as a value of the soil condition).

The RSS consists of a series of 16 transects selected to give a good distribution of geo-located sites across DAFWA agricultural regions and geographical land system zones, with a total of over 4000 individual sites and a road distance of about 8,000 km. Each is rated for percentage ground cover, soil texture, slope, vegetation height, soil detachment, wind and water erosion, land use and land management. From the observations, erosion hazard is calculated as a function of cover x texture x detachment, reported from nil to very high in six intervals. The transects are traversed twice yearly; once at maximum biomass (October-November) prior to harvest, and the second at minimum biomass (April-May) prior to seeding.

Preliminary analysis of six years of observations with five interseasonal (maximum to minimum biomass) "transitions", has identified that transitions from one level of erosion hazard to another occurs in about 15% of all transitions over the period of observation. This percentage varies from district to district and may be dependent on seasonality and land management.

This presentation covers the rationale behind the methodology and outline the additional products (such as predictive hazard model and efficient sampling regime) coming out of the RSS. These are used to identify practice change in agriculture of WA.

# Understanding salinity manifestations and pathways for better salinity management – experiences from the Jugiong catchment, NSW, Australia

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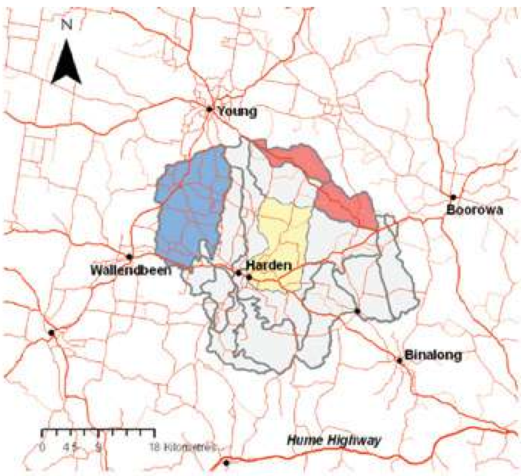
## Abstract

Simplistic views of salinity management and restoration have led to inefficient use of time, money and landholder good will. In some cases inappropriate management decisions have led to negative hydrological impacts. The Jugiong catchment is regarded as one of the most saline catchments in NSW, Australia. Efforts to manage salinity in this catchment have met with mixed results. The NSW Office of Environment and Heritage, NSW Department of Primary Industry and University of Canberra have undertaken a study of the salinity characteristics and variation across the Jugiong catchment. Using modelling, available biophysical data, rapid stream assessment, soil survey, water quality assessment and the Hydrogeological Landscape (HGL) methodology a picture of salinity in the Jugiong catchment has been ascertained. Only 6 of the 16 HGLs in the Jugiong catchment were shown to be major contributors to stream salinity. The aim of the study is to arrive at better targeted and more successful management outcomes.

## Introduction

The Jugiong catchment is widely regarded as highly saline and was ranked the most saline catchment in NSW in the 2009 Salinity Audit (DECC, 2009). Despite investment in salinity mitigation measures minimal improvement has been shown for most indicators. A Hydrogeological Landscape (HGL) study was conducted in the Jugiong catchment to inform the Murrumbidgee Catchment Action Plan (CAP) Assessment Project of the influence and behaviour of salinity in this landscape and to provide more effective land management recommendations. Work was completed in July 2011. The Assessment Project's aims were to integrate natural resource management (NRM) issues, including water quality impacts, land salinity and salt load impacts. For the project to meet its aims a number of issues including differences in scale, location and extent of salinity, variability across landscapes, linkage to modelling such as 2C Salt, the nature and positioning of on-ground works, data provision at an operational scale and prioritisation of areas and actions needed to be addressed. This paper reports the main findings of the HGL study, which integrated natural resource management knowledge on these issues, to satisfy the aims of the CAP Assessment Project.

## Study area



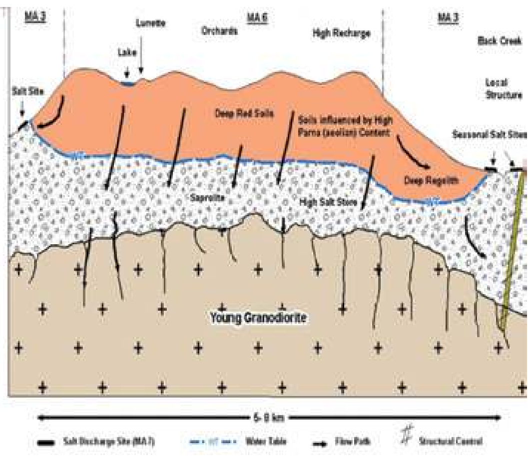




**Figure 1 – (left) Location diagram for the Jugiong catchment study area with the Wombat HGL (blue), Tiverton HGL (yellow) and Moppity Road HGLs (red) Figure 2 – (right) Conceptual diagram for the Wombat HGL.**

The Jugiong catchment lies on the gently undulating southwest slopes region of NSW, approximately 340 km from Sydney and 130 km north-west of Canberra. The Jugiong River flows into the Murrumbidgee River and is part of the Murray-Darling Basin. The study area includes the town of Harden.

**Methodology**

Rapid stream assessment consisting of EC, temperature and pH measurements was conducted at over 60 stations on the same day. This exercise has been conducted five times between August 2010 and July 2012 to gain time series data across a range of seasonal and flow conditions. Stream flow data was collected at a subset of stations (12 to 17 sites).

Landscape shape was determined using a Multi Resolution Valley Bottom Flatness index (MrBVF), digital imagery (ADS40 imagery) and field observation. The landscape was further partitioned into landform elements using available geological data, Landform 7 (lf7) modelling (Summerell *et al*., 2005; Wilford *et al*., 2007), regolith depth as modelled by the Weathering Index (Wilford, in press), soil landscape maps and reports (Andersson and McNamara, 2010) and existing salt outbreak and EC data.

A soil survey was undertaken to gain an appreciation of soil types and soil characteristics across the catchment. Standard soil test methodology was employed (National Committee on Soil and Terrain, 2009). Profile and soil layer information was recorded on NSW Soil Data Cards (Milford *et al.*, 2001). Soils were classified according to the Australian Soil Classification (Isbell, 2002). Samples were taken on a horizon basis and soil EC values determined from 1:5 soil:water extracts (Rayment and Higginson, 1992).

The HGL framework (Jenkins *et al.*, 2010) was employed to integrate information and to partition the catchment into discrete landform units with similar salinity characteristics. A HGL unit is a structured comparison of saline landscape characteristics including water pathways through the landscape, salt stores, relative mobility of salt within the landscape, salinisation processes and salt signature within streams. Conceptual models are developed to describe unique characteristics within each HGL. Areas were split into HGLs by the integration of geology, radiometric data, weathering index developed from radiometrics and soils, and landforms defined by MrVBF landscape modelling.

**Results**

There are 16 different landscape behaviours, shown spatially as HGLs (Figure 1), developed for the Jugiong Catchment. Each HGL has a conceptual diagram (Figure 2) showing landscape shape, relative depth of soil and regolith materials, salt stores, salt out breaks, pathways of water movement and landscape management areas as defined by field observation and lf7 modelling. Stream salinity does not vary equally across all sites in response to rainfall, reflecting differences in connectivity within the different HGLs (Figure 3).

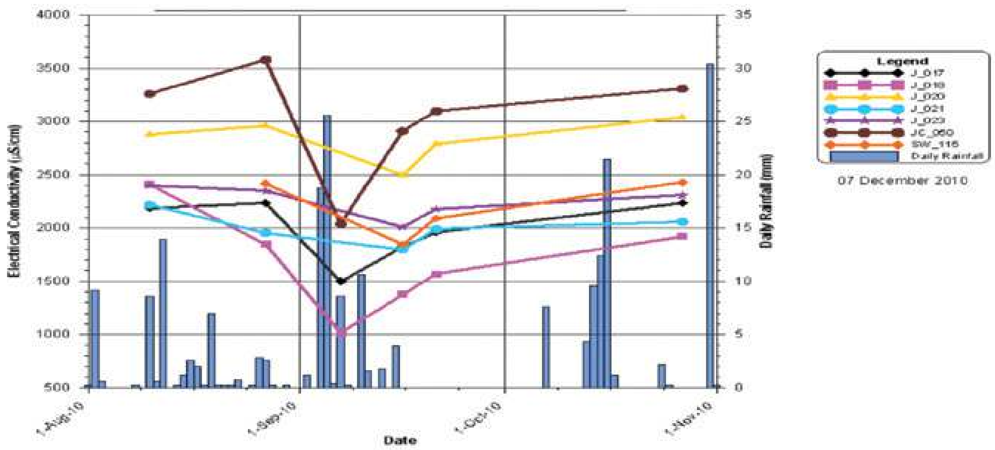



**Figure 3 - Stream EC values for selected stations across the Jugiong catchment showing the variable response to rainfall events**

Salinity behaviour in each HGL is defined by the following factors: land (dryland salinity), load (stream salt load t/year) and stream EC with an overall hazard determined. The priorities for salinity intervention can be ascribed to particular HGL landscapes and landscape elements (MAs) within the landscape. There is a marked variation in salinity impact (land/load/EC) across the catchment.

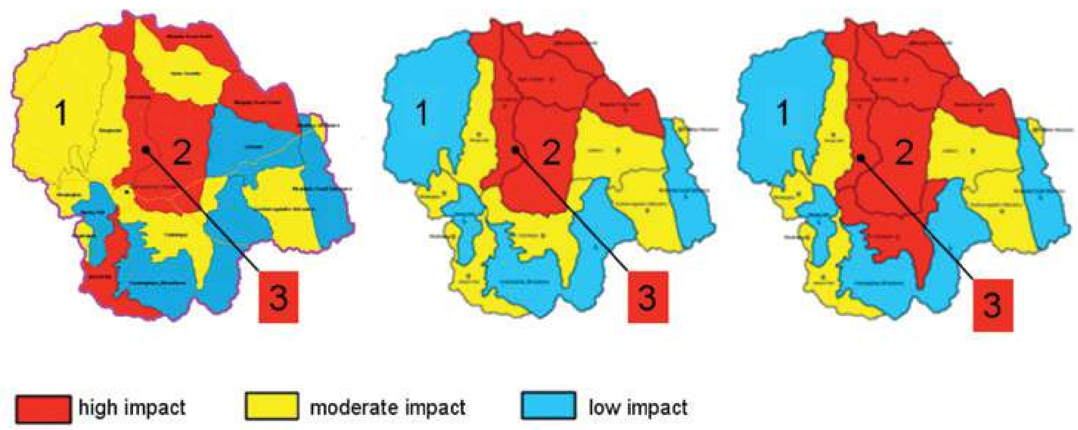



**Figure 4 - Land, Load and EC (from left to right) showing the marked difference in salinity response across the Jugiong catchment (1 Wombat HGL, 2 Tiverton HGL and 3 Currawong HGL).**

A number of HGLs to the south and west have better water quality and less land salinity. These tend to relate to sub-catchments that are in steep granite areas, or on the Mountain Creek Volcanics (interbedded rhyollite and sediments). The Tiverton HGL in the centre of the study area is marked by significant areas of Spike Rush *(Juncus acutus)* in the mid slope area of the landscape, that are wet, salty and yield significant salt load (Figure 4). The regolith is thinner and less influenced by parna, a potential source of salts, than for the Wombat (1) HGL (Figure 1). In the Wombat HGL the salinity is around the plateau landform, and in the drainage lines. This is not well connected to local groundwater systems. In contrast the Tiverton (2) HGL has high salt levels that are well connected to both surface and groundwater.

The northern catchments, especially Spring Creek are a major source of salt and impart a saline plume further down the system. There are some very high EC readings of 6-10 dS/m emanating from the contact between the Devonian sediments and the surrounding granites. The Wombat area in the west of the catchment has land salinity, but does not appear to be connected to the groundwater salinity evident in the rest of the catchment. There is a high degree of variability in the granite landscapes. A number of catchments to the south have better water quality and less land salinity. There are other granite areas particularly in the north of the catchment that have high salinity impacts. This often occurs at or near the boundary with other geologies and landscape features such as the boundary between the granite and Devonian sediments of the Douglas Range sound in the north east of the study area. The regional fault in the Currawong (3) HGL, which runs north from the centre of the catchment, has a major impact. Even during drought, wet springs "pop up" adjacent to the fault line. Salinity is concentrated along this fault line and occurs irrespective of the rock type.

Water samples were taken from the catchment in late August 2010 as a response to the high pH and carbonates encountered on the first EC stream run. Samples were analysed for water chemistry by the University of Canberra in early September 2010. The data were related to the behaviour of the HGL and there were a wide range of water chemistry variations across the area.

**Table 1: Variation in water chemistry from selected sites in the Jugiong catchment**

| Station | Na_ mmole/L | Mg_ mmole/L | Ca_ mmole | k_mmole | so4_ mmole | Cl_mmole | $CO_3$/ $HCO_3$_ mmole |
|---|---|---|---|---|---|---|---|
| J17 | 9.63 | 4.16 | 1.08 | 0.08 | 0.70 | 5.65 | 3.68 |
| J33 | 6.73 | 3.00 | 0.93 | 0.12 | 0.37 | 3.29 | 3.12 |
| J50 | 12.43 | 6.23 | 1.29 | 0.10 | 0.43 | 8.40 | 3.52 |
| U2 | 5.89 | 2.63 | 0.98 | 0.11 | 0.35 | 2.95 | 3.12 |
| U3 | 1.16 | 0.51 | 0.39 | 0.16 | 0.09 | 0.31 | 0.88 |
| W2 | 1.55 | 0.53 | 0.42 | 0.27 | 0.10 | 0.52 | 1.00 |
| W4 | 3.14 | 5.83 | 0.52 | 0.16 | 0.31 | 1.42 | 4.80 |

Variation in water quality between sites is consistent with our other observations of differences in salinity between sub-catchments. This indicates that salinity signals are very strongly tied to specific sub catchments and that the source of salinity varies across the Jugiong catchment. The full data was plotted into piper diagrams to better understand cation-anion interactions. The data shows generally low potassium and calcium in the water sample, that anions are bicarbonates/carbonates and chlorides mostly with limited sulphates. Water chemistry in the non granite HGLs in the south east of the study area including station SW_002a are predominantly Na and Mg chlorides. The eastern stations (SW_030 and 031) south of Wombat HGL lack parna deposits and are mostly Mg and Na bicarbonates. The granite HGLs (SW_ 021 and 023) to the north yield more Mg chlorides.

## Conclusions

Detailed examination of the Jugiong catchment enabled characterisation of both the catchment's salt store and the transfer processes causing salinity to manifest in the catchment. Landscapes within the Jugiong catchment behave differently in their responses to land / load / EC due to variation in regolith depth and salt stores, underlying geology, soils, landform and the interaction of groundwater and surface water. This research illustrates that different fluid flow and salt flux pathways operate in different landscapes across the catchment. The range and combination of pathways and fluxes in each landscape must be understood to formulate effective management strategies and actions to address salinity.


## Acknowledgements

Brian Murphy NSW OEH for water chemistry review and Kathleen Harvey, Murrumbidgee CMA for piper diagrams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# Soil micromorphology: Soil degradation processes within a drained coastal acid sulfate soil

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Soil micromorphological studies is a powerful tool to help understand complex soil processes in degraded landscapes. Microscopic features in acid sulfate soils can assist with estimating contemporary and relict oxidation rates and acidification potential, which is critical for their management. In this paper, soil micromorphology and scanning electron microscopy (SEM) were used to characterise and describe micro-scale weathering pathways and mechanisms that occur under changing hydrological, physical and biogeochemical conditions within a degraded an acid sulfate soil landscape at Gillman, South Australia. The soil profile developed in a supratidal regime but tidal inundation was totally excluded in 1935.
The profile classifies as Sulfuric, Salic Hydrosol (Isbell 2002) or Typic Sulfaquepts (Soil Survey Staff 2010). Sulfide oxidation was evidenced by the formation of iron oxyhydroxide pseudomorphs (goethite crystallites and framboids) after pyrite, jarosite and gypsum crystals. The morphology of pseudomorphs after pyrite formation can be used as a palaeo-indicator of the soil's physio-chemical conditions prior to oxidation. The micromorphology of weathering features of pyrite crystals, framboids and mottles provided insight into the pH and redox conditions (i.e. within micro-environments) and weathering rates as oxidation proceeded. Iron oxide pseudomorphs after jarosite spheroids within salt efflorescences also indicated dynamic oxidising micro-environments. The preservation of pyrite crystals within sulfuric material occurred due to armouring with clay coatings and by impregnation or inclusion within gypsum and halite crystals. Armouring of pyrite in this hypersaline environment was prevalent and has implications for characterisation of acid sulfate soil types by ageing as well as for their management.

# Tuesday 4 December

## THE LIVING SOIL

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# Estimating the relative contributions of edaphic properties and microbial communities to ecosystem functioning

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[2] Swedish University of Agricultural Sciences, Institute for Microbiology, Uppsala, Sweden

Linking changes in microbial communities to functional properties of ecosystems is one of the most difficult challenges facing microbial ecologists. Here, we use a model-based approach to partition variation in denitrification potential in a survey of two Swedish agricultural fields among variables associated with the abiotic and biotic environment, including denitrifier abundance, diversity, and community composition. The approach classifies variables to categories (abiotic and biotic; community-level traits, trait distributions/functional diversity, and idiosyncratic species effects) and uses model-averaging to estimate the importance and effect size associated with these variables and classes. Nitrate-N concentration was the most important single predictor of denitrification potential; however, the inclusion of biotic variables (particularly the evenness of nirS communities, the abundance of one nirS sequence type, and the number of nirK copies) had a greater effect on model precision than the inclusion of abiotic variables (nitrate-N and soil penetration resistance). These results suggest that ignoring aspects of microbial functional diversity can greatly reduce the precision of predictive models linking environmental parameters to ecosystem properties. The approach described here represents a valuable tool for enhancing the predictive power of ecosystem models and for explicitly linking microbial communities to ecosystem functioning.

# Carbon and nitrogen dynamics in grassland soils as affected by adoption of perennial Kikuyu

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The adoption of summer active $C_4$ perennial grasses such as kikuyu (Pennisetum clandestinium) in temperate areas of Australia has generated considerable interest due to their ability to persist in seasonally water deficient environments and in poorer soils, and expected longer utilisable period for grazing. Plant functional traits such as the extensive rooting systems of kikuyu have been suggested to increase below-ground carbon (C) storage. However, this increased rooting may also confer a greater level of competition for nitrogen (N). Greater rhizodeposition of C, and suggested increases in plant N uptake may result in microbial N limitation. We used six paired sites on the Fleurieu Peninsula, SA, to investigate three aspects of this complex interaction: 1) the extent to which kikuyu-C was present in soil C fractions, 2) the effect of kikuyu on N availability and turnover, 3) the resultant effect on the soil microbial nitrifier communities. Total soil organic carbon storage to 30 cm was significantly greater in sites sown to kikuyu, as was soil solution nitrate, ammonium and free amino acid concentrations. Kikuyu also appeared to stimulate peptide- and amino acid-N turnover in the top 20 cm of the soil. The relationships between altered C and N pools and differences in the community structure of ammonia oxidising bacteria and archaea are discussed. We conclude that the introduction of kikuyu had significant effects on the chemistry and microbiology of soils to which it was introduced, most likely due to its increased rooting density, and associated nutrient uptake and rhizodeposits.

# Is soil biodiversity important in production lands in New Zealand?

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This paper evaluates the significance of the biological component of soils in production lands, considering the assemblages of soil rhizospheres, soil fauna and microbial communities. The native soil biota is relatively poorly described in New Zealand although, for example, original macrofaunal assemblages are known to have been readily and heavily compromised in the transition to agriculture; many native species of earthworms were marginalised, eventually to be almost entirely replaced by just a few European species of Lumbricidae. However, is there any evidence that this loss of native biodiversity is symptomatic of declining soil quality? Whilst most biodiversity is below ground, linkages between biodiversity and functionality are only weakly defined. This is particularly surprising in view of the obvious visible differences in soils with differing biological integrity. We question whether and how it is possible to demonstrate that less modified soils provide any more beneficial ecosystem services. Focusing particularly on the morphological variability of native root systems and the ecology of native and exotic earthworm communities, we show the results of field experiments that have investigated the chemical composition of soil pore water collected in variable soil profiles. These findings potentially have relevance to soil ecosystem services, in termas of soil carbon storage, drainage water quality and nutrient cycling. A wider context is the protection of native biodiversity, the value of native plants in riparian barriers and field margins, improved cultural landscapes and the possibility of 'added value' in production landscapes through more consideration of soil ecology.

# Developing a national soil research, development and extension strategy

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In March 2012, the Primary Industries Standing Committee agreed to the development of a national, cross-sectoral soil research, development and extension (RD&E) strategy under the National Primary Industries RD&E Framework. The strategy is currently being developed in partnership with Australian, state and territory governments; CSIRO; rural research and development corporations; and the industry, tertiary education and consultancy sectors. The strategy will provide a framework for better collaboration across sectors and research and extension organisations; and aims to improve the efficiency and effectiveness of national soil RD&E investment by prioritising actions and avoiding duplication of effort. Current areas of focus include developing to a process for co-investment; finding better ways to manage physical infrastructure such as laboratories, equipment, long-term field sites and soil archives; improving availability, quality and access to soil data and information; clarifying the roles and responsibilities of organisations working in soil science and extension; finding better ways to communicate the outcomes of soil R&D; and building the skills and capacity of those delivering this work. As a result, it is envisaged that Australia will have world class soil RD&E that meets the needs of end users enabling increased on-farm productivity and enhanced soil security. This presentation will introduce the wider soil science community to the strategy, report on progress to date and seek the ideas and views of this important stakeholder group.

# Tuesday 4 December

## THE LIVING SOIL

*Presented Posters*

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# Effects of climate change on gross nitrogen dynamics in grassland soils

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The climate is warming, mainly due to anthropogenic $CO_2$ emissions. Terrestrial ecosystems dampen the rise in atmospheric $CO_2$ by sequestering C, but there is concern about the persistence of this terrestrial C sink, as elevated $CO_2$ may lead to a progressive N limitation (PNL). Plant N availability is dependent on soil N cycling processes, particularly on the mineralization of organic N, but this has not been considered in the PNL concept. Increased gross N mineralization under elevated $CO_2$ may alleviate PNL, as more mineral N is made available for plant uptake. Here we investigated gross N dynamics, by $^{15}N$ labeling, in two grassland free air $CO_2$ enrichment (FACE) experiments.

The NZ-FACE is the only grazed FACE experiment world-wide and there are indications for PNL. We investigated gross N dynamics after 14 years of $CO_2$ fumigation and compared results to findings from an earlier $^{15}N$ experiment. Specifically we tested the hypothesis that the response of gross N dynamics to elevated $CO_2$ will change over time.

Climate change experiments mainly manipulate one climatic factor only, but there is evidence that single-factor experiments do overestimate the effects of climate change. Therefore, we investigated how simultaneous increases in $CO_2$ and temperature affect gross N dynamics with soil collected at the TasFACE, in which a native Tasmanian grassland is exposed to elevated $CO_2$ and to elevated temperature in a two-factor design. Specifically, we tested the hypotheses that elevated $CO_2$ and warming have interactive effects on soil N dynamics.

# Soil biota under seeded belts: What is driving the system?

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Many agricultural in Australia have been subject to extensive clearing, contributing to reduced levels of native biodiversity and land degradation. There is now an increased awareness of the need to rehabilitate to provide essential ecosystem services that have been altered due to extensive agriculture. Native seeded belts or shelter belts have been implemented extensively to provide ecosystem services. However, there is little information on the effect of such belts on soil biotic diversity and function. Soil biota are directly and indirectly involved in key ecosystem processes, playing a major role in determining nutrient cycling, decomposition and energy flow. They can rapidly respond to land use change or management. This change in soil biotic community structure can determine the rate of decomposition and carbon uses efficiency. Restoration needs to consider soil biota when assessing the effectiveness of restoration techniques as the system is not complete until the full complement of soil biota is present and functioning. My project will aim to quantify the affect that seeded belts are having on soil biotic communities and function. The objective is to obtain a better understanding above ground and below ground linkages, in particular the effect of acacia (legume) and eucalypt (non legume) trees in restored landscapes and the influence of soil biotic community's in carbon dynamics. A greater understanding of soil biological processes in revegetated sites can advance restoration success and help the development of management practices which promote the beneficial attributes of soil organisms, essential for sustaining ecosystem function.

# Is there niche separation of archaeal and bacterial nitrifying populations in semi-arid soil?

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For Sustainable N cycling in agricultural systems requires N supply matches plant demand while N losses from nitrate leaching and gaseous losses are minimised. This study investigated the nitrification pathway in agricultural soils under different management practises in the Western Australian grain-belt using molecular and isotopic techniques. At one study site, the soil had been amended with or without organic matter on a regular basis since 2003; while the second study site the soil had been amended with or without lime in 2009. Soil was sampled at 0–2.5 cm, 2.5–5 cm, 5–-7.5cm, 7.5–10 cm, 10-–20cm and 20–30cm from replicated plots at each study site. DNA was extracted and ammonia oxidising archaea (AOA) and ammonia-oxidising bacteria (AOB) abundance was determined by qPCR. At both study sites, AOB gene abundance decreased with depth whilst AOA abundance increased with depth. Nitrate ($NO_3^-$) and ammonium ($NH_4^+$) traces at both sites followed the trend of the AOB abundance and also decreased with depth. Determining where nitrifying micro-organisms proliferate within the soil profile may enable the development of management strategies that aid in the retention of N in agricultural soils for plant uptake, minimising losses via leaching and gaseous emissions.

# X-ray computed tomography of large intact columns of a texture-contrast soil: Macro-porosity before and after root growth

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Studies to understand soil structure, root growth and root-soil interactions in undisturbed conditions have long been limited by the difficulty in 'seeing' into the opaque medium of soil. In this study the interaction between plant root growth and macro-porosity was investigated for a texture-contrast soil using the annual species canola, the herbaceous perennial lucerne and the woody perennial saltbush. The hypothesis was that perennial and/or woody species may be inherently better adapted to penetrating the dense clay dominated B horizon characteristic of a texture contrast soil, which will increase macro-porosity, whereas, the annual species may be more reliant on utilising pre-existing macro-pores. Medical x-ray CT was used to characterise in 3D the macro-porosity of relatively large (150mm diam. 500mm depth) undisturbed columns of a texture contrast soil, and further to visualise changes in that macro-porosity following root growth by the three species grown for twelve weeks. The total amount of water added was equivalent to 262mm and applied in a pattern to mimic seasonal rainfall in a Mediterranean type climate for the period May to September. Grey scale images of the intact soil cores, reconstructed from the medical X-ray CT scans, showed that the interface between the sandy A horizon and the clay B horizon was composed of a layer of soil much denser than that of either the A or B horizon, largely due to saline sodic crust development. Macro-porosity in the intact soil columns was significantly altered by root growth, although there were no observed differences between the changes made by canola, lucerne or saltbush. Cracking in the clay layer and at the interface with the sand occurred as the columns dried out. The visualisations identified differences between species in the architectural development of macro-root (>8mm diam.) systems, showed the exploitation of pre-existing soil pores by the developing root systems prior to penetration of dense soil matrix which created new macro-pores. Clear evidence was provided that saltbush was potentially the best candidate as a primer plant for duplex soils by penetrating deeper and faster.

# Microbial dynamics in forest and sugarcane soils of Queensland

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Sugarcane cropping systems in Australia suffer from a number of soil biological constraints, which reduce yields and profitability. These constraints are collectively known as 'yield decline', and are the result of continual intensive monoculture which is common in most sugarcane growing regions. Previous research has identified reductions in soil microbial biomass as a result of long term sugarcane monoculture, as well as increases in populations of pathogenic fungi and nematodes. Yet, the cause of 'yield decline' remains elusive and knowledge of soil biology scant. Here, we investigated dynamics and activities of microbial communities in response to plant litter additions in long-term sugarcane soil and a nearby forest remnant. We used laboratory-based soil microcosms and a series of analyses including enzyme assays and community level physiological profiles (CLPPs). We hypothesised that, compared to forest soil, the microbial community in sugarcane soil has reduced microbial activities and a reduced capacity to degrade complex litter. We found that, while microbial biomass was lower in the sugarcane soil, it increased very rapidly in response to litter addition. Similarly, enzyme activities were decreased in sugarcane soil, however $CO_2$ respiration in response to litter and CLPP substrates was higher than the forest soil. These findings suggest contrasting microbial community dynamics and microbial strategies in the different soils, and also highlights the importance of careful method selection when comparing soil biological communities in contrasting land use types. Additionally, microbial nitrification of ammonium was significantly faster in the sugarcane soil, which has implications for nitrate runoff to the Great Barrier Reef.

# Tuesday 4 December

## SOIL CARBON AND CLIMATE CHANGE (CHEMISTRY/BIOLOGY)

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# Quantifying the allocation of soil organic carbon to the biologically resistant fraction

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Soil organic carbon (SOC) is composed of a wide range of different materials. Quantifying the allocation of SOC to labile, stable and resistant forms is important to assess the vulnerability of SOC to subsequent loss. Allocation of SOC to particulate (50 – 2000 µm particles), humus (<50 µm particles) and resistant (polyaromatic structures <2000 µm) enabled the substitution of conceptual pools of SOC with measurable SOC fractions within the FullCAM carbon cycling model. Identification and quantification of the allocation of SOC to resistant polyaromatic carbon structures is of particular importance because of its inherent biological stability. In the Soil Carbon Research Program, a methodology based on the use of solid-state $^{13}C$ nuclear magnetic resonance (NMR) was developed to quantify the allocation of SOC to the resistant carbon fraction. Although NMR is well suited to quantify the chemical structure of most forms of SOC, quantifying the content of resistant polyaromatic carbon using the typical cross polarisation NMR methodology was challenging because of its low detection efficiency. A more quantitative, but also more time consuming, direct polarisation NMR methodology was used to obtain quantitative estimates the amount of resistant polyaromatic carbon and to develop a correction factor that could be applied to the more rapid cross polarisation NMR analysis. Results obtained indicated that resistant polyaromatic forms of SOC can account for 0 to >70% of the carbon in both the 2000-50µm particulate organic carbon and <50µm humus fractions. Accurate allocation of SOC to the resistant polyaromatic form is critical to accurately model soil organic carbon dynamics and responses to agricultural management practices.

# Fingerprints of organic matter in soils of New Zealand

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We elucidated the structural and molecular chemistry of organic matter in an extensive range of soils from North and South Islands of New Zealand using solid-state cross polarization magic angle spinning $^{13}C$ nuclear magnetic resonance (NMR). It revealed clear differences in the structural composition, which included O-alkyl, alkyl, O-aromatic, aromatic, and carboxyl carbon, as seen by $^{13}C$ NMR in whole soils from different regions. Generally, N-and O-alkyls and acetals dominated in almost all soils, while alkyl C was the second and aromatic C (aryl + O-aryl) the third quantitatively most important C types seen by $^{13}C$ NMR. Overall, the proportion of O-alkyl C in these soils varied from 23-60%, alkyl C 18-47% and aromatic C ranged from 9-22%. The carboxyl C was the least abundant group as revealed by the NMR. The abundance of O-alkyl C in the samples was likely due to the large concentration of carbohydrates from simple sugars to starches and cellulose in majority of soils under pasture. The NMR spectrum of one soil from the North Island was conspicuously different from the other soils studied. It showed very strong signal for aliphatic C. The molecular composition, which included protein, carbohydrates, lignin, charcoal, carbonyl and aliphatic components, as established using the elemental composition and $^{13}C$ NMR data in a molecular mixing model also showed substantial variations in the molecular nature of organic matter with some soils having appreciable amounts of char and lignin components. This has implications in the risk assessment of xenobiotics.

# Structures of allophane nanoaggregates and the characteristics of carbon and DNA in buried allophanic soils, New Zealand

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Allophane comprises tiny (3.5–5 nm), Al-rich spherules with extremely large surface areas up to 1500 $m^2g^{-1}$ that form strong associations with organic matter and tend to form nanoaggregates up to 100 nm in size. Allophanic tephra-derived soils (Andisols) contain the largest concentration of sequestered organic carbon after organic soils (Histosols), but the reasons why and how allophane is able to hold so much carbon strongly is still unknown. The clay minerals possess sorptive capacities for DNA also, and so allophanic soils are potentially capable of protecting DNA for thousands years. The buried allophanic soils in North Island developed on dated tephra layers provide a paleobiological “laboratory” for studying the preservation of ancient DNA and organic carbon as well. Synchrotron-based facilities have enabled the integrated study of characteristics and spatial variability of organic carbon at micro- or nano-metre scale, and hence synchrotron radiation provides a promising tool to investigate carbon and DNA in allophanic soils at nano-metre-scale resolution.

Our study aims to investigate the mechanisms of organic carbon preservation in allophanic soils by characterizing the forms of carbon and particularly DNA and their associations with allophane. This aim will be achieved through a combination of synchrotron-based IR microscopy and near-edge X-ray absorption fine structure (NEXAFS) of C, N, and P. We have been granted beamtime for X-ray absorption spectroscopy, XPS, FTIR, and X-ray microscopy at the National Synchrotron Radiation Research Center in Taiwan during June and July 2012, and the results will be processed and presented before the end of 2012.

# Soil respiration rates in red pine forests as affected by temperature and fertilizer

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## Abstract

This study was conducted to evaluate the dynamics of soil respiration following forest fertilization in red pine forests in Korea. Different combinations of fertilizer (N:P:K=113:150:37 kg/ha; P:K=150:37 kg/ha; control), which reflect current practices, were applied in April 2011 and total soil respiration rates (autotrophic and heterotrophic respiration) were monitored from April 2011 to March 2012. Temperature was the dominant factor controlling soil respiration rates. Rates of total soil and heterotrophic respiration were similar among treatments in months. Annual total soil respiration rates were not significantly different between fertilizer (0.474-0.504 g $CO_2$/ $m^2$/h) and control (0.500 g $CO_2$/ $m^2$/h) treatments. Heterotrophic respiration rates were 78% of total respiration in the PK treatment, 74% in the NPK treatment, and 63% in the control. These results suggest that current fertilizer practices will not substantially affect soil respiration rates.

## Key Words

Soil $CO_2$ efflux, fertilization, autotrophic respiration, heterotrophic respiration, pine forest

## Introduction

The quantitative evaluation of soil carbon dioxide ($CO_2$) efflux after fertilizer application in forest ecosystems is a key process for understanding carbon dynamics. However, the effect of fertilization on soil respiration rates has received little attention, with conflicting results, in that reports have suggested both an increase and a decrease in soil respiration rates. For example, Gallardo and Schlesinger (1994) found an increase in soil respiration when nitrogen was added to forest soils in central North Carolina, while soil respiration was significantly lower for fertilized than for unfertilized plots because of reduced root respiration in red pine plantations in Northern Wisconsin, USA (Haynes and Gower 1995). Lee and Jose (2003) reported that nitrogen fertilization had a significant negative effect on soil respiration in a cottonwood plantation, but no effect was observed in loblolly pine plantations.

Since soil respiration results from two main sources, root respiration (autotrophic respiration) and the microbial decomposition of organic matter (heterotrophic respiration), these conflicting reports could be the result of fertilizer-induced differences in carbon fixation and allocation patterns among different tree species, and decomposition of organic matter (Raich and Tufekcioglu 2000, Lee and Jose 2003). In contrast to total soil respiration, heterotrophic respiration was reduced due to 20 years of N and NPK applications (Franklin *et al*. 2003), while autotrophic respiration would be expected to increase along with an increase in forest production after fertilization. However, autotrophic respiration could potentially decrease if the trees decrease the allocation of carbon to roots in response to higher nutrient availability (Giardina *et al*. 2003).

Red pine (*Pinus densiflora* S. et Z.) forests are the most important type of coniferous tree species in Korea and occupy more than 23.5% (1.5 million ha) of Korean forest lands (Korea Forest Service 2006). Despite the progress made in quantifying the carbon balance of many coniferous forests in Korea, little is known about the underlying relationships between soil $CO_2$ efflux and environmental factors, which might change in response to fertilization. The objective of this study was to determine the effects of fertilization on the relationships between autotrophic respiration and heterotrophic respiration in red pine stands.

**Methods**

The study was conducted in the Wola National Experimental Forest. The soil is a slightly dry, dark-brown silt loam forest soil (mostly Inceptisol, United States Soil Classification System) originating from sandstone or shale. The site index indicates low forest productivity (site index 8-10 at 20-year-old base age) suggesting poor soil fertility. The treatment plots were established on slopes with the same aspect in a completely randomized design with 2 blocks involving three treatments [total 18 treatment plots (3 treatments (NPK, PK, Control) ×2 blocks×3 replicated plots)] in mature red pine stands (plot size = 10 x 10m). Fertilizer (N: P: K=113:150:37 kg $ha^{-1}$) application was based on the guidelines of forest fertilization in Korean forests (Joo *et al*. 1982). Fertilizer (N: P: K=113:150:37 kg $ha^{-1}$), PK (P: K=150:37 kg $ha^{-1}$) was applied manually in April 2011.

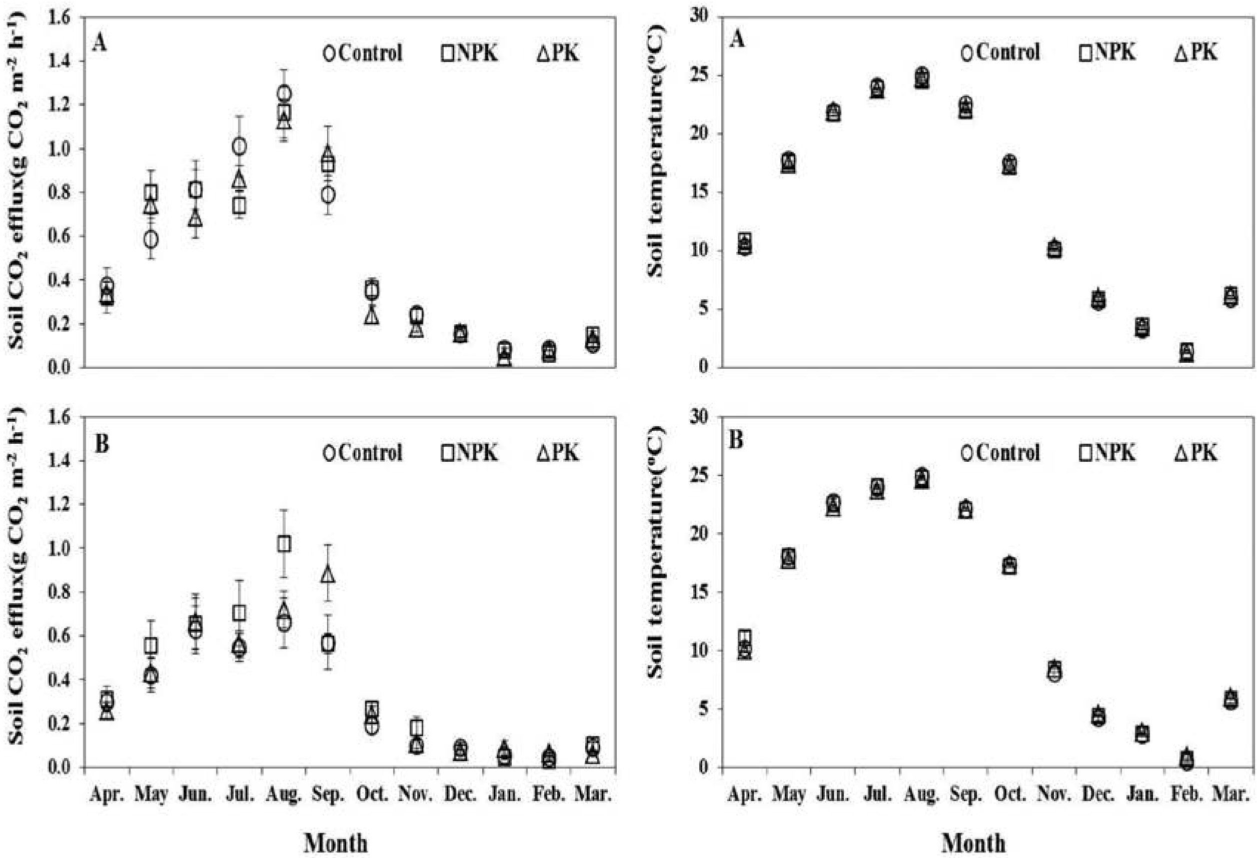


**Figure 1. Monthly variations of total soil (A) or heterotrophic (B) soil respiration and soil temperature by different combination of fertilizer in red pine forests. Bars represent one standard error (n=12).**

Mean total soil respiration rates during the study period were not significantly different among the combinations of fertilizer, although the rates were slightly higher in the N:P:K (0.504 g $CO_2/m^2/h$) and control (0.501 g $CO_2/m^2/h$) than in the P:K (0.474 g $CO_2/m^2/h$) treatments (Figure 2). However, contrasting data on the effect of fertilizer on total soil respiration rates in forest ecosystems have been reported. For instance, total soil respiration rates on the forest ecosystems can be affected (Lee and Jose 2003) or unaffected (Kim 2008) by forest fertilizer applications. Total soil respiration rates after fertilization can decrease (Haynes and Gower 1995), increase (Gallardo and Schlesinger 1994) or fertilizer application can have no discernible effect (Kim 2008). The effect of forest fertilization on soil respiration rate could be attributed to the change of the conditions for the decomposition of organic matter and root respiration, which are the two main sources of soil respiration. For example, soil environmental change after fertilization is an important variable affecting soil $CO_2$ efflux rates because it is closely related to microbial biomass activity and nutrient availability (Lee and Jose 2003, Kim 2008). Annual mean total soil respiration rates during the study period were significantly higher (0.49 g $CO_2/m^2/h$) than heterotrophic (0.35 g $CO_2/m^2/h$) respiration, probably because root respiration rate was reduced by trenching.

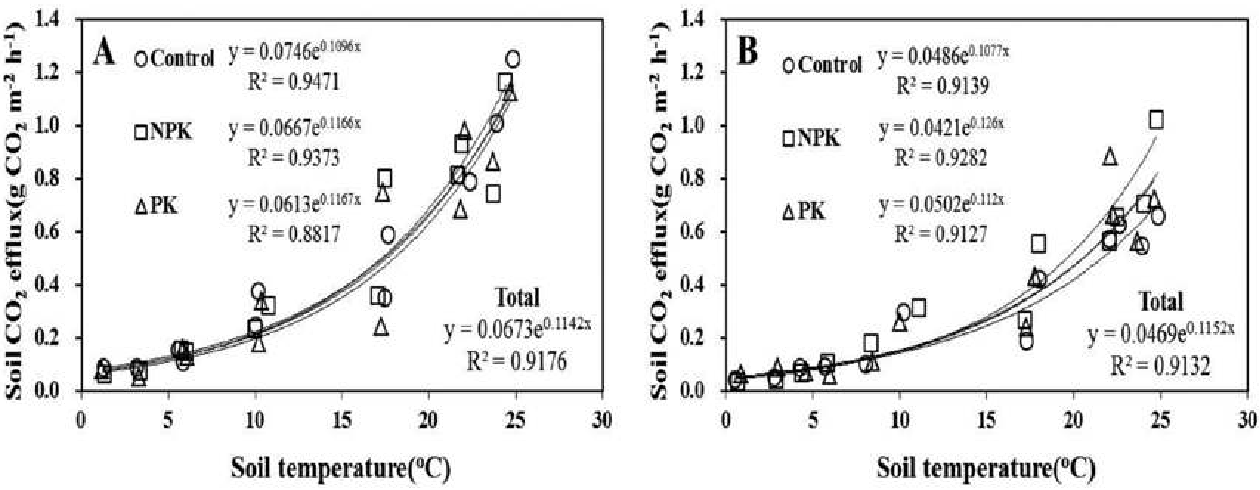


**Figure 3. An exponential regression of total soil (A) and heterotrophic (B) soil respiration rates against the corresponding soil temperature at a depth of 8 cm.**

## Conclusion

Fertilizer applications induced inconsistent changes in monthly rates of total soil and heterotrophic respiration rates, while annual total soil respiration rates were little affected by the change of nutrient availability with different combinations of fertilizer in red pine forests. Regressions of total soil or heterotrophic respiration rates against soil temperature were similar for different fertilizer treatments, and showed high sensitivity to soil temperature.

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# A long-term study of biochar carbon stability and its priming potential in clay soil

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The stability (mean residence time, MRT) of biochar carbon (C) is the major determinant of its value in long-term C sequestration in soil. However, biochar C stability and its priming effect on soil C are not well understood. A long-term (5 year) incubation experiment was conducted using the natural 13C-isotope difference between biochar and soil C. Biochars from C3-biomass sources (*Eucalyptus saligna* wood and leaves, paper sludge, poultry litter, cow manure) produced at 400 ºC and 550 ºC were applied to a vertosol containing C4 organic matter. Between 0.5% and 8.9% of the applied C was mineralised over 5 years. The C in manure-based (poultry litter and cow manure) biochars mineralised faster than that in the plant-based biochars (wood and leaves), and C in the 400 ºC biochar mineralised faster than that in the corresponding 550 ºC biochars. The MRTs of C in biochars varied between 90 and 1600 years. These are conservative estimates because they are skewed towards the MRT of labile and intermediate biochar C components, which are preferentially mineralised over the first 5 years. Furthermore, biochar C mineralisation rate in the laboratory may be considerably faster than under field conditions. While the 550 °C wood biochars were highly stable (MRT ~1300-1700 years), a significant proportion of C in other biochars is also likely to remain in the soil for >100 years, i.e. the timescale considered "permanent" for emissions trading. Biochars, especially those from manure feedstocks, stimulated native soil C mineralisation in the short-term (up to 2 years). Biochar-soil interactions over time caused stabilisation of native soil C. We found strong (R2 = 0.87–0.95), non-linear, relationships of cumulative biochar C mineralisation and MRT with the proportion of non-aromatic C and the degree of aromatic C condensation of biochars, thereby suggesting the use of these measures as ready predictors of biochar C stability in soil.

# Tuesday 4 December

## SOIL FERTILITY AND SOIL CONTAMINANTS (SOIL TESTING/NEW APPROACHES)

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# Applicability of Diffusive Gradients in Thin-films (DGT) to measure short term sulphur availability in agricultural soils

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Potential for sulphur deficiency to become prevalent in broad acre agricultural soils in Australia is increasing due to relatively low application rates in fertilisers and substantial removal in harvested products. It is debatable whether currently available soil tests methods provide accurate assessment for available S. Thus, new technology may offer an alternative approach. Diffusive gradient in thin films technology (DGT) has been successfully applied to assess P availability in many Australian agricultural soils where conventional methods were unreliable in defining a P pool that correlated with crop P uptake and response to fertiliser. This paper reports on the development of DGT for measuring available S in Australian soils. The performance of the selected binding agent was tested, in simple solutions varying in pH (3-9), with the agent demonstrating high affinities and large sink capacities S and also P. Poor to moderate correlations were obtained (R2 = 0.11-0.56) between DGT –S and established soil tests for available S using a range of soils from New England region, NSW. However, when the soil S test methods were compared with short-term maize S uptake and response to S applications under glasshouse conditions, DGT had the greatest accuracy (R2 = 0.55 (relative yield), 0.80 (uptake) compared to resin S (R2 = 0.48, 0.57), KCL-40 (R2 = 0.34, 0.57) and MCP (R2 = 0.4, 0.29). Further glasshouse and field studies are required to assess the full benefits of using DGT as a tool for measuring plant available S in Australian broad acre agriculture.

# Soil P tests for evaluating P availability in Vertosols and Dermosols of the northern Australian grains region

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Vertosols and Dermosols comprise ~7Mha of the northern grains region (central and southern Queensland and northern NSW). The rainfed cropping history of these soils has resulted in rundown of available P, particularly in the subsurface layers. A P depletion glasshouse trial (nil and 50 kg P/ha) was undertaken on 17 soils using forage sorghum as the test crop. The sorghum was sequentially harvested on 5 occasions (re-planted after harvest 3) and relative dry matter yields and cumulative P uptakes measured. By the final harvest, plants in 9 soils were unable to grow because of extreme P deficiency. Colwell-P (0.5 M $NaHCO_3$), BSES-P (0.005 M $H_2SO_4$), soil solution P (at field capacity) and DGT-P were determined prior to planting and following final harvest (except DGT-P). Initial values of all tests except BSES-P were significantly (*$P<0.001$*) correlated with relative yield (nil P/plus P) at harvest 1 although there were obvious outliers with higher relative yields than would be expected from their soil test values: Colwell-P, 0.90; DGT-P, 0.89; soil solution P, 0.86; BSES-P, 0.19. When cumulative P uptake data from all soils over the 5 harvests were correlated with change in Colwell-P (r=0.90), there were 4 outliers with higher P uptakes than expected for the change in Colwell-P. However, when cumulative P uptake by all soils was plotted against change in BSES-P, the outliers fell on the same regression line as the other soils (r=0.98). These results confirm that sparingly soluble P mineral sources contribute to available P in these soils.

# Prediction of phosphorus sorption and availability in European soils using mid and near-infrared diffuse reflectance spectroscopy

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The development of techniques for the rapid, non-costly and precise determination of phosphorus (P) sorption and availability in soil samples is crucial in terms of increasing the efficiency of added P to crop plants. Thus, fertility is increased and fertilizer inputs and pollution problems related to the transport (runoff or leaching) of excessive quantities of P are reduced. This paper describes the development of partial least squares (PLS) regression models for the prediction of isotopically available P (labile P determined using $^{32}P$) and P sorption in soils using mid and near-infrared (NIR and MIR) spectroscopy. For the development of the models, 500 calibration soils (covering the spectral variation of the full sample set) were selected from a set of 4813 soils of the GeoSurveys Geochemical Mapping of Agricultural Soils and Grazing Land Soil of Europe (GEMAS) project. The optimum models were applied to the prediction of P sorption and availability in the remaining 4313 unknown soils and a European map for the prediction of these variables was developed. This study has demonstrated the potential of the infrared PLS method for the prediction of P sorption and labile P in a range of European soils.

# Chemical speciation in soils: What spectroscopists don't always tell you (but should)

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Various spectroscopic techniques are widely used to determine the speciation of many elements in soils. However, there are systemic issues with this approach that appear to be unappreciated by both spectroscopists and end-users alike. Spectroscopic analysis of a complex material like soil is challenging and it is all too easy to fall into common traps. This paper identifies three of these issues (overlapping signals, ambiguous assignments and non-uniform detection), discusses examples of each and provides tips on how to overcome or avoid them.

## Introduction

Chemical speciation is important in many areas of soil science, from predicting toxicity and nutrient availability to understanding element cycles and soil forming processes. In recent decades, traditional "wet" chemical analyses have been joined and, in many cases, replaced by spectroscopic techniques, including IR, NMR and most recently synchrotron-based X-ray spectroscopy. Spectroscopic techniques are based on the fact that energy levels of atoms, groups of atoms and molecules are quantised and absorb or irradiate electromagnetic radiation of a specific frequency in going from one specific energy state to another. Crucially, these energy levels vary depending on the chemical environment of the atom or group of atoms and so the frequency of radiation absorbed or irradiated carries direct information about chemical speciation. Thus we get a "spectrum" in which the intensity of irradiation or absorption is seen to vary with frequency.

It should be remembered that spectroscopic signal is exactly that – it is a measure of the variations in intensity of some form of radiation. It does not provide a direct measure of the amounts of substances in the way that, for example, weighing does. Determining amounts of species from spectroscopic signals requires:

- That separate signals can be identified and quantified where they overlap
- That the signals can be identified unambiguously
- That all species produce equivalent amounts of signals (or at least produce signals in known proportions)

This may seem obvious, but soil science is littered with examples where one or more of these criteria is not satisfied, resulting in incorrect or unreliable speciation. This paper shows some examples and discusses what can be done to avoid or remedy these problems.

## Discussion

*Overlapping signals*

Figure 1 shows part of a solution $^{31}$P NMR spectrum of a soil extract (reproduced from Turner *et al.* (2003)). It clearly contains numerous overlapping signals. In order to quantify the amount of signal in individual peaks, the spectrum has been analysed using *deconvolution*, a mathematical procedure that finds the linear combination of multiple simple peaks that gives the "best-fit" to the spectrum. Clearly a good fit is achieved, but how quantitatively reliable is the data derived in this way?

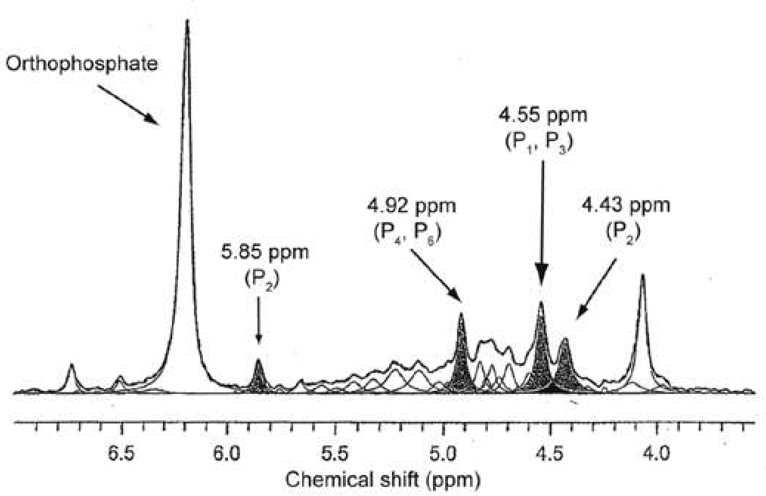


**Figure 1: (reproduced from Turner et al. (2003)). 31P NMR spectrum of a NaOH-EDTA extract of a lowland permanent pasture soil. The shaded peaks are phytate resonances that have been identified using spectral deconvolution.**

Figure 2 shows the corresponding section of $^{31}$P NMR spectra (adapted from Bünemann, *et al.* (2008)) of two "real" soils (Hart and Wagga) and two "model" soils (mixes of pure sand + clay). All four soils were incubated with cellulose for 25 weeks. In each case there are numerous sharp peaks that can be attributed to small, specific P-containing molecules. However, the "real" soils also contain a broad underlying signal that is probably due to P in large complex "humic" molecules, which contain P in a plethora of slightly different chemical environments.

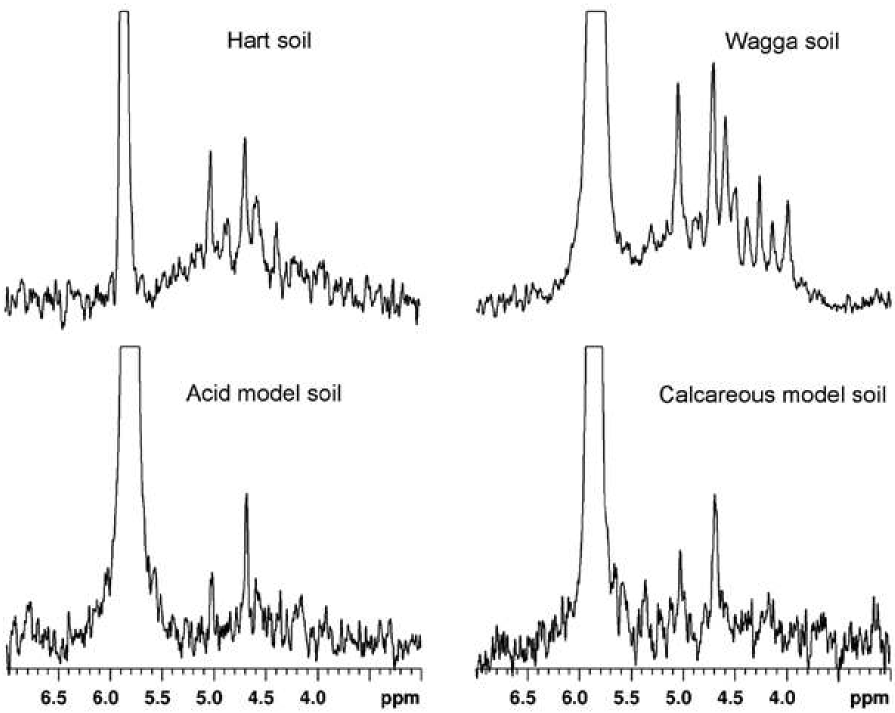


**Figure 2: (adapted from Bünemann et al.( 2008)). Expansion of the monoester region of 31P solution NMR spectra of NaOH-EDTA extracts of a calcareous soil (Hart), an acidic soil (Wagga) and acidic and calcarous model soils follwing a 25-week incubation with cellulose addition.**

Looking back at the spectrum in Figure 1, it would appear that it too contains a broad underlying signal and that fitting this broad signal as numerous small peaks may not be the best approach. Furthermore, if this is the case, then too much signal has been attributed to the sharp signals in Figure 1, i.e. the quantification has been biased.

NMR is by no means the only technique affected by this issue. Indeed the relatively sharp and simple signals of NMR spectra enable much more detailed speciation than methods such as XANES, for which signals of individual species are complex and extend across virtually the whole spectral range. As a consequence, P XANES analysis can generally only identify a maximum of 2-4 species in soil samples.

*Mis-identification of spectroscopic signals*

Obviously, accurate speciation from spectroscopic techniques requires accurate identification of spectral signals. This can be harder than it looks. Figure 3 (reproduced from Smernik and Dougherty (2007)) is again an example from solution $^{31}P$ NMR analysis of soil extracts. The point here is that what look like spectral signals of phytate in the bottom traces of each pair, were shown not to be phytate through careful spiking experiments. Great care needs to be taken when analysing such a complex material as soil to not make assumptions about composition. Furthermore, it never hurts to check the veracity of an analytical technique by adding a small amount of a known material and checking that it produces the signal where you expect and at the intensity you expect.

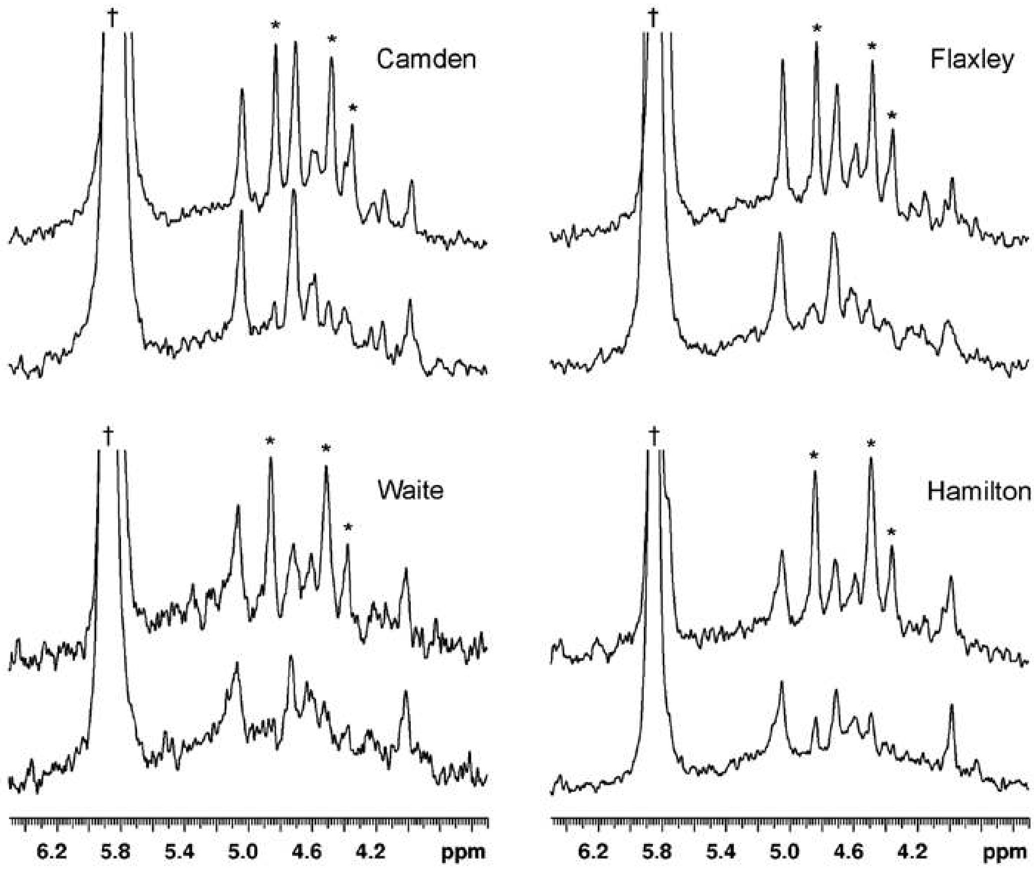


**Figure 3 (reproduced from Smernik and Dougherty (2007)). 31P NMR spectra of unspiked NaOH-EDTA soil extracts (bottom traces of each pair) and phytate-spiked (top traces) NaOH-EDTA soil extracts. Phytate resonances are marked with an asterisk (*); the orthophosphate resonance is marked with a dagger (†).**

*Quantification of spectroscopic signals*

Too often quantification of spectroscopic signal is equated with quantification of the species that give rise to that signal. This is only true if all species give rise to the same amount of signal, independent of chemical environment. For some spectroscopic techniques, such as IR spectroscopy, this is never true. For example, the C=O group gives a very strong IR signal. Therefore the presence of a strong C=O peak doesn't necessarily indicate that it is abundant. Even for a technique such as NMR, which is innately quantitative, problems still arise in soil analyses. Figure 4 (reproduced from Smernik and Baldock (2005)) shows solid-state $^{15}N$ NMR spectra of organic matter isolated from four soils. Each spectrum is dominated by one signal and that can be identified as amide N. So does this prove the vast majority of N in these soils is amide? No, because other types of N, such as heterocyclic N are poorly detected by solid-state $^{15}N$ NMR.

For example, caffeine produces about 85% less signal per unit N than protein (Smernik and Baldock 2005). *Spin counting* indicated that the spectra in Figure 4 represent only 50-84% of N present in these samples, so there remains the possibility that they contain substantial non-amide N that is undetectable by NMR.

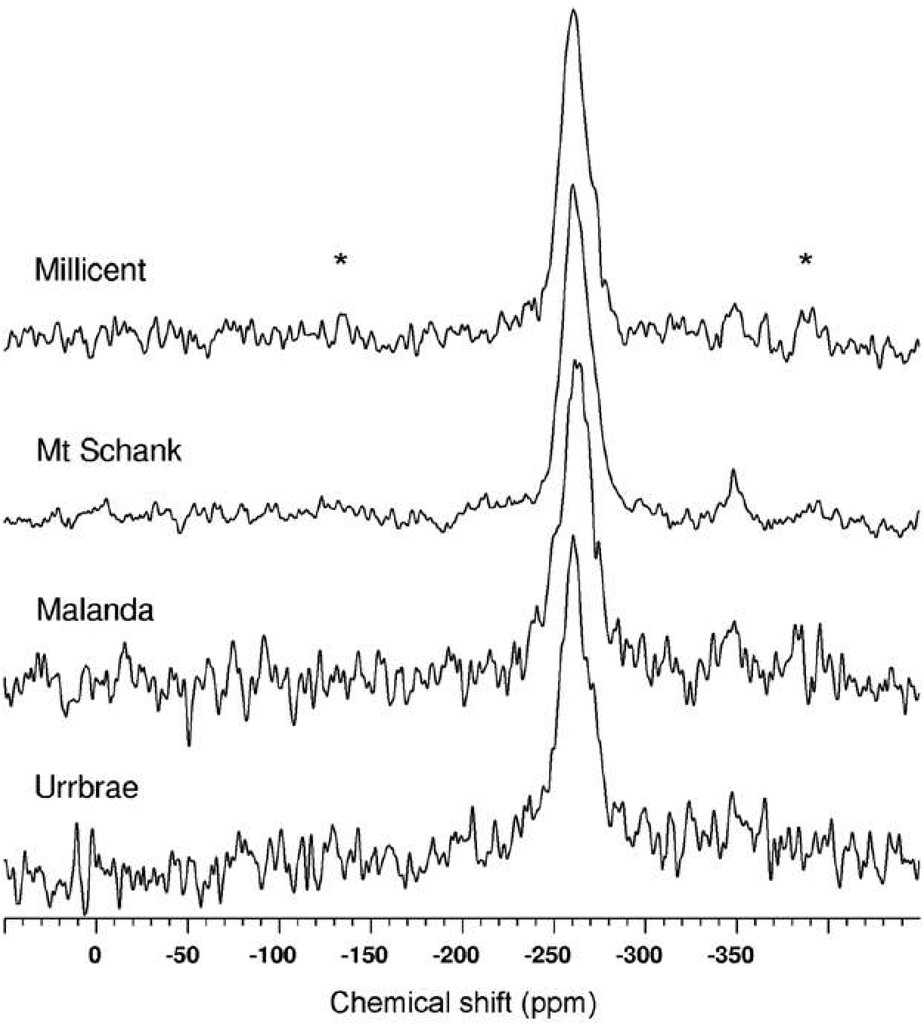


**Figure 4 (reproduced from Smernik and Baldock (2005)). Solid-state 15N CP NMR spectra of four HF-treated soil fractions.**


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# Using solid phase microextraction to identify compounds in soil water runoff from a dairy farm in south eastern Victoria

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Identification and quantification of organic pollutants in agriculture can be difficult. In this study we use solid phase micro extraction (SPME) to sample soil water extracts from a dairy farm in south eastern Victoria. Two types of SPME fibers were used for extraction. To extract compounds from the soil water extracts, the fibers were either placed in the headspace (HS-SPME) above the sample or directly immersed into the sample (DI-SPME). After extraction, the fibers were placed in a gas chromatography-mass spectrometer for analysis. Eleven compounds were identified in the soil water. Most compounds that were identified were aromatic, along with sulfides, ketones, an aldehyde and a phenol. In general, compounds were more difficult to detect with a polydimethylsiloxane (PDMS) fiber than with a CarboxenÔ-polydimethylsiloxane (CAR-PDMS) fiber. HS-SPME provided a better response when using the CAR-PDMS fiber (Average response: 1.4 ´ $10^4$ vs. DI-SPME 1.2 ´ $10^4$) while DI-SPME was better for the PDMS fiber ((Average response: 5.9 ´ $10^3$ vs. HS-SPME 3.3 ´ $10^3$).

## Introduction

Identification and quantification of organic pollutants in agriculture can be difficult, with contamination of sampling equipment and samples a particular concern. Solid Phase Microextraction (SPME) is a relatively recent technique that facilitates the extraction and pre-concentration of a sample from a mixture *in situ*. Subsequently, the collected organic molecules can be discharged into an analytical device. The SPME apparatus consists of a fiber made from silica which is coated with various materials that may have a wide range of properties for adsorbing and absorbing moieties onto or into the fiber (Heaven and Nash 2012). SPME is commonly used for headspace (HS-SPME) sampling where the fiber is placed just above the sample solution and direct immersion (DI-SPME) sampling where the fiber is placed in the solution. Once the SPME fiber has been used to collect compounds, it can be directly inserted into the analytical apparatus, usually gas chromatography-mass spectrometry (GC-MS) or liquid chromatography-mass spectrometry equipment. In the analytical apparatus, compounds are detached from the fiber and their chemical components determined. Unlike many analytical techniques, SPME can measure very low concentrations (up to pg/g) of compounds without need for extensive sample preparation.

In this study we used GC-MS to investigate the effectiveness of SPME for the detection of compounds in soil water. Two types of SPME fiber were used to extract compounds from soil water collected from a dairy farm in south eastern Victoria. Both HS-SPME and DI-SPME were used to extract the compounds from solution.

## Material and Methods

The soil water samples were collected from three different plots on a dairy farm at Poowong (38° 16’ 41”S, 145° 44’ 25” E) in the Gippsland region of Victoria. The soil water samples were taken from a field trial established in 2009 (Watkins *et al.* 2012). Three cultivated, ryegrass (*Lolium sp.*) and clover (*Trifolium sp.*) plots receiving 35 kg P/ha annually were sampled using soil corers to a depth of 2 cm. Fifty cores were recovered from each plot and centrifugation used to extract soil water.

### *SPME Fiber selection*

This study used two SPME fibers to investigate the attributes of soil water sample with a Milli-Q water sample as a procedural control sample. The two types of SPME fiber used in the study were:

a) Polydimethylsiloxane (PDMS) fiber, which consisted of a polymer of siloxane which is the base onto which other materials are often coated (PDMS SPME fiber assembly, 100 μm, Supelco, Sigma-Aldrich, NSW, Australia). PDMS is non-polar and is generally used for sampling non-polar compounds. Notably, PDMS SPME fibers have also been used in the analysis of alcohols, esters, aldehydes and terpenes (Heaven and Nash 2012).

b) CarboxenÔ (CAR) is a coating used on a PDMS base (CarboxenÔ-PDMS (CAR-PDMS) SPME fiber assembly, 75 μm, Supelco, Sigma-Aldrich, NSW, Australia). The CAR-PDMS SPME fiber coating consists of particles of carbon which act as molecular sieves. They have been used for the analyses of alcohols, ketones, acids, sulfur and furans (Heaven and Nash 2012).

*Sample processing for GC-MS analyses*

The soil water samples were collected in appropriately cleaned 100ml Schott bottles, frozen immediately (-5°C) and remained in that state until thawed prior to analysis. During the experiment, the test solutions were mixed with a Teflon coated stir bar for 20 minutes to equilibrate the solution with the headspace prior to sampling. The SPME fiber was then added to the headspace or into the solution and stirring was continued for a further 20 minutes after which the SPME fiber was removed and immediately placed into the GC-MS for measurement of the analytes. The SPME fiber was reinserted into the GC-MS after the initial measurement at least once per sample and experiment type, to estimate any carryover from previous runs (i.e. a blank run). No carryover effects were detected. Procedural blanks using Milli-Q water as the sample solution were also performed on the SPME fibers with any compounds identified considered artifacts. All experiments were run at room temperature (approximately 20ºC). All sampling and analyses were performed in triplicate.

*GC-MS details*

The GC-MS (CP8400 GC and Saturn 2200 ITMS; Varian Inc., Middelburg, the Netherlands and Walnut Creek, USA) was equipped with a 1177 split-splitless injector operated at 280 ºC. The split vent was initially at 1:20, opened to 1:80 after 0.20 minutes and reduced to 1:15 after 1.0 minute. A Varian Factor Four capillary column VF-5ms 30 m × 0.25 mm ID and 0.25 μm film thickness was used for separation using helium carrier gas pressure programmed to a constant flow (1 ml/min.). The column oven was programmed to start at 5ºC for 1 min, to increase to 250ºC at 10ºC/min and at 25.5 min to increase to 325 ºC at 30ºC/min until the end of the experiment at 31 min. The transfer line to the mass spectrometer was heated to 170 ºC and the trap was operated at 150 ºC. In MS mode, the scan range was 40-350 amu with a 0.61 sec/scan. The quantity of each compound was determined by the response of the mass detector on the GC-MS as measured by the area of the peaks (i.e. the count) on the resulting chromatogram.

Mass spectra were compared with the National Institute of Standards and Technology/Environmental Protection Agency/National Institutes of Health 2005 mass spectral library (NIST/EPA/NIH 2005 mass spectral library, Gaithersburg, MD), with all computer spectral matches (minimum $R^2 \geq 80\%$) checked manually.

**Results and Discussions**

Eleven compounds were identified that match the NIST/EPA/NIH 2005 mass spectral library at $R^2 > 80\%$ (Table 1). The compounds were mainly aromatic, with sulfides, ketones and aldehydes also identified.

Overall, the CAR-PDMS fiber was better at extracting compounds than the PDMS fiber (e.g. average response using HS-SPME for dimethyl disulfide: CAR-PDMS fiber: 5.1 ´ $10^4$; PDMS fiber: 2.4 ´ $10^3$). The average response for the CAR-PDMS was 1.4 ´ $10^4$ when using HS-SPME and 1.2 ´ $10^4$ for DI-SPME. For the PDMS fiber, the response was an order of magnitude lower with an average response of 3.2 ´ $10^3$ for HS-SPME and 5.9 ´ $10^3$ for DI-SPME. Also, the CAR-PDMS fiber extracted 10 of the 11 compounds identified while the PDMS fiber only extracted 8 compounds. This is not unexpected as the compounds identified were generally polar and so would be more likely to be extracted by a polar fiber (i.e. CAR-PDMS). As an example, the compound 2-ethyl hexanal, which has a polar aldehyde group at one end with two alkane chains at the other, was only identified using the CAR-PDMS fiber.

It was only possible to tentatively identify how one of the five aromatic compounds came to be present in the soil water extracts. The compound 2-tert-butyl-3,4,5,6-tetrahydropyridine is from a class of compounds known to produce the aroma of heated grains and grasses. It is possible then that this compound was formed in the GC-MS injector (280ºC) from a precursor compound found in the rye or clover on the plots. Why the aromatic compounds 3-methoxydiphenylmethane, 1-methyl-3-(phenylmethyl)-benzene, 2,2'-dimethylbiphenyl and 2,4,6-tris(1-methylethyl)-phenol were present is unknown but they were not detected in the procedural blanks.

Two sulfur compounds were found in the soil water with dimethyl disulfide (R.T. = 5.90 min) producing the strongest response in the soil water extract (Average count: 5.1 ´ $10^5$). The sulfur compounds were most responsive to CAR-PDMS HS-SPME with dimethyl tetrasulfide only found when using this method and fiber. These results are consistent with the use of CAR-PDMS SPME for detecting sulfur compounds (Heaven and Nash 2012). There are various reasons why sulfur compounds may have been found in the soil water samples. It has been reported that sulfur compounds are common volatile organic carbons (VOCs) found in dairy cow emissions (Shaw *et al.* 2007). Sulfur compounds are also byproducts of anaerobic microbes that are ubiquitous to soils and waters, are breakdown products of fumigants and, especially the disulfides, are very stable in soil (Arnault *et al.* 2004).

The other compounds were two ketones and two aldehydes. The compound 1-[3-hydroxybenzyl]-6-methoxy-3,4-dihydroisoquinolinecarbaldehyde is from a class of compounds called alkaloids. Alkaloids are known metabolites for a variety of bacteria, fungi, plants and animals. The other aldehyde, 2-ethylhexanal, is a scent molecule for a variety of flowers. The ketones are cyclic compounds that are often found as metabolites in both plants and animals.

In conclusion, SPME allowed for the detection of a variety of different compounds, most of them likely to be metabolites from the pasture and microbes in the soil. Only the sulfur compounds could be tentatively sourced as coming from dairy cow emissions. The CAR-PDMS fiber was found to extract these compounds better than the PDMS fiber. If the PDMS fiber should be used, it is best to directly immerse the fiber into the sample solution.

**Table 1: Tentative identity of compounds found in soil water samples as determined by GC-MS (NIST/EPA/NIH 2005 mass spectral library, Gaithersburg, MD, $R^2 > 80\%$). Methods examined included headspace SPME (HS-SPME) and direct injection SPME (DI-SPME). CAR-PDMS and PDMS SPME fibers were used. Three extractions using the SPME fiber were used to determine the average count.**

| | | HS-SPME (Average count) | | DI-SPME (Average count) | |
|---|---|---|---|---|---|
| Retention Time (min) | Tentative Identity ($R^2$>80%) | CAR | PDMS | CAR | PDMS |
| 5.90 | dimethyl disulfide | 50659 | 2396 | 9825 | 5278 |
| 9.66 | 1-[3-hydroxybenzyl]-6-methoxy-3,4-dihydroisoquinolinecarbaldehyde | -[a] | 2392 | 936 | 2094 |
| 9.86 | 2-ethyl-hexanal | 12701 | - | 30372 | - |
| 11.12 | 2-tert-butyl-3,4,5,6-tetrahydropyridine | 37711 | 9861 | 27347 | 15104 |
| 14.24 | dimethyl tetrasulfide | 2817 | - | - | - |
| 15.95 | 2,2,6,7-tetramethyl-10-oxatricyclo[4.3.0.1(1,7)]decan-5-one | 792 | - | 3239 | - |
| 17.97 | 2,4,6-tris(1-methylethyl)-phenol | 2959 | 1963 | 12389 | 5943 |
| 18.10 | 2,2'-dimethylbiphenyl | 2256 | 1314 | 1021 | 1349 |
| 18.45 | 1-methyl-3-(phenylmethyl)-benzene | 3340 | 3240 | 2612 | 2443 |
| 19.43 | 3-methoxydiphenylmethane | - | 1807 | - | 3429 |
| 22.50 | 7,9-di-tert-butyl-1-oxaspiro(4,5)deca-6,9-diene-2,8-dione | - | - | 24055 | 11475 |

[a]- = not detected

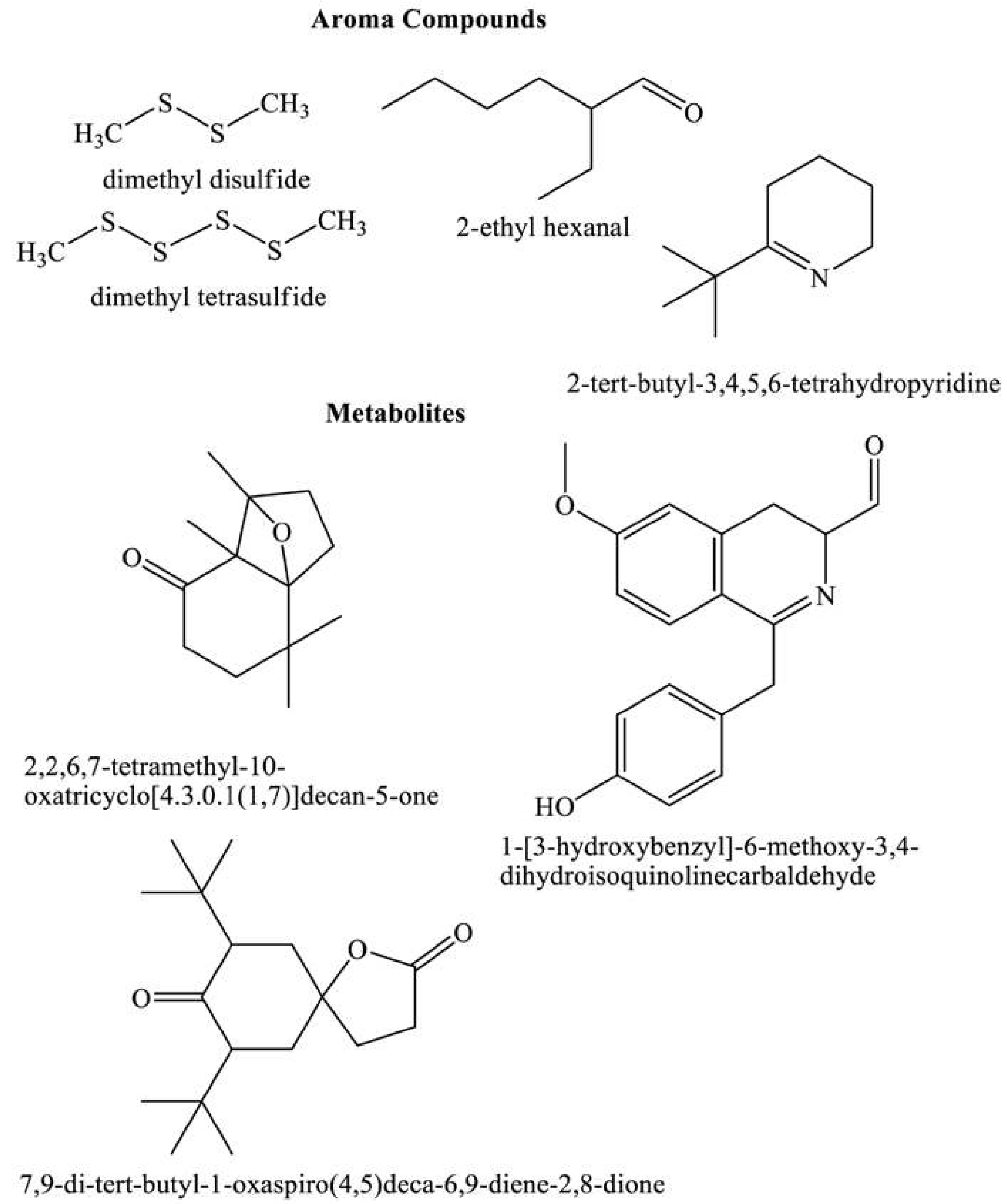


**Figure 1. Structure of compounds tentatively identified by GC-MS.**

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## SOIL AND LAND DEGRADATION (13A)

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# Monitoring soil condition in Tasmania

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The Soil Condition Evaluation and Monitoring program (SCEAM) is a long term monitoring program established to measure the health of key Tasmanian soils, under a variety of land uses, using easily measured, nationally agreed indicators. Unlike some other projects that took a snapshot of soil health the emphasis for SCEAM is to determine trends in condition. The first sites were sampled in 2004/05 and each site will be resampled every five years. A total of 285 sites have been sampled.

At December 2012 approximately two thirds of the sites have undergone one round of resampling. This oral paper will present the methods used in the project together with the results from the original sampling round and, for those sites that have undergone resampling, provide a comparison with the new data.

Six key soil condition parameters were measured at each monitoring site. Soil condition parameters reported relate to the potential soil degradation issues of: structure (bulk density, sodium saturation and aggregate stability), organic matter and biological activity (organic carbon), acidity (pH), nutrient depletion (plant available phosphorus), and erosion susceptibility (aggregate stability).

For each soil condition parameter an acceptable target range has been identified depending on the land use and the type of soil. The defined soil condition targets take account of the natural variability of soils from different soil orders, are land use specific, and are a balance between maximizing agronomic production and minimizing environmental impact.

The project is supported by the Tasmanian NRM regions and will provide information to farmers to make better informed decisions concerning sustainable land management.

# The effects of agricultural practices on acid sulfate soils and nutrient release in groundwater dependent systems of southwestern Australia

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The ecosystems on the coastal plains of south-west Australia (SW WA) (e.g. Swan Coastal plain, Scott River plain) are groundwater dependent. The interactions between surface water and groundwater can greatly influence water quality of these systems. Redox conditions of groundwater can remove nitrate, while evaporation from surface waters can concentrate soluble salts. Acid sulfate soils also widely occur across the coastal plains particularly in wetlands.

Practices associated with agricultural development such drainage, agroforestry and application of fertilisers has altered the natural hydrology, disturbed acid sulfate soils and impacted on water quality. To understand the impact of these practices on acid sulfate soils, nutrient cycling and subsequent effects on water quality for SW WA, a combination of column experiments and monitoring of surface water and groundwater was used. The study sites were located at Torbay, Scott Coastal Plain, Stratham and Peel Harvey area.

The study found acidification of surface water from acid sulfate soils had a limited spatial extent and was confined to disturbed wetlands and the surface runoff from these wetlands. The high acid neutralising capacity of many coastal soils helped to minimise widespread acidification of surface water but salinity impacts from acid sulfate soils disturbance were more dispersed. The depth of the summer watertable was the main determinant of the depth to the sulfidic layer in the acid sulfate soils. Over much of the coastal plain the summer watertable and sulfide layer was below the depth of agricultural drainage. Bluegum plantations can lower watertables thereby exposing acid sulfate soils and causing acidification of groundwater, as well as surface waters.

Acid sulfate soil oxidation formed iron oxy-hydroxides which sorbed phosphorus. The strong reducing conditions of groundwater and soils increases the risk of denitrification but acidification restricted nitrification. Reducing conditions also led to reduction of iron oxides and release of phosphorus into the water column. a

The coastal podzols with humic layers led to strong reducing conditions in groundwater where sulfate concentrated by evaporation was reduced to form sulfides. Our results suggest that acid sulfate soils on many parts of the coastal plain resulted from in-situ groundwater processes rather than marine inundation.

# Evidence of improved land management practices and soil protection in South Australia

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Soil erosion is a major threat to agricultural land in South Australia. Innovative monitoring techniques were used to determine the extent of soil protection across the cropping areas most susceptible to erosion. The relative amount of ground cover and soil disturbance were assessed four times per year at critical stages of the annual plant growth cycle and seasonal cropping activities. The average annual period of protection from soil erosion increased from 272 days in 2003 to 328 days in 2011. Significant improvements occurred despite several seasons of well below average rainfall and other challenging management issues such as snails and mice. Data from telephone surveys give strong evidence of increased adoption of more sustainable land management practices. Since 1999, the proportion of crop sown with no-tillage methods has increased fourfold to 66%, and the associated use of cultivation and burning during preparation for cropping has declined. Survey data provides quantitative evidence to guide policy development and investment in soil management programs. The measured improvements demonstrate the benefits of collaborative partnerships between government agencies, Natural Resources Management Boards and industry stakeholders, and the growing desire by farmers to reduce the risk of erosion and adopt more sustainable farming practices.

# Salt manifestation in the Boro Road area, north-west of Braidwood NSW: Regolith controls on solute storage and transport

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Surface scalds, the presence of salt tolerant species, poor vegetation cover and salt efflorescence in road cuttings and at the land surface, are the physical expression of land salinisation in the Boro Road area, 40km north-west of Braidwood in south east NSW. Local surface water sampling indicates that there is a moderate salt load and in-stream concentrations (ECw) are also elevated. Salinity and soil sodicity have been limiting factors with respect to agricultural production in this area. A program of stream sampling and detailed soil and regolith analysis of the materials formed on the weathered Silurian felsic Long Flat Volcanics was carried out to define micro- (matrix diffusion) to macro- (texture contrast and joint controlled) fluid movement pathways within the landscape. The fluid flow pathways are influenced by: pedogenesis, mineralogy/chemistry of regolith materials, physical properties of the materials, structural impediments to flow, and the configuration of hydraulic controls in the subsurface. A more detailed knowledge of the nature and distribution of soil and regolith materials in this landscape enables clearer understanding of: the capacity for this landscape to store salt, and where and how this takes place; the mechanisms for transport of the salt; controls on the distribution of discharge zones; and, the species and origin of salt present. This provides a fuller explanation of why salt is manifesting in particular parts of the landscape, details the hazard and risk for key stakeholders due to land and water salinisation, and informs land management planning in this area.

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# Tolerance to toxic aluminium in an acidic soil for new perennial legumes

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Aluminium (Al) toxicity is a major limiting growth factor for perennial legume species, such as lucerne (*Medicago savita*), in the high rainfall zone (HRZ >600 mm p.a) in Australia. New perennial legumes, such as Tedera (*Bituminaria bituminosa var. albomarginata* cv. Tedera 27), Caucasian clover (*T. ambiguum* cv. Kuratas) and Birdsfoot trefoil (*Lotus corniculatus* cv. LC07AUYF) offer the potential to be more adapted to the HRZ, however, their tolerance to Al toxicity has not been quantified. A pot experiment was conducted using a soil with low pH and high Al, that was adjusted with lime to 4 pH levels ranging from 4.2 to 5.7 in $CaCl_2$. Lucerne (cv. Sardi 10), subterranean clover (*T. subterraneum* cv. Leura)) and Greater lotus (*L. pedunculatus* cv. Maku) were included in the experiment to compare the relative Al tolerance of the 3 new cultivars. All the treatments were inoculated with an appropriate *rhizobia* strain. Half of the pots received extra mineral nitrogen as $NH_4NO_3$ solution. Pots were harvested at 63 days after seed germinated. Results showed that shoot weight increased with increased lime rates for all species, but root weight responded to liming differently depending on species. LC07AUYF lotus was unresponsive to liming in root weight while Maku lotus decreased root weight at high lime rate, indicating that lotus species are more tolerant to acidity than lucerne and subclover. Applying lime in general improved the nodulation for all species, but at the highest lime rate, the nodulation score decreased for all species except for Leura subclover.

# HGL – a framework for salinity understanding and management

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Failure to address salinity is due in part to lack of understanding of how the landscape functions and the use of inappropriate management techniques. Simplistic views of salinity management and restoration have led to negative hydrological impacts or poor outcomes. The NSW Department of Primary Industry the NSW Office of Environment, Geosciences Australia and University of Canberra have developed a Hydrogeological Landscape (HGL) system for understanding how salinity manifests in the landscape, how differences in salinity are expressed across the landscape and how salinity may best be managed at a site and in a landscape context. The process of HGL determination relies on the integration of a number of factors: geology, soils, slope, regolith depth, and climate; an understanding of the differences in salinity development (plumbing); and the impacts (land salinity/ salt load/ stream EC) in landscapes. A feature of the methodology is that it provides information in a hierarchical structure that allows interaction at different scales. An outcomes of the methodology is that a HGL product can be used at a strategic or at a tactical level. HGLs have been successfully implemented at a variety of scales and for a variety of clients. HGL products have been developed: for use in urban planning and urban water management in NSW including Bathurst, Dubbo and Western Sydney; to target natural resource investment for a number of NSW catchments including the Murrumbidgee; and, for waterway management in the Braidwood region.

# Changes in soil chemistry under eucalypts irrigated with highly saline and sodic wastewater

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Land disposal of highly saline industrial wastewaters requires careful consideration of inputs and site responses to identify and mitigate avoid adverse effects to the environment. An experiment was set up to investigate the sustainability of irrigating a eucalypt plantation with highly saline and sodic effluent from a gelatine manufacturing plant with a high organic loading and nitrogen content. The 3 year experiment compared the soil responses of a duplex brown dermosol under two Eucalypt species (Eucalyptus teriticornis and E. moluccana) planted at two densities and a Rhodes Grass pasture established in early 2001. Irrigation applied a maximum of 40 tonnes of salt per annum. Leaching could not be achieved with a leaching fraction and responded only to large rainfall events ($P<0.05$). High water use rates of higher density stands resulted in higher profile salt concentrations. Cation ratios down the profile changed significantly over time ($p<0.001$). Sodium saturated the deeper part of the profile ($P<0.001$), magnesium and calcium increased in the upper profile and reduced at depth, and potassium concentrations remained stationary. Mean pH values reduced from 5.3 in 2001 to 3.8 in 2004 with a pH value of 1.11 recorded under E. moluccana at low density. This is attributed to the conversion of $NH_4^+$ to $NO_3^-$ and subsequent leaching of $NO_3^-$. Under more intensive irrigation schedules soil structural degradation may decrease the soils capacity to accept irrigation water for leaching, exacerbating the salt loading problem.

# Controls on salt storage and urban/peri-urban land salinisation in the Greater Launceston Area: Preliminary hydrogeological landscape characterisation.

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Localised expression of land salinisation, with associated salt load and elevated electrical conductivity levels in nearby streams, is observed in an urban/peri-urban setting in the greater Launceston area. Evidence for salt presence in parts of the landscape includes: etching and rotting of brickwork; ‘rising damp’ in masonry walls; fracturing and flaking of pavements and roads; salt efflorescence as a ‘tide mark’ on some brick and concrete walls; blistering of paintwork; formation of scalds in grassy areas; presence of salt tolerant species in water courses; and, acid and saline chemistries in some natural waters. Informed management of salinised land and water through: skilled urban planning; targeted infrastructure maintenance; use of appropriate construction materials and methods; minimisation of excavation and land-use practices that impact land and water salinity; provision of information to stakeholders; and, other tailored solutions by local councils and water authorities, requires an understanding of how and why salt is manifesting in some parts of the landscape.  Hydrogeological landscape (HGL) characterisation explains: the configuration of soil, regolith and fractured rock materials in the landscape; the capacity for salt storage and release in different media; how water moves over and through these materials; the mechanisms for salt mobilisation in the landscape; and, the likely provenance of the salt. Preliminary findings indicate that land salinisation typically impacts localised areas on: deeply weathered Jurassic dolerite; moderately to deeply weathered bedded paleo-estuarine sediments of the Paleogene Tamar/Esk River system; some Quaternary terrace deposits along the Tamar and Esk Rivers; and some Holocene estuarine sediments.

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# The regulatory challenge for soil science

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### Abstract

There has been an increased demand for soil science in a regulatory context in Queensland over the last 20 years. Soil science as a discipline has not always been effective at engaging policy-makers in finding effective ways of incorporating what are often semi-qualitative concepts into regulatory frameworks. There is now widespread use of soil science in many regulatory instruments but the science is not always appropriately applied in correct contexts. There is a significant challenge for soil scientists to better engage policy-makers and develop new and innovative ways to ensure that policy and regulation accurately reflect the realities of soils and landscapes and the nature of the science that describes them.

**Key Words**
soil science, policy, regulation, salinity

### Introduction

Over the last 20 years there has been a substantial increase in the number of regulatory instruments (policies, Acts etc.) in Queensland that include or rely upon aspects of soil science. Earlier instruments included the State Planning Policy (SPP 1/92) – Development and the Conservation of Agricultural Land and the Vegetation Management Act (1999). In 2002, an SPP related to acid sulphate soils was introduced (SPP2/02) and in 2005, the Wild Rivers Act. The most recent Act to rely upon soil science and the one that draws the most upon the discipline is the Strategic Cropping Land Act (2011). Elements of soil science are also relied upon in the regulation of civil construction, mining, gas and industrial activities under the Environment Protection Act (1994) and Waste Reduction and Recycling Act (2011). Other notable references to soil science in a regulatory context include the NEPM site contamination guideline (EPHC 1999) and the ANZECC Water Quality Guidelines (2000).

Given the widespread use of soil science in regulatory frameworks in Queensland, it could easily be assumed that there would be strong ties between policy and soil scientists and the way in which soil science underpins natural resources legislation would be well explored. Unfortunately, those creating and administering regulations do not always have soil science training and the number of soil scientists available within government has been diminishing over the last decade. Regulation of the rapidly expanding coal seam gas (CSG) industry and the recent SCL legislation are good examples of the difficulties associated with placing both the quantitative and qualitative aspects of soil science in regulatory instruments.

#### *The regulatory challenge*

Those involved in administering legislation are all too aware of the requirement of regulations to be clear-cut in terms of decision-making. Vagueness in decision-making all too often leads to legal evaluation which is costly for all parties. 'Crisp' decision points are easily defined in some regulatory areas e.g. cadastral boundaries, but less so in others e.g. boundaries of vegetation communities or soil map units. A parallel may be drawn with the 'tyranny of the map'. Traditional soil maps imply a crisp boundary between map units but this is rarely the case in reality. Regulations generally infer crisp decision points but in the case of natural resource systems this is often not the case. There are many drivers for regulations to be crisp, in particular the perception by policy-makers that they are less costly to administer, less costly for proponents and can be administered by staff without expert knowledge in natural resources. Unfortunately, such approaches can often lead to inappropriate regulatory frameworks and perverse outcomes. There is a challenge for the soil science community to better engage policy-makers in Australia and develop new, robust approaches that support regulations that are both scientifically rigorous and readily/economically applied.

Very few attributes of soils and landscapes are readily described, measured or evaluated with a high degree of precision and certainty. Morphological attributes (colour, structure etc) are semi-quantitative at best and subject to operator bias. Analysis of samples has many areas of potential error, starting with site collection, and sampling handling through to laboratory technique. Virtually all analytes have multiple possible methods with the result often varying substantially between methods. Explanation of the vagaries of measuring soil attributes to policy-makers can all too easily lead to either complete abandonment of soil science in policy or reinforcement of stereotypes that soil scientists can't make up their mind, often with the result of policy-makers creating their own interpretation of the science. Two examples can be used to illustrate such difficulties associated with the use of soil science in natural resource legislation and the challenge for scientists and policy-makers alike.

*1. Strategic cropping land*

The Strategic Cropping Land Act (2011) revolves around a core principle of the spatial delineation of soil/land that passes thresholds for eight criteria. The criteria are assessed at a point but the legally defined entity is a spatial unit. The criteria and thresholds were developed by a group of experienced government and private sector soil scientists including this author. The process exposed a number of possible short-comings in the Australian Soil and Land Survey Field Handbook (NCST, 2009), the need for the soil survey community to be more explicit about some of its methods and the need for further method development.

*The tyranny of the map in SCL*

By definition, the SCL Act requires a proponent to spatially delineate the areas of land they deem to be/not be SCL. This immediately invokes questions regarding the fundamentals of mapping (soil survey), such as map scale and site density. While there are published tables for determining aspects of these (McKenzie *et al*. 2008) experienced soil surveyors know that these are only a guide. Actual site density is determined by factors such as access, landform, infrastructure/land tenure, landscape/soil complexity and skill/knowledge of the practitioner. Such factors are, however, as much an art as a science and not easily captured in an explicit sense. Thus there is a necessary degree of vagueness required in the definitions of required scale and site density for SCL assessments. The same issues arise in broader scale assessments for soil management plans in CSG developments. Clear rules regarding site density make it easy for costing of surveys and assessment of compliance. If strict rules are used, however, there is invariably a risk of a survey being over-sampled or under-sampled. This is further compounded by the cost-pressures associated with private sector activities.

A second aspect of mapping causes difficulty in SCL. A core component of the SCL Act is the 'trigger map'. This was compiled from Agricultural Land Class (ALC) data associated with some of the existing land resource mapping in relevant parts of Queensland. The mapping varied in scale from 1:50 000 soil/land suitability surveys to 1:500 000 land systems surveys. The ALC has previously been allocated to map units for land planning purposes related to SPP 1/92. Class A land is land that is arable. In detailed surveys, the ALC is derived semi-quantitatively from land suitability data. For land system surveys however, they have been allocated on the basis of expert opinion and analysis of existing land use. The validity of allocating an ALC to a land system is of course questionable, particularly for highly heterogeneous map units which may comprise a mix of arable and non-arable lands. A specific ALC (B2) can be used to reflect mixed units but it has not always been employed. Unfortunately, most users of the data are not cognisant of its limitations or matters such as map unit purity and hence are inclined to have an inappropriate level of belief in the certainty of the maps. The subsequent re-interpretation of ALC data for other purposes can of course lead to a compounding error. There lies a significant challenge for the managers of soil survey data to develop better methods of communicating the data to ensure users are more fully aware of map unit purity and the limitations of interpreted attributes. A related challenge exists with respect to raster soil mapping – it is not currently recognised in any regulatory frameworks.

*SCL criteria*

The criteria used to assess land in the SCL Act are slope, rockiness, microrelief, soil depth, drainage, pH, salinity and soil water storage. Some criteria are independent, while some, such as moisture availability, are dependent on other soil attributes and/or criteria. Criteria are ordered from easy to hard and within each there was an attempt to introduce a hierarchy of measurement methods. For instances in which an attribute/measurement was clearly well above or below a threshold, a less onerous/robust method could be employed. As values approach a threshold however, more accurate methods are required. Few people argue regarding the relevance of each criteria and their importance in determining agricultural land use. Much debate has occurred, however, regarding the thresholds. Surprisingly little debate has centred on measurement methods despite this being the aspect most likely to cost proponents money and to potentially be evaluated in a legal context.

Methods for measurement and description of soil and landscape attributes have been clearly documented for more than 20 years via the various editions of the Australian Soil and Land Survey Field Handbook (ASLSFH) (McDonald *et al*. 1984; McDonald *et al*. 1990; NCST 2009). Little has changed since the first edition and thus it could be assumed that the descriptors meet all needs. This may be the case for general soil survey but when attempting to place these in a regulatory environment it becomes clear that methods used in 'normal' soil science may not be sufficiently robust or rigorous. Rockiness, gilgai and soil water are three examples.

The size and frequency of coarse fragments, both on the soil surface and in any given horizon, are estimated in a semi-quantitative manner in soil survey. A ruler or tape may be used to check size, and frequency pictograms are given in the ASLSFH. While these methods have always been sufficient for the general uses of soils data they are unlikely to stand up to significant scrutiny in a legal setting. Operator error in visual estimates can easily be ±10% but the criteria threshold in SCL is crisp (20%). The crisp threshold implies a level of certainty that does not exist i.e. 19% is a pass and 21% is a fail (± 1%). It is not possible to resolve rockiness frequency to 1% in the field, although developments in image capture and analysis in recent years now enables more accurate determinations using tools as simple as a digital camera. Other options include the use of a quadrat – a tool frequently used in field archaeology but rarely in soil science.

Gilgai are a somewhat esoteric feature to the average person and even some soil scientists – not too many are present in southern Australia! While the ASLSFH gives terms for the types of gilgai and descriptors for measuring horizontal and vertical interval, it provides far from sufficient detail for the measurement of gilgai in a regulatory context. This is not surprising, as the SCL Act is probably the first in the world in which gilgai have entered the regulatory domain. In the context of SCL and agriculture in general, the critical attributes are the depth of gilgai and the proportion of the land that they occupy. While capacity exists to describe mound, shelf and depression it is generally the depression that causes the most difficulty to agricultural practices thus for the purposes of SCL, only the areal extent of the gilgai depression needs to be determined. Doing so is not covered in the ASLSFH so a method had to be derived. Various options were considered including measuring the gilgai encountered in a fixed distance at fixed directions (compass points) from a site. Such a method is flawed, however, as the number of gilgai encountered is a function of their size. A distance/size rule could be created but the effort in doing so was not considered worthwhile. In the end, the simplest method was chosen – describe the nearest 10 gilgai to a site and take the average. Statisticians could easily mount arguments regarding the size of the sample population etc., but as is apparent with all of the SCL criteria, a balance is required between achieving an outcome that is both practical/economical and has a sufficient level of scientific rigour.

The measurement and estimation of soil water is an extensive topic with many laboratory and field methods available, as well as pedotransfer functions. No method is perfect, however. The most significant change over the last 20 years has been to describe plant available water capacity in the context of crop type, rather than just a single number for a soil. Pedotransfer functions, while often simpler and less expensive to use, have a degree of imprecision which is not always clearly defined. A simple pedotransfer approach, in the form of a lookup table based on texture, was employed in SCL to enable users to determine if a soil was clearly going to pass or fail the criteria. For soils close to the threshold, more detailed assessment is required. The most commonly applied field method in Queensland (and probably in Australia) is that of Dalgliesh and Foale (1998). This method, however, requires a full growing season to determine crop lower limit. A lengthy measurement duration is not practicable for SCL thus a compromise method was derived to achieve results that were robust and achievable in a shorter time period.

*2. How much salt is too much?*

A different case study involves soil salinity. The rapid development of the CSG industry in Qld has placed increased focus on the question of how much salt is too much? In agriculture, the addition of salt to soil is generally, but unfortunately not always, a self-limiting process. Most landholders, with the exception of those disposing of intensive livestock effluent, operate in an environment of water scarcity. CSG companies on the other hand operate in an environment of water surplus – and one in which the water is saline and declared a waste. Disposal of water via irrigation after amendment/treatment is one option currently used.

The application of saline waters/waste to soils is regulated under a variety of legislation in Queensland, including the Environment Protection Act (1994) and Waste Reduction and Recycling Act (2011). The

ANZECC water quality guidelines (Chapter 4) are a much referred to document in regulatory frameworks but something that is frequently misunderstood and misused. Invariably at some point in the regulatory frameworks there is a need to define some point at which 'harm' is caused to the soil (or other feature such as water). The NEPM guidelines provide thresholds for many contaminants – for instance, to trigger investigations the ANZECC guidelines give both short-term and long-term trigger values for heavy metals and metalloids in irrigation water. The ANZECC guidelines also give trigger values for prevention of foliar injury in plants by Na and Cl. Agronomic thresholds for soil salinity have been published for decades – the most frequently quoted are tabulated in SalCon (1997) and to a lesser extent in the ANZECC guidelines. There are, however, no specific thresholds for maximum soil salinity as either electrical conductivity (EC) or chloride. This is not surprising as the various thresholds published historically have all come from an agricultural focus rather than the perspective of defining harm to the landscape. The regulatory paradigm most often employed when dealing with application of saline waters to land is the concept of 'keeping the root zone salinity below a threshold value' which stems from a section in the ANZECC guidelines. While leaching fractions and leaching requirements are well founded in science, they are only one half of salinity risk –the other half being the landscape salinity risk – related to the unsaturated zone etc.

The use of agronomic thresholds is generally a valid approach in instances where the land has been/will be used for cropping. A simple approach would be to set salinity limits within the root zone relevant to the least tolerant crop historically and commonly grown at a site. At the cessation of irrigation with any saline waters the site must still be capable of growing its historical range of crops. Much of the CSG development is, however, in pasture lands with no history of cropping and tree crops (leucaena forage, eucalypt plantations) are chosen rather than grain/fibre. Pastures are a mix of natives and introduced species, and salinity tolerance data is frequently not available for such species. In the absence of an agronomic threshold to work to, it is necessary to refer back to a baseline. While this is simple in concept it is more complex in reality. Prior to disturbance, a soil is generally in equilibrium with rainfall and other salt inputs (e.g. mineral weathering), and losses via washoff, lateral flow, deep drainage etc. This equilibrium is reflected both vertically in the profile and horizontally in the landscape. This is the *natural baseline*. Upon disturbance (clearing of native vegetation), the general outcome is an increase in deep drainage and downwards movement of salts. The site is no longer in a steady state but rates of change can be very low. A *transient baseline* may be obtained at any point. If salt is added to the landscape via irrigation water, which of these applies? If you use the natural baseline, then the salinity of the profile may be increased. If, however, the transient baseline is used, profile salinity may have to be kept low which can only be achieved through the use of very low salinity irrigation water or leaching. Both of these can lead to perverse outcomes. There are a number of other questions to be considered. How is the salinity of a profile represented as a baseline: as a profile weighted mean, values at fixed depths, or as a maximum value within the root zone? What is the root zone? How much variation around a baseline value is acceptable – 10%, 20%? Clearly the establishment of soil salinity baseline values in a regulatory context is not as straightforward as some would perceive.

## Conclusions

There has been a growing recognition of the importance of soil science in environmental regulation in the last 20 years in Queensland but the discipline has not always been up to the challenge. Current demands on soil science to support the development and implementation of the SCL Act and environmental regulation for the CSG industry highlight the need for soil science to better engage with policy makers and derive innovative, robust ways in which soil science can be used to create practical regulatory frameworks.

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# Thursday 6 December

## INVITED SPEAKERS

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# What's happening in soil fertility (and contamination) R, D and E in Australia and New Zealand

Ants Roberts

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In our world where food shortages loom and people go hungry and projections of global population growth suggest future food security crises, the continuing sustainable food producing capacity of our soils, while minimising off site impacts, is imperative. The unprepossessing title disguises the treasure trove of information being generated in this field by scientists from not only Australia and New Zealand but also by researchers working around the world from Vietnam to Africa and Iran to China and places in between. The paper takes the 'temperature' of current research, development and extension in both countries as displayed by some 93 poster and oral papers given at this Conference.

As expected there is a diversity of topics covered ranging from new soil test developments, soil nutrient supply and cycling including in plant and animal systems, modifying soils to improve their performance, pine forest nutrition, N fertiliser efficiency to N and P losses and contaminant additions to the environment as well as assessment of potential mitigations of these impacts.The paper will attempt to draw together the main themes of the R, D and E being undertaken, briefly discuss some of the insights provided by the information and comment on the perceived usefulness of this for land users.

# ’Tis an ill wind that blows nobody any good

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Anthropogenic climate change is regarded as an ill wind, but soil scientists should all rejoice because after many years of neglect the benefits of soil carbon (C) have suddenly leapt into public prominence. Under the theme ‘soil carbon and climate change’, this review analyses submitted papers and the wider literature from the perspective of (1) what do we know about the effect of climate change on soil C, and (2) what do we know about the efficacy of building up soil C in mitigating climate change? With respect to plant growth and enhanced returns of organic residues to soil, the positive effects of elevated carbon dioxide concentrations ($CO_2$) and higher temperatures on growth tend to be counterbalanced by the negative effects of increased respiration and possibly decreased water availability. Generally, the positive effect of reduced cultivation/no-till has been overstated because of an inadequate depth of sampling and expression of results in terms of equivalent depth rather than equivalent soil mass. Although the most effective change in agricultural land use to increase soil C is from cropland to permanent pasture, emissions of methane from grazing ruminants may negate the positive effect of C storage.

Although models such as RothC can simulate changes in soil C in closely controlled, small plot experiments, for extensive areas under Australian conditions the precision of the simulated change in C decreases markedly. Similarly, because of spatial variability in soil C and analytical uncertainties, confidence in measured soil C contents is low unless a large number of samples is collected and analysed, which becomes very costly. Nevertheless, simulation modelling of soil C is at the core of FullCAM in the Australian National Carbon Accounting System, so it is essential that the model be continually validated and improved for Australia to meet its Tier 3 accounting commitment under the Kyoto Protocol.

The potential greenhouse gas abatement from soil C storage under optimum management across all intensively and extensively managed land is about 21 percent of current national emissions. If one assumes average management across this large area, the figure falls to 12 percent. However, the Department of Climate Change and Energy Efficiency’s own projection for abatement from soil C sequestration in the year 2020 is 0.5-1.5 Mt $CO_2$-equivalent, amounting to only 0.1-0.3 percent of current emissions. Therefore, the prime motivation for increasing soil C should be to improve the soil condition and enhance crop productivity in a world facing potentially serious food shortages.

## Introduction

Why the strange title one may ask? Well, a former prime minister once labelled climate change ‘the greatest moral challenge of our time’ because of its far-reaching adverse consequences. However, soil scientists should all rejoice because in the past few years, soil carbon (C) has leapt into public prominence. Back in 1938, Albrecht observed that ‘soil organic matter is one of our most important national resources; its unwise exploitation has been devastating.’ Now, after many subsequent years of soil scientists extolling the virtues of building up soil organic matter (SOM), the topic has suddenly come of age in the form of the Australian government’s Carbon Farming Initiative (CFI). So, the ill wind of climate change has blown a good change in public policy with respect to soil C. More of that later.

‘Soil carbon and climate change’ has been a popular theme at this conference, judging by the number of papers and abstracts submitted. I was asked to give an overview of a selection of these papers, 20 in all, which covered a range of issues, only some of which bore directly on the theme title. I will not discuss these in detail – the individual authors can do that better than I. Rather I want to discuss the findings in the context of this broad theme under two headings:

1. What do we know about the effect of climate change on soil C, and
2. What do we know about the efficacy of building up soil C in mitigating climate change?

## The effects of climate change on soil carbon

Climate change can be a natural process occurring over millennia. Climate varies naturally around the earth. So if under this heading we consider the components of daily climate or weather, such as temperature, rainfall, wind and the related effects of drought, fire, floods, freezing and thawing, water and wind erosion, we can point to a wealth of knowledge about these effects. We can go further and look at differences in land use and management practices, many of which depend on the prevailing climate and its associated weather, and have been extensively studied. Finally, in the context of anthropogenic climate change, we can look at the concomitant effects of rising carbon dioxide ($CO_2$) concentrations and rising mean temperatures on plant growth and the balance between the gain and loss of soil C.

**Temperature, rainfall, fire and related factors**

Broadly, the effect of these variables can be interpreted in terms of the balance between C inputs to and losses from soil. Where these variable factors favour greater plant growth and consequent return of more residues to the soil relative to the loss of C through decomposition, erosion and leaching, the soil organic carbon (SOC) will increase. In this conference, Heath et al. (2012) (paper 17) reported on the effect of wildfire in a Eucalypt forest on soil C and water repellency and found that found that both decreased with the intensity of the burn. Further, Liu et al. (2012) (paper 434), working in a wet sclerophyll forest in SE Queensland, found that biennial burning significantly decreased soil microbial biomass relative to the control, but quadrennial burning had no effect on either microbial biomass or denitrifier activity.

The effect of soil wetness on C storage was demonstrated by Read et al. (2012) (paper 323) who measured the change in SOC in salt-scalded Vertosols in the central west of NSW following water ponding. The C density, measured on an equivalent mass basis, increased from 19 to 25 t C/ha in five years, an accretion rate of just over 1 t/ha/yr. This is a significant rate of C accumulation that has the potential to be permanent if the soils remain waterlogged. Jeong et al. (2012) (paper 91) measured the interaction between fertilizer and the annual change in temperature on soil respiration under a pine forest and found that although respiration was very sensitive to temperature, there was no interaction with fertilizer input.

**Conclusion**

These results are consistent with existing knowledge. For example, with respect to the effect of rainfall and soil wetness, of the approximately 75 billion tonnes of C stored in European (EU) soils, some 20 percent is in peatlands in the northern countries. Drainage of organic soils and land use change have resulted in the largest emissions of CO2 from EU soils, amounting to as much as 20-40 t $CO_2$/ha/yr (Alterra 2008). Hence, one of the main ways of avoiding emissions from agriculture is to conserve peatlands and other wetland soils.

**Land use and management effects on soil carbon**

Table 1 summarizes the effects of land use change and different management practices on SOC.

**Table 1: Summary of conclusions from recent experiments/surveys on the effect of land use and/or management on SOC or rate of SOC change**

| Land use/land use change | Management/ man-agement change | Effect on rate of SOC change | Comments | Authors |
|---|---|---|---|---|
| Grassland to Leucaena-grass pastures in Central Qld | Leucaena-grass plots of 9, 22, 34 and 40 years | Increase of 0.3 t C/ha-0.3 m/yr under the Leucaena rows | Rate of SOC storage declined after 22 years due to P and S depletion | Conrad et al. (2012) (paper 251) |
| Vegetables, turf, dairy/grazing and orchards in NSW | One application of compost at 40 t DM/ha | Significant increase in SOC (0-0.1 m) at 54 out of 62 sites. | SOC decreased at sites under intensive vegetable production | Brunton & Orgill (2012) (paper 275) |
| Pasture, forest and cropping in New England | Estimate of the theoretical stable C of Ferrosols; soils under pasture & cropping undersaturated in C; those under forest in balance or oversaturated | | | Khandakar et al. (2012) (paper 284) |
| Pine forests and pastures (>22 yr) in central NSW | Based on soil respiration, lower below ground C allocation and lower turnover of organic C in soil under pines than pasture | | | Matta et al. (2012) (paper 344) |
| Pasture and cropland on Ferrosols in Tasmania | Pasture soils had 33% and 50% more C than cropped soils in 1997 and 2010, respectively. SOC (0-0.15 m) was decreasing faster in cropped soils than in pasture soils, but changes not significant between 1997 and 2010 | | | Parry-Jones et al. (2012) (paper 397) |
| Eucalyptus seedlings in a glasshouse | Biochar (made from macadamia kernels) at doses of 0, 2, 5, 10, 20, 50, 80 and 100 t/ha with two rates of fertilizer | Seedling growth up to 268 days responded to fertilizer, but no effect of biochar rate | | Wrobel-Tobiszewska et al. (2012) (paper 464) |
| Perennial pastures in SE NSW | Because of variability in SOC concentration (0-0.3 m), there was no significant difference in C stocks calculated either by the fixed depth or equivalent soil mass methods, regardless of sample core diameter | | | Orgill et al. (2012) (paper 412) |

**Conclusion**

The results in Table 1 are consistent with reports in the literature regarding land use change and addition of organic residues. For example, in a survey of EU soils, Alterra (2008) reported that grassland soils consistently accumulated C (1-45 Mt per year), forest soils generally accumulated C and cropland acted as a highly variable source of $CO_2$-C (up to 39 Mt per year).

Powlson et al. (2012) reported on the extent to which SOC stocks in agricultural soils in the UK could be increased through management changes, focusing on the effect of switching from conventional cultivation (ploughing to at least 0.2 m) to reduced tillage (cultivation <0.15 m depth) and no-till (direct drilling), and the addition of organic materials such as farm manures, digested biosolids, cereal straw, green manure and paper crumble. Although the average annual increase in SOC derived from reduced tillage was 0.31±0.18 t C/ha-0.3 m, they noted that farmers in the UK and NW Europe who practised reduced tillage often carried out mouldboard ploughing every 3 to 4 years (‘rotational ploughing’) to relieve soil compaction and for weed control. This would largely eliminate any gains in SOC made during the reduced tillage phase. Significant gains in SOC could be made from the addition of organic materials, depending on their source and rate of application, but in many cases these were not real gains because their use on agricultural soils involved a transfer of material from another site, which was therefore deprived of the C input. The exceptions were waste materials such as paper crumble and green compost that would otherwise go to landfill and become a source of methane ($CH_4$) emission. Although paper crumble was effective in increasing SOC, it had such a high C/N ratio that extra N fertilizer would be needed when the land was cropped, which had the undesirable consequences of extra energy consumption (and greenhouse gas (GHG) emissions) and of increased nitrous oxide ($N_2O$) release – the net effect on GHG emissions was negative. For green compost, the net effect was small positive (≈0.1 t $CO_2$-equivalent ($CO_2$-e)/ha/yr).

In a review of Australian results, Sanderman et al. (2010) concluded that improved management of cropland, whether through enhanced rotation, adoption of no-till or stubble retention, had resulted in a relative gain of 0.2 – 0.3 t C ha/yr compared to conventional soil management. However, when time series data were available, even the improved management often showed significant absolute decreases in SOC stocks, which, in many cases, was likely a direct result of these soils still responding to the initial cultivation after clearing. More specifically, Chan et al. (2011), reporting on long term trials (13-25 yrs) in SE Australia, found that under continuous cropping, even with conservation practices such as no-till, stubble retention and crop rotation, a high initial SOC was at best maintained but more often decreased when tillage and stubble burning occurred. The only system to show consistent annual gains of 0.5-0.7 t C/ha-0.3 m was permanent pasture under improved soil nutrient and grazing management.

Chan et al. (2011) found no difference in SOC (to 0.3 m) between perennial and annual pastures, whereas Sanderman et al. (2010) in their report suggested that perennial pastures might accumulate more C at depth than annuals. Additionally, sampling depth is important in estimating changes in SOC when comparing conventional cultivation with no-till. For example, Baker et al. (2007) found that in North American studies where conservation tillage was found to increase SOC in the 0-0.3 m layer, in almost all cases where the soil was sampled more deeply there was no consistent increase in the C stock, a result confirmed by Luo et al. (2010) from a meta-analysis of changes in SOC to at least 0.4 m depth following conversion from conventional tillage to no-till in 69 paired experiments worldwide. Partly this contradictory result is due to the difference obtained when the C stock is calculated on the basis of a fixed depth as opposed to an equivalent mass (Powlson and Jenkinson 1981), because no-till soils usually have a higher bulk density than conventionally cultivated soil (Ellen and Bethany 1995), and partly because returned C in no-till soil is more concentrated in the surface 0.1 m, whereas in conventionally cultivated soil it is more evenly distributed in the top 0.4 m.

### Modelling soil carbon

**Table 2: Summary of the results of modelling to simulate SOC status and C dynamics under different land uses at a range of scales**

| Objective of modelling | Methodology | Model performance and uncertainties | Conclusions | Authors |
|---|---|---|---|---|
| Estimation of pre-clearing SOC in soils of eastern Australia | Multiple regression using temperature, parent material, rainfall, aspect and soil depth | 61% of variation in ‘topsoil’ SOC accounted for | Could be used to improve SOC estimates in C accounting models; also insights into environmental factors controlling SOC in Australian soils | Gray et al. (2012) (paper 167) |
| Estimate SOC change in a part of the Lachlan Catchment, Central West NSW | SOC measured for 0-0.1 m and pedotransfer function used to estimate SOC to 0.3 m | Precision for detecting changes in SOC over 5 years was 1.5-2.0 t/ha-0.3 m, i.e. 0.05% in a soil of 1% C | Pedotransfer function inadequate to measure changes over 5 yr and soil sampling too expensive | Murphy et al. (2012) (paper 206) |
| Simulation of SOC across in a 1445 km$^2$ catchment in northern NSW | Roth-C with spatial input data and standard parameters run for 30 yr | CV of SOC predictions for different soils and land uses was large | Uncertainty underlines the perils of using models with naive input and parameter assumptions | Karunaratne et al. (2012) (paper 306) |
| Estimate the potential C sequestration in an annual wheat system | SOC simulated by APSIM at 557 sites in the Australian wheat belt for 122 yr | Spatial variation in SOC estimated by regression on P, T, radiation, PAWC and $C_{inert}$; uncertainty in predictions of 20-60% | Under optimum management, sequestration potential was -10 to 20 t C/ha-0.3 m; need for adequate N input was emphasized | Wang et al. (2012) (paper 326) |

| | | | | |
|---|---|---|---|---|
| Test two hypotheses for the effect of tillage under wheat-sorghum on SOC in SE Qld | SOC simulated by APSIM using different model parameters affecting decomposition rate | SOC change was well simulated provided that parameters for HUM or inert C fraction were adjusted for tillage intensity | Results did not enable the relative importance of the two approaches to be decided | Luo et al. (2012) (paper 328) |

**Conclusions**

The results for simulation modelling in Table 2 illustrate the great difficulty, at the landscape scale or larger, in modelling with acceptable certainty either the current stocks of SOC or the changes in SOC with time. As the paper by Luo et al. (2012) shows, agreement between measured and simulated C contents can be very good for closely controlled, small plot experiments with adequate replication. However, the other papers in this group show that the precision of the simulated output decreases markedly as the area to be modelled increases. The main reasons for this outcome are (a) a lack of reliable data on initial SOC values, especially when legacy data are used (through inadequate sampling and/or analytical errors), (b) lack of data on the distribution of land use, weather variables and soil properties at a high spatial resolution and how these change over time, and (c) naive assumptions about soil and crop management, e.g. that optimum management with respect to cultivation, fertilizer use and pest and disease control prevails through time.

Discussing the reporting of SOC changes under Article 3.4 of the Kyoto Protocol, Smith (2004) noted that such reporting must take account of 'uncertainties, transparency in reporting, verifiability'. Following the results of Garten and Wullschleger 1999, Smith (2004) noted that for a given land use the minimum difference detectable with a reasonable sample number (16) and good statistical power (90% confidence) was 5 t C/ha±1 (10±15% of background C level), and that most agricultural practices would not cause soil C accrual rates as high as this during a 5 year period. The problem of uncertainties in measurement or modelling was also emphasized by Goidts et al. (2009), who recommended when assessing SOC stock at the landscape scale, one should focus on the precision and accuracy of SOC analyses in the laboratory, the reduction of SOC spatial variability (using bulk samples, accurate re-sampling, high sampling density or stratified sampling), and the use of equivalent soil masses for C stock comparison.

Other points revealed by measurements on long-term experiments and by modelling are that (a) the SOC takes many years to attain a new equilibrium level following a change in land use or management (Sanderman et al. 2010, Powlson et al. 2012), and (b) as a corollary, the rate change in SOC (increase or decrease) diminishes with time. Sanderman et al. (2010) reported that the largest gains were generally found within the first 5-10 years with the rate of change diminishing to near 0 after 40 years, while Powlson et al. (2012) found for European conditions the rate of change was most rapid in the first 20 years, but could continue for a further 100 years.

Sanderman et al. (2010) point out that measuring and statistically verifying small changes in soil C stocks against a large and heterogeneous background are difficult at best, even at the farm scale, let alone at the regional or national level. Possibly for this reason, simulation modelling is an essential component of the soil C module of FullCAM in the Australian government's National Carbon Accounting System (NCAS), a Tier 3 system (IPCC 1997) that assumes, inter alia., soil C fluxes can be 'monitored and verified' at a national level (Smith 2004). However, Falloon and Smith (2003) found that the sampling and analysis of long term experiments needed to be improved if the data were to be used to test model validity and reduce the uncertainty of model projections. Further, given that the accuracy of remote sensing data of land clearing in NCAS has been questioned (Macintosh 2007), the national accounting of soil C fluxes may not meet the verifiability standards of a Tier 3 system.

**Changes in the carbon flux between the soil and atmosphere**

Higher $CO_2$ concentrations in the atmosphere promote plant photosynthesis. The increase in average surface temperatures over the past 60 years has accelerated plant physiological processes, including respiration, but the overall effect has been to promote plant growth, which in turn can potentially increase the return of C residues to the soil. Several papers under review focused on the effects of an elevated $CO_2$ concentration on crop growth, the possible limiting factors to growth, and the likely effect on soil C accumulation. The results are summarized in Table 3.

**Table 3: Summary of the effects of elevated $CO_2$ ($eCO_2$) concentrations on crop growth, return of residues to the soil and subsequent soil N availability**

| Objective of experiment/analysis | Methodology | Treatment effects | Comments and conclusions | Authors |
|---|---|---|---|---|
| Measure the effect of $eCO_2$ on net soil N mineralization | Wheat and field peas grown at ambient and 550 ppm $CO_2$ in AGFACE | Net mineralization significantly greater with $eCO_2$ and supplementary water | No differences in mineralization between wheat and peas, but net mineralization was very small in all treatments | Mathers et al. (2012) (paper 306) |
| Measure the effect of $eCO_2$ on crop growth and residual effect of N in residues | Barley and field peas grown at ambient and 550 ppm $CO_2$ in AGFACE; $^{15}N$-labelled residues returned for a subsequent crop | $eCO_2$ increased the growth of N-fertilized barley and field peas; residues stimulated growth of 2nd phase wheat | Available N in pea residues from $eCO_2$ may be less than from ambient $CO_2$ because of higher C/N ratio | Lam et al. (2012) (paper 347) |
| Determine the effect of $eCO_2$ on N dynamics in grain and legume crops | Meta-analysis of 127 literature studies | $eCO_2$ increased crop yields, but this required increased N supply | N supply to crops under $eCO_2$ will require greater fertilizer N inputs and/or greater use of legume intercropping or cover crops | Lam et al. (2012) (paper 341) |
| Determine the effect of $eCO_2$ and N fertilizer rate on wheat and field pea growth | Dual labelling ($^{15}N$ and $^{13}C$) of wheat and field peas grown at ambient and 550 ppm $CO_2$ in AGFACE | $eCO_2$ increased the growth of both crops and increased the shoot/root ratio, but more C allocated to the soil under $eCO_2$ | Effect of $eCO_2$ on crop growth confirmed, but contradictory effects on soil C accumulation | Butterly et al. (2012) (paper 427) |

**Conclusion**

Although these results confirm that elevated $CO_2$ concentrations increase plant growth, this usually requires an increased N supply from the soil, fertilizer, or a previous legume cover crop. Furthermore, the increased growth of leguminous crops may result in an increased C/N ratio in the tissue with the result that N mineralization is depressed when the residues are returned to the soil. Certainly, this increased growth should lead to increased C returns to the soil, but contradictory claims as to whether these returns result in net C accumulation and hence potential sequestration are made by Perry and D'Antuono (1989) and Khan et al. (2007) on the one hand, and Sanderman et al. (2010) on the other. The equivocal nature of these conclusions is summed up by Alterra (2008) who stated 'we have not found strong and clear evidence for either an overall combined positive or negative impact of climate change (raised atmospheric $CO_2$ concentration, temperature, precipitation) on terrestrial carbon stocks. There are suggestions for enhancing soil C stocks at higher atmospheric CO2 concentration and reducing soil C stocks when temperatures are rising.'

**Soil carbon and the mitigation of climate change**

In the Australian government's CFI, sequestering soil C, by which is meant storing C in soil for a long time (100 years is considered to be permanent), is envisaged as one of the mechanisms for reducing (a direct effect) or avoiding (an indirect effect) $CO_2$ emissions, and hence mitigating climate change (Carbon Credits (Carbon Farming Initiative) Act 2011). However, none of the papers reviewed here specifically addressed the question of soil C sequestration and climate change mitigation, possibly because of the explicit or implicit assumption that $CO_2$ emissions from agriculture were contributing to human-induced climate change and therefore that sequestering C in soil was a good thing. In this discussion, I reserve the term 'sequestration' for the long term storage of C in soil; otherwise, I refer to C storage or accumulation.

Certainly, Australian soil scientists have long advocated actions to reverse the trend of declining SOM with the aim of arresting soil degradation (e.g. Vallis et al. 1984), and many have extolled the virtues of SOC for its contribution to 'soil biological, chemical and physical properties and processes (e.g. Baldock et al. 2010). Thus, practices that increase SOC but have little or no benefit for climate change mitigation are likely to be beneficial, emphasizing the point made by Janzen (2006) that the biological turnover of organic C in soils is important for function, not merely the amount accumulated.

With respect to climate change mitigation from agriculture in England and Wales, Powlson et al. (2012) concluded from an extensive survey that there was very limited benefit to be derived from increasing C stocks in agricultural soils, either through greater adoption of reduced or no tillage or through increased applications of organic materials. Their reasons for reaching this conclusion were several: (a) C gains under no-till tended to be overestimated because most comparisons had not been made on the basis of equivalent soil masses, (b) any gains under no-till or reduced tillage tended to be negated by farmers' ploughing every 3-4 years to control weeds and reduce compaction, (c) N2O emissions may be increased under no-till, which could negate the benefit of increased soil C storage, (d) changes in land use and soil management were reversible and therefore not necessarily permanent, and (e) only gains from applying organic materials (e.g. compost, biosolids, farm manures, cereal straw, paper crumble) that were diverted from landfill, or otherwise being disposed of as waste, could be counted as net gains.

Although biochar applications generally resulted in greater C storage and GHG mitigation potential than other practice changes, their effect on crop productivity and soil condition is uncertain because of the lack of data from properly replicated field trials (Powlson et al. 2011, World Bank 2012). However, the GHG mitigation benefit of biochar requires a complete life cycle assessment that covers the source of the organic material, the energy balance in its manufacture and the need for additional fertilizers such as N. The addition of biochar may enhance the stabilization of non-biochar organic C fractions in soil, but more research is required before this can be confirmed (Powlson et al. 2011).

Powlson et al. (2011) argued that converting land from annual cropping to forest, grassland or perennial crops, or slowing the rate of deforestation, were the most effective ways of removing C from the atmosphere or avoiding $CO_2$ emissions. However, converting arable land to pasture that is grazed by ruminants runs the risk of increasing emissions of CH4, which has approximately 23 times the global warming potential of $CO_2$. Previously, Smith et al. (2008) concluded that re-flooding drained organic soils and wetlands offered one of the largest contributions to the biophysical potential for GHG mitigation in agriculture in Europe and globally. While the practice of 'set-aside land' is being phased out in the EU, Sanderman et al. (2010) pointed to the success of the Conservation Reserve Program in the USA in accumulating C in soil, but noted such a scheme required landholders to be paid money for genuine C sequestration. From their survey of Australian and international trial data, Sanderman et al. (2010) concluded that for traditional agricultural systems, Australian soils may be only mitigating losses and not actually accumulating additional atmospheric C. They suggested that the greatest gains in C accumulation could be expected from the conversion cropland to permanent pasture and retirement and restoration of degraded land (marginal land).

Given these sobering national and international reports, it is instructive to calculate the potential contribution of soil C storage to the mitigation of Australia's GHG emissions. Australia's net emissions for the year June 2011 to June 2012 were 550 Mt CO2-e, of which agriculture contributed 16 percent and soil 3 percent (Australian National Greenhouse Accounts 2012). The area of actively 'managed' land is approximately 50 M ha, equally divided into cropland and modified pastures, for which we might assume (from Sanderman et al. 2010) that an average net C accumulation of 0.2 and 0.4 t C/ha-0.3 m, respectively, was achievable annually under best management practices. Sanderman et al. (2010) also suggested that a small accumulation of 0.04 t C/ha-0.3 m might be possible in the less intensively managed 420 M ha of grazing lands. Table 4 shows the results of these calculations, assuming both average and optimum management practices.

The estimated potential reduction in $CO_2$-e emissions of 117 Mt C per year, attributable to soil C storage under optimum management, is comparable to Smith et al.'s (2008) estimate of 100 Mt C per year for Australian soils. However, the assumption of optimum management throughout the large areas involved is unrealistic and the lower estimate of c.70 Mt C per year, or 12 percent of Australia's current GHG emissions is more likely. Looking ahead, even an accumulation rate of 18 Mt C per year may not be sustainable because, as Sanderman et al. (2010) and Powlson et al. (2012) point out, the rate of accumulation is fastest in the first 10-20 years after a change in soil management and declines to a much slower rate as a new equilibrium SOC is approached.

In reality, even by the Department of Climate Change and Energy Efficiency's own projections (Australian Carbon Traders, personal communication), the abatement from soil C sequestration in the year 2020 is expected to be as small as 0.5-1.5 Mt $CO_2$-e, depending on the level of uptake in the CFI. This amounts to 0.1 to 0.3 percent of the current national emissions, which is insignificant.

**Conclusions**

In their 2011 paper, Powlson et al. raised two major concerns (1) that an over-emphasis on sequestering C in soil as a means of climate change mitigation may eclipse other issues that are at least as important, and (2) that the limitations of C sequestration, whether in soil or vegetation, tend to be overlooked and misunderstood, with result that the possibilities for climate change mitigation by this means are exaggerated. With respect to (1), these authors identified the predominant influence of N2O in GHG emissions from agriculture, either lost directly from soil or indirectly contributed through the manufacture of N fertilizers. This issue is also of concern in Australia, especially because the larger crop yields that can return more C to the soil can only be attained through an increased soil N supply.

**Table 4: Summary of the effects of changed land management on the potential accumulation of C in Australian soils and the contribution made to a reduction in national GHG emissions**

| Land use | Approximate area (M ha) | Potential soil C accumulation per year (t C/ha) | Total C accumulation per year (Mt C) | Reduction in $CO_2$-e per year (Mt) | Reduction in overall GHG emissions (%) |
|---|---|---|---|---|---|
| Cropland – average management | 25 | 0.1 | 2.5 | 9 | 1.6 |
| - optimum management | | 0.2 | 5 | 18 | 3.3 |
| Modified pastures – average management | 25 | 0.3 | 7.5 | 28 | 5.1 |
| - optimum management | | 0.4 | 10 | 37 | 6.7 |
| Grazing lands - average management | 420 | 0.02 | 8.4 | 31 | 5.6 |
| - optimum management | | 0.04 | 16.8 | 62 | 11 |
| Combined effect - average management | 470 | | 18.4 | 68 | 12 |
| Combined effect - optimum management | 470 | | 31.8 | 117 | 21 |

Sanderman et al. (2010) appeared to take a more optimistic view of the potential for soil C sequestration in Australian agriculture, possibly because they see that by improving soil management, especially in arable soils, the rate of SOC decline that has prevailed since clearing can be slowed and substantial CO2 emissions avoided at the same time. Consistent with field observations (e.g. Chan et al. 2011, Parry-Jones et al. 2012, Conrad et al. 2012), Sanderman et al. (2010) concluded that the best means of increasing SOC by some 0.3-0.6 t C/ha/yr was by converting arable land to permanent pasture. A disadvantage of this approach is that the soil C gains can be negated by increased CH4 emissions from the ruminants grazing these pastures. Another disadvantage that applies globally is that such conversions decrease the efficiency of resource use (sunlight, land and water) in food production at a time when the global population continues to expand, living standards in developing countries are improving, and the demand for food is increasing.

The Australian government has also taken an optimistic view of soil C storage in formulating its CFI. Provided they adopt an approved methodology, and satisfy an additionality condition, landholders will be awarded Australian Carbon Credit Units (ACCUs, 1ACCU = 1 t CO2-e abated) for genuine soil C sequestration. These ACCUs, which are non-Kyoto units, can be traded only in the small voluntary C market that operates internationally (although the Australian government may also buy ACCUs by tender). The system is flexible in that if a landholder decides at a future time to change to another land use that is more profitable than the accredited one (with its associated ACCUs), he/she can voluntarily relinquish those credits and discharge their 'obligation' by purchasing an equivalent number of credits in the voluntary market. Once relinquished in this way, there is no longer a requirement for the permanence (100 years) of C storage on that land. In this case, the contribution that the ACCUs were making to the abatement of the national emissions falls away (so much for the concept of 'permanence'!).

At present, the value of non-Kyoto units in the international market is approximately $1.25 /t CO2-e, so that given the inevitable compliance costs of accreditation, there is no incentive for landholders to seek approval for a soil C project under the CFI. Even if soil C sequestration became Kyoto-compliant and the units could be traded in the higher-value mandatory market, storing soil C under the CFI is unlikely to induce farmers to change their land use or management unless that change improves the profitability of their farming enterprise.

Accumulating soil C, even at a very low rate, through conservative tillage, including legumes in rotations of crops and pasture, no burning of residues and making use of organic materials that would otherwise be wasted or returned to landfill is a 'no regrets' practice because not only can it contribute to GHG abatement, but it can also improve a soil's productive capacity. This seems to be the approach of EU governments which have decided not to pay farmers to store soil C. However, farmers who receive payments under the Common Agricultural Policy are required to carry out an annual 'Soil Protection Review', which contains questions about measures that should increase or preserve soil C (A. Whitmore, personal communication). Hence the emphasis in the EU is on increasing soil C for productivity purposes rather than climate change mitigation, and this should be the focus of Australian efforts to encourage farmers to increase their SOC.

Given the uncertainties of SOC measurement in a large and diverse landscape, establishing an accurate baseline of SOC values on a range of soil types in different climatic environments under a variable level of management is a huge and costly task. Hence the further development of soil C models, such as RothC embedded in FullCAM, offers the best way forward in developing a robust soil C accounting method, provided that the model can be validated against reliable measurements of soil C and its fractions as it is developed.

**Acknowledgements**

I am most grateful to Emeritus Professor David Powlson, Professor Andy Whitmore and Dr Kevin Coleman of Rothamsted Research in the UK for their comments and information provided; also to Mr Ben Keogh of Australian Carbon Traders for his input.

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# Thursday 6 December

## SOIL CARBON AND CLIMATE CHANGE (ELEVATED $CO_2$)

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# More nitrogen may be required for future grain production systems: A meta-analysis on the effect of elevated [$CO_2$] on nitrogen dynamics

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## Abstract

By 2070, atmospheric carbon dioxide concentration [$CO_2$] is expected to double that observed in 1950. In this higher [$CO_2$] world the sustainability of global crop production may be in jeopardy unless current nitrogen (N) management strategies are changed. There have been many studies that have quantified the effect of increased [$CO_2$] on plant production and N utilization, but the individual results have been generally inconclusive and contradictory. To interpret the information available and provide new insight on crop management in the near future, we examined the effects of elevated [$CO_2$] on N dynamics in grain crop and legume pasture systems using meta-analytic techniques (366 observations from 127 studies). This analysis revealed that elevated [$CO_2$] increases crop production, however, to achieve this increase an adequate supply of N, derived from the soil, fertilizer or biological N-fixation is required. Since N demand and removal in many grain cropping systems is predicted to increase under future $CO_2$-enriched environments, current N management practice needs to be revised. These practices may include higher rates of fertilizer N application, greater use of legume intercropping, or legume cover crops.

## Key Words

Elevated [$CO_2$], meta-analysis, grain N removal, residue C:N ratio, symbiotic $N_2$ fixation, fertilizer N recovery, $N_2O$ emission

## Introduction

Understanding nitrogen (N) removal and replenishment in soil is crucial for sustained crop production under rising atmospheric carbon dioxide concentration ([$CO_2$]). It is unclear how elevated [$CO_2$] will shift the N balance of cropping systems where grain harvest is a significant source of N removal from the system. Previous quantitative reviews show that elevated [$CO_2$] generally stimulated grain yield by 11–54%(Kimball, 1983; Long *et al.* 2006), but reduced the grain protein concentration of $C_3$ non-legumes (10–15%) and had little effect on legumes (–1.4%)(Jablonski*et al.* 2002; Taub*et al.* 2008). However, a quantitative review on the [$CO_2$] effects on the amount of grain N removal (the product of grain yield and grain N concentration of the same crop in the same study) is lacking.

Soil N removed from agroecosystems can be replenished by both fertilizer application and $N_2$ fixation by legumes. Literature on the effect of elevated [$CO_2$] on fertilizer N recovery shows inconsistent results (e.g. Martín-Olmedo*et al.* 2002; Kim *et al.* 2011; Lam *et al.* 2012). While a number of measurements of $N_2$ fixation under elevated [$CO_2$] have been reported (e.g. Edwards *et al.* 2006; Cheng *et al.* 2011), there has been no systematic synthesis of the effects of [$CO_2$] on the amount of N fixed by crop and pasture legumes.

Nitrous oxide ($N_2O$) emissions from soil in agroecosystems play a significant role in accelerating climate change (Forster *et al.* 2007). A recent meta-analysis concluded that elevated [$CO_2$] increases the emission of $N_2O$ in terrestrial ecosystems by 1.2 Tg $N_2O$-N $yr^{-1}$ (van Groenigen *et al.* 2011). However, the interaction of changing N dynamics with increased $N_2O$ emissions from grain crops and legume pasture systems under elevated [$CO_2$] is not well understood.

In this meta-analytic review, we used 300 observations reporting the effects of elevated [$CO_2$] on N dynamics in grain crop and legume pasture systems. The objective of this review was to examine the effect of elevated [$CO_2$] on the demand and supply of N in grain crop andlegume pasture systems.

## Methods

This meta-analysis is based on studies of the effects of elevated [$CO_2$] on N dynamics (grain N removal, $N_2O$ emissions, fertilizer-N recovery, nodule growth, nitrogenase activity and $N_2$ fixation) in cropping systems (cereals, oilseeds, pulses and ley legume pastures) that were conducted in growth chambers, open-top chambers or free-air $CO_2$ enrichment (FACE) facility. The potential patterns of variation in [$CO_2$] effects were also assessed by including categorical variables in the meta-analysis models. These variables include plant functional group ($C_3$ non-legume, legume or $C_4$) and legume type (grain legume or ley pasture legume).The [$CO_2$] effects were averaged across other experimental treatments e.g. water regime (rainfed or irrigated), and fertilizer N input (low and high) where applicable.

We used the natural log of the response ratio ($r$ = response at elevated [$CO_2$]/response at ambient [$CO_2$]) as a metric for analyses (Rosenberg *et al*. 2000). Theseresults were reported as the percentage change under elevated [$CO_2$] (($r$ – 1) ´100). The [$CO_2$] effects on the amount of grain N removal, $N_2O$ emissions and N fixed were also expressed as $U$ = amount at elevated [$CO_2$] – amount at ambient [$CO_2$] (van Groenigen*et al*. 2011).

The effect sizes in our analysis were weighted by replication (Adams *et al*. 1997; van Groenigen *et al*. 2011). Mean effect sizes and 95% confidence intervals were generated by bootstrapping (4,999 iterations) (Adams *et al*. 1997) using MetaWin 2.1 (Rosenberg *et al*. 2000). The effects of elevated [$CO_2$] were considered significant if the confidence intervals did not overlap with zero. Means of different categorical variables were considered significantly different from one another if their 95% confidence intervals did not overlap.

## Results and Discussion

### *Effect of elevated [$CO_2$] on grain N removal*

Overall, elevated [$CO_2$] significantly increased grain N removal by 16.6% (Fig. 1a). This increase suggests that future grain cropping systems will require more N to maintain soil N availability and sustain grain yields. The increased grain N removal under elevated [$CO_2$] resulted from a 26.8% increase in grain yield but an 8.1% reduction in grain N concentration (Fig. 1). The increase in grain N removal of legumes (35.8%) was significantly higher than that of $C_3$ non-legumes (10.7%) and $C_4$ crops (13.9%) (Fig. 1a). This was associated with a significantly greater increase in grain yield of legumes (38.7%) under elevated [$CO_2$] compared to that of $C_3$ non-legumes (23.0%) and $C_4$ crops (23.7%) (Fig. 1b). In contrast, elevated [$CO_2$] resulted in a small and non-significant reduction in grain N concentration of legumes (2.1%), but a greater and significant reduction in grain N concentration of $C_3$ non-legumes (10.0%) and $C_4$ crops (7.9%) (Fig. 1c).

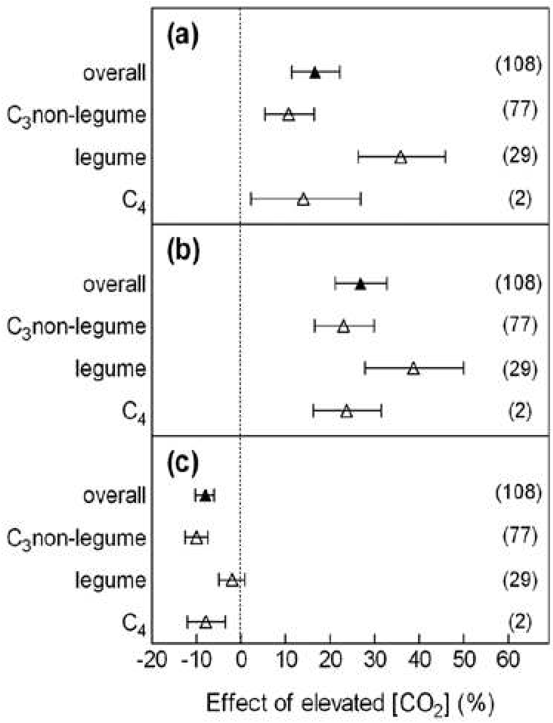


**Figure 1. Effect of elevated [$CO_2$] on (a) grain N yield, (b) grain yield and (c) grain %N. Means and 95% confidence intervals are depicted. Numbers of experimental observations are in parentheses.**

### *Effect of elevated [$CO_2$] on $N_2$ fixation*

Our analysis shows that elevated [$CO_2$] significantly increases whole plant nodule number (32.7%), nodule mass (39.1%), and total nitrogenase activity (36.7%), but resulted in a non-significant increase in the proportion of plant N derived from the atmosphere (Ndfa)(9.6%). Nonetheless, the amount of Ndfa was significantly increased by 38.2% under elevated [$CO_2$] (Fig. 2). This stimulation is associated with greater availability of photoassimilate for nodules and the host plant under elevated[$CO_2$] (Rogers *et al.* 2009). The amount of N fixed by legumes might be underestimated in our synthesis because fixed N in nodulated roots was not examined in some studies included in our database.

*Effect of elevated [$CO_2$] on fertilizer N recovery*

Our meta-analysis indicates that elevated [$CO_2$] increases fertilizer N recovery in the plant by 16.8% (Fig. 3a). Five out of 11 observations in the database of fertilizer N recovery examined the recovery of fertilizer N in the soil-plant system. These five observations show that the recovery of fertilizer N in the soil-plant system was generally not affected by elevated [$CO_2$]. The effect of elevated [$CO_2$] on the distribution of assimilated fertilizer N in different plant parts has rarely been examined, but one study (using $^{15}N$ tracer technique) indicated that elevated [$CO_2$] increased the remobilization of fertilizer derived N into grain by 46% under ambient temperature and 14% under elevated temperature (ambient +2°C) (Kim *et al.* 2011), suggesting that more fertilizer N will be removed in grain and a higher fertilizer application rate will be required to replenish the soil N pool under future elevated $CO_2$ atmospheres.

*Effect of elevated [$CO_2$] on nitrous oxide emission*

Our analysis indicates that overall, elevated [$CO_2$] increases $N_2O$ emission by 27.0% (Fig. 3b). The increase was associated with a 24.0 and 33.4% increase in above-ground and below-ground biomass, respectively, at the experimental location where $N_2O$ fluxes were measured. This finding suggests an association between $N_2O$ emission and soil C substrate availability. While $N_2O$ emission has little effect on the N balance of a cropping system, the [$CO_2$]-induced increase in $N_2O$ emission suggests that denitrification is likely to be stimulated under elevated [$CO_2$]. This was observed in a study where denitrification increased under elevated [$CO_2$] (Baggs and Blum, 2004). This increase was attributed to greater C substrate input (Baggs *et al.* 2003) and/or improved soil moisture (Leakey*et al.* 2009) under increased [$CO_2$].

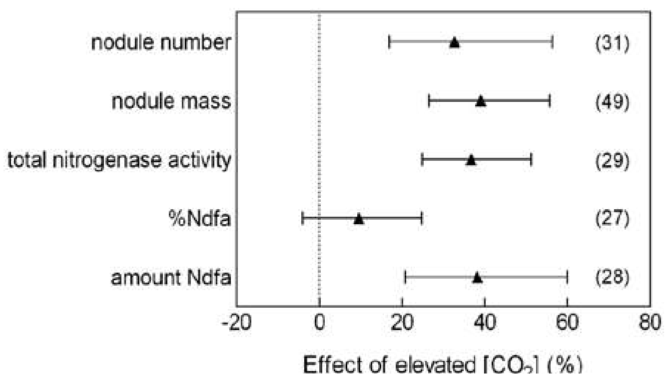


**Figure 2: Effect of elevated [CO2] on nodule number, nodule mass, total nitrogenase activity, %N derived from the atmosphere (Ndfa) and amount of Ndfa. Means and 95% confidence intervals are depicted. Numbers of experimental observations are in parentheses.**

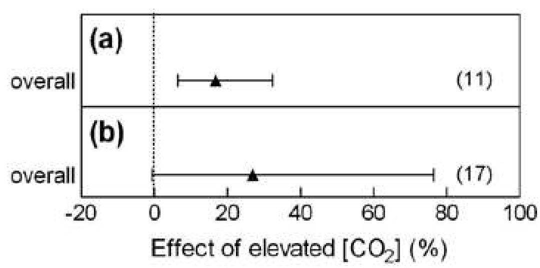


**Figure 3: Effect of elevated [CO2] on (a) fertilizer N recovery in plant and (b) N2O emission from soil. Means and 95% confidence intervals are depicted. Numbers of experimental observations are in parentheses.**

*[CO2]-induced changes in N balance*

We evaluated the effect of elevated [$CO_2$] on N balance in various cropping systems by the [$CO_2$]-induced changes in N input and N output of the systems (Table 1). Growing pasture legumes under elevated [$CO_2$] (without altering fertilizer inputs) resulted in a net N gain of 53.0 kg N $ha^{-1}season^{-1}$, while a net N loss occurred for grain legumes (–35.2 kg N $ha^{-1}season^{-1}$), $C_3$ non-legumes (–12.6 kg N $ha^{-1}season^{-1}$) and $C_4$ crops (–12.0 kg N $ha^{-1}season^{-1}$) (Table 1). The results suggest that N demand and removal in grain cropping systems will increase under elevated [$CO_2$]. The extra amount of N fixed under elevated [$CO_2$] by grain legumes can partly compensate for the additional N removal of grain crops in grain legume rotation. Incorporating pasture legumes into a crop rotation will increase N input to cropping systems under future higher [$CO_2$] atmospheres (Table 1).

**Table 1: [CO2]-induced changes in N balance in various cropping systems.**

| | $[CO_2]$-induced changes in | | | | | | |
|---|---|---|---|---|---|---|---|
| | Grain N removal (I) | | $N_2O$ emission (II) | | Amount of N fixed (III) | | N balance (III – I – II) |
| Plant system | mean | 95% CI | mean | 95% CI | mean | 95% CI | |
| | kg N $ha^{-1}$ $season^{-1}$ | | | | | | |
| $C_3$ non-legume | 12.4 | 4.6 to 20.4 | 0.22 | –0.06 to 0.50 | 0 | NA | –12.6 |
| Grain legume | 59.6 | 35.8 to 86.7 | 0.60 | 0.13 to 1.06 | 25.0 | 5.3 to 53.0 | –35.2 |
| Pasture legume | 0 | NA | –0.04 | –0.12 to 0.05 | 53.0 | 28.3 to 81.1 | 53.0 |
| $C_4$ | 11.8 | 1.5 to 22.1 | 0.16 | –0.04 to 0.36 | 0 | NA | –12.0 |

In summary, our analysis suggests that N demand and removal in grain cropping systems will increase under elevated $[CO_2]$. Extra N input will be necessary to maintain soil N availability and sustain crop yields. This additional N may be obtained via applying higher rates of fertilizer N, greater use of legume intercropping, or legume cover crops under future elevated $CO_2$ atmospheres.

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# $^{13}CO_2$ pulse-labelling of wheat and field pea: Effect of elevated $CO_2$ and N rate

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## Abstract

Future increases in atmospheric carbon dioxide ($CO_2$) concentration are expected to increase plant growth. However, few studies have considered the effects of $CO_2$ concentration on below-ground processes. We conducted a column experiment at the soil free-air carbon dioxide enrichment (SoilFACE) facility to investigate the interactive effect of $CO_2$ concentration and nitrogen (N) fertiliser application on carbon (C) and N partitioning. Wheat and field pea plants were grown at ambient (370 ppm) or elevated (550 ppm) $CO_2$ concentrations and with two levels of N fertiliser (40 and 100 mg N/kg soil) in columns containing a Vertosol soil. Crops were pulse-labelled with $^{13}CO_2$ at seven times throughout the growing season. The N fertiliser was added to soil as solution in four separate applications. This study showed that elevated $CO_2$ increased the growth of both crops with the increase being greater for shoots than for roots. In addition, greater $^{13}C$ enrichment of plant tissue, rhizosphere and bulk soil under elevated $CO_2$ indicated a greater allocation of C to soil than under ambient $CO_2$ concentration. This effect was reduced when plants were grown at the high Nrate.

## Key Words

FACE, carbon and nitrogen partitioning, climate change.

## Introduction

Elevated $CO_2$ ($eCO_2$) concentrations are expected to increase plant growth due to enhanced photosynthesis and water and N-use efficiency (Kimball *et al*., 2002). However, little information exists for semi-arid agricultural systems, which are practiced in southern Australia, and in particular few studies have considered changes in C and N partitioning in annual cropping systems. Above- and below-ground biomass responses to $eCO_2$ are highly variable, particularly between seasons (Ainsworth and Long, 2005; Lam *et al*., 2012). Traditional techniques used to quantify below-ground C allocation may underestimate actual amounts due to incomplete root recovery and lack of sensitivity. In contrast, stable isotope approaches, such as $^{13}C$ pulse-labelling, provide better estimates of C in plant and soil pools. The aim of this study was to quantify the effects of $eCO_2$ concentration and N rate on C and N partitioning in the plant-soil system of wheat and field pea.

## Methods

### *Column experiment details*

A Vertosol soil (Isbell, 1996) was collected from a virgin site adjacent to the SoilFACE facility, Department of Primary Industries, Horsham. The soil had a $pH_{CaCl2}$ of 7.7 and total C and N contents were 14.2 and 0.8 g/kg respectively. Sieved (<4 mm) Vertosol was mixed 1:1 with triple washed white sand (mean diameter 430 µm) to aid root harvest, and 12 kg (dry weight equivalent) of the soil: sand mix was added to plastic columns (15 cm ID × 60 cm long) creating a bulk density of 1.2 g/cm$^3$. The following basal nutrients, in mg/kg, were applied to the soil: sand mix: $CaCl_2$ 186; $CoCl_2$ 0.4; $CuSO_4$ 6; $FeCl_3$ 0.6; $K_2SO_4$, 103; $KH_2PO_4$, 70.4; $MgSO_4$ 122; $MnSO_4$ 6; $Na_2B_4O_7$ 1.6; $Na_2MoO_4$ 0.4 and $ZnSO_4$ 8. Soil columns were adjusted to 80% field capacity and sown with either wheat (*Triticum aestivum* L. cv. Yitpi) or field pea (*Pisum sativum* L. cv. PBA Twilight) which had been inoculated with Group E inoculum. The soil columns were transferred to the SoilFACE facility on the 16$^{th}$ June 2011. SoilFACE consists of eight bunkers (3.7 m wide) in a completely randomised 4 × 2 arrangement with either ambient $CO_2$ ($aCO_2$; 370 ppm) or elevated $CO_2$ ($eCO_2$; 550 ppm) concentrations. The $eCO_2$ concentration represents the projected atmospheric $CO_2$ concentration for the year 2050. The FACE technology is described in detail in Mollah *et al*. (2009).Wheat and field pea plants were thinned to three and five plants/column respectively after 3 weeks.

### *$^{13}C$ and $^{15}N$ isotope labelling*

Dual $^{13}C$ and $^{15}N$ isotope labelling was performed at various times during the 2011 growing season. $^{13}C$ pulse-labelling was performed seven times (24$^{th}$ August, 15$^{th}$ September, 21$^{st}$ September, 6$^{th}$ October, 12$^{th}$ October, 19$^{th}$ October and 31$^{st}$ October) using an air-tight chamber (140 cm × 140 cm × 150 cm). Briefly, soil columns containing plants were removed from the bunkers, placed within the chamber and sealed at the base with PVC tape. The chamber was covered with black plastic and air within the chamber was diverted through an alkali trap (1 M NaOH) to lower the ambient $^{12}CO_2$ concentration. After 1 h, the alkali trap was isolated, $Na_2{}^{13}CO_3$ was reacted with acid inside the chamber to generate $^{13}CO_2$ and give a target $CO_2$ concentration of 1000 ppm. Air within the chamber was allowed to mix for 10 min before uncovering the chamber. Labelling continued for ~3.5 h before returning the columns to bunkers. Plants were also labelled with $^{15}N$ by direct application of $Ca(^{15}NO_3)_2$ solution to soil. Plants received two N application rates, 40 or 100 mg N/kg, which were added as four equal applications of either 10 or 15 mg N/kg at sowing and at three times (25th August, 15th September, 5th October) during the growing season. Columns only received additional water when N was added (10% of field capacity) and in the final 3 weeks before sampling (20% of field capacity).

*Sampling and analyses*

At peak biomass (2nd November 2011), columns were removed from SoilFACE and destructively sampled. Plant shoots were removed, rinsed with reverse osmosis (RO) water and dried at 70°C for 3 days. The soil was separated into depths of 0-10 cm, 10-25 cm and 25-50 cm. Roots were carefully removed, and rhizosphere soil was collected by shaking. The remaining bulk soil was thoroughly mixed and subsampled. All soils were air-dried at 25°C. Roots were washed with RO water, dried at 70°C for 3 days and then ground (<0.5 mm) using a Retsch ZM200 centrifugal mill. Ground plant material and whole soil was thoroughly mixed and subsamples were ground to a fine powder using a Retsch MM400 mixer mill (Retsch GmbH, Haan, Germany) for analyses. Total C and N, $^{13}C$ and $^{15}N$ content were determined using an Isotope Ratio Mass spectrometer (IRMS) (Hydra 20-20, SerCon, Crewe, UK). The $\delta^{13}C$ values are calculated from the isotope ratio ($^{13}C/^{12}C$) of the sample against standard Pee Dee Belemnite (PDB) in parts per thousand ($^0/_{00}$).

*Statistical analyses*

For each species, a two-way analysis of variance (ANOVA) in a randomised complete block split-plot design was used to test the effects of $CO_2$ concentration (main-plots) and N level (sub-plots) on shoot and root parameters and soil properties. To improve the accuracy and interpretation, root parameters were analysed separately for each depth. Significant ($P$ = 0.05) differences between means were established using the least significance difference (LSD) test.

**Results and Discussion**

*Plant growth*

Shoot dry weight (DW) was greater ($P$=0.05) at the $eCO_2$ than at the $aCO_2$concentration for both wheat and field pea (Table 1). At $aCO_2$ concentration, N level had no effect on plant shoot growth. This suggested that N was not limiting wheat shoot growth at $aCO_2$ concentration. However, under $eCO_2$ wheat shoot growth was enhanced at the higher N level, indicating an increased N demand. In contrast, N level did not affect field pea growth at either $CO_2$ concentration. Legume species can fulfil their N requirement via $N_2$ fixation. The effect of $eCO_2$ and N level on root growth was much less than that of shoots (Table 2). Root growth of field pea was not affected by $CO_2$ concentration. In contrast, wheat root growth increased under $eCO_2$ but only in the deeper (25-50 cm) soil layer. A greater increase in root growth under $eCO_2$ was expected. A number of studies have shown that $eCO_2$ stimulates wheat root growth (Wechsung *et al*., 1999; Lam *et al*., 2012) and increases the root: shoot ratio (Lou *et al*., 2006). N level had no effect on root growth for either species.

**Table 1: Effect of CO2 concentration and N level on shoot dry weight, C:N ratio and 13C in shoots of wheat and field pea. Values in brackets are the standard error of the mean (n=3).**

| $CO_2$ conc. (ppm) | N rate (mg/kg) | Shoot DW (g/column) | Shoot C:N | $^{13}C$ in shoot ($\delta$ PDB) |
|---|---|---|---|---|
| ***Wheat*** | | | | |
| 370 | 40 | 27.7 (0.3) | 39.9 (1.2) | 348 (57) |
| 550 | 40 | 37.0 (0.4) | 46.2 (0.1) | 463 (20) |
| 370 | 100 | 28.9 (0.8) | 21.3 (1.3) | 149 (8) |
| 550 | 100 | 43.2 (1.8) | 27.0 (0.7) | 251 (36) |
| ***Field Pea*** | | | | |
| 370 | 40 | 26.0 (0.6) | 17.8 (0.4) | 242 (7) |
| 550 | 40 | 32.3 (0.5) | 15.7 (0.4) | 225 (31) |
| 370 | 100 | 25.9 (1.4) | 17.4 (0.7) | 243 (12) |
| 550 | 100 | 35.2 (1.5) | 17.5 (0.2) | 236 (42) |

**Table 2: Effect of $CO_2$concentration and N level on root dry weight and C:N ratio, and 13C partitioning in wheat and field pea roots, rhizosphere and bulk soil. Values in brackets are the standard error of the mean (n=3).**

| $CO_2$ conc. (ppm) | N rate (mg/kg) | Root DW (g/column) | Root C:N | $^{13}C$ in root ($\delta$ PDB) | $^{13}C$ in rhizo. soil ($\delta$ PDB) | $^{13}C$ in bulk soil ($\delta$ PDB) |
|---|---|---|---|---|---|---|
| ***Wheat 0-10 cm*** | | | | | | |
| 370 | 40 | 6.36 (0.18) | 76.2 (2.5) | 172 (28) | -1.1 (3.5) | -7.6 (2.2) |
| 550 | 40 | 7.02 (0.34) | 104.7 (10.5) | 255 (17) | 4.5 (3.5) | -7.7 (1.9) |
| 370 | 100 | 6.55 (0.20) | 38.2 (1.7) | 112 (23) | -3.1 (3.2) | -10.5 (0.9) |
| 550 | 100 | 7.09 (0.21) | 45.0 (2.1) | 134 (40) | 0.1 (5.3) | -5.8 (2.3) |
| ***Wheat 10-25 cm*** | | | | | | |
| 370 | 40 | 5.08 (0.18) | 49.6 (1.3) | 198 (26) | 6.0 (4.0) | -13.9 (1.3) |
| 550 | 40 | 5.22 (0.17) | 63.0 (2.1) | 266 (21) | 32.8 (18.3) | -10.6 (1.3) |
| 370 | 100 | 5.22 (0.09) | 29.5 (1.4) | 107 (17) | -0.8 (1.6) | -14.9 (1.1) |
| 550 | 100 | 5.53 (0.16) | 37.7 (1.4) | 188 (58) | 10.1 (1.0) | -13.1 (0.6) |
| ***Wheat 25-50 cm*** | | | | | | |
| 370 | 40 | 7.57 (0.34) | 46.0 (1.9) | 181 (34) | 24.4 (9.2) | -12.6 (1.2) |
| 550 | 40 | 8.75 (0.30) | 54.6 (2.3) | 233 (28) | 51.6 (11.9) | -7.9 (2.2) |
| 370 | 100 | 6.82 (0.25) | 28.7 (0.7) | 104 (20) | 7.4 (4.4) | -17.0 (0.3) |
| 550 | 100 | 8.46 (0.17) | 32.0 (1.0) | 152 (45) | 22.7 (4.5) | -14.1 (0.8) |
| ***Field Pea 0-10 cm*** | | | | | | |
| 370 | 40 | 4.67 (0.04) | 16.8 (0.3) | 139 (20) | -4.9 (0.9) | -14.5 (0.2) |
| 550 | 40 | 4.73 (0.03) | 16.3 (0.7) | 111 (9) | -8.7 (2.5) | -12.6 (0.6) |
| 370 | 100 | 4.69 (0.08) | 17.1 (0.4) | 141 (10) | -12.4 (1.3) | -16.1 (0.4) |
| 550 | 100 | 4.83 (0.11) | 16.6 (0.5) | 108 (27) | -9.9 (1.6) | -14.4 (1.2) |
| ***Field Pea 10-25 cm*** | | | | | | |
| 370 | 40 | 4.57 (0.07) | 17.4 (0.4) | 158 (32) | -12.1 (1.8) | -16.0 (0.7) |
| 550 | 40 | 4.72 (0.08) | 17.2 (1.3) | 122 (18) | -12.2 (1.8) | -16.2 (0.3) |
| 370 | 100 | 4.59 (0.03) | 18.1 (1.1) | 164 (16) | -13.1 (0.8) | -16.7 (0.3) |
| 550 | 100 | 4.57 (0.03) | 17.3 (0.5) | 136 (26) | -9.5 (1.2) | -16.4 (1.0) |
| ***Field Pea 25-50cm*** | | | | | | |
| 370 | 40 | 5.18 (0.07) | 17.4 (0.5) | 167 (20) | -7.1 (0.8) | -19.0 (0.3) |
| 550 | 40 | 5.46 (0.10) | 18.0 (1.9) | 116 (17) | -8.3 (2.8) | -18.0 (0.8) |
| 370 | 100 | 5.21 (0.14) | 17.8 (1.7) | 160 (15) | -7.8 (3.2) | -18.4 (0.3) |
| 550 | 100 | 5.49 (0.11) | 16.5 (0.2) | 134 (32) | -7.2 (3.4) | -17.9 (0.4) |

*13C in plant roots and shoots*

$CO_2$ concentration did not affect $^{13}C$ enrichment of field pea shoots and roots (Tables 1 and 2). However, a greater $^{13}C$ enrichment of wheat shoot and root occurred at $eCO_2$ than at $aCO_2$. Further, the $\delta^{13}C$ of roots did not change with depth. The greater enrichment of wheat shoots under $eCO_2$ could indicate a greater uptake of $^{13}CO_2$ or a greater retention of $^{13}C$ within the shoot. This is in contrast to what was expected since enhanced photosynthesis under $eCO_2$ has been shown to dilute the $\delta^{13}C$ value in other studies (Leavitt *et al*., 2001). Since repeated pulse-labelling was used in this study, the $\delta^{13}C$ value of plant material represents both recently assimilated C and the $^{13}C$ pulse assimilated at previous times through the season. It is possible that the greater $\delta^{13}C$ under $eCO_2$ was influenced by the soil water status when the final $^{13}C$ pulse-labelling was undertaken. Plants growing at $eCO_2$ could have been less water stressed than those at $aCO_2$ and assimilated a greater proportion of the final $^{13}C$ pulse. Enhanced water-use efficiency under $eCO_2$ is commonly reported in FACE experiments (Garcia *et al*., 1998; Kimball *et al*., 2002). In contrast to $CO_2$ concentration, $^{13}C$ enrichment in wheat shoots and roots was greater at the lower rather than at the higher N application rate. In general, the effect of N on $\delta^{13}C$ was greater than that of $CO_2$ concentration. The decrease in $\delta^{13}C$ at higher N application rates could indicate enhanced photosynthesis of wheat under $eCO_2$ and the subsequent dilution of the $\delta^{13}C$ value due to a greater concentration of $^{12}C$ photosynthates. Alternatively, a lower proportion of the final $^{13}C$ pulse could have been incorporated into wheat biomass at the high N rate compared to the low N rate. As previously stated, it is possible that plants were under water stress when the last pulse-labelling was undertaken. If wheat plants growing at the high N level were water stressed, reduced photosynthetic activity may explain the lower $\delta^{13}C$ value. However, photosynthesis was not quantified in the current experiment. Interestingly, the difference in $\delta^{13}C$ between wheat roots and shoots grown at low N was much greater than that of plants growing at the high N level.

*$^{13}C$ in rhizosphere and bulk soils*

The amount of $^{13}C$ in the rhizosphere soil was proportional to that in the roots but much lower in magnitude (Table 2). In general, $eCO_2$ increased $\delta^{13}C$ of the wheat rhizosphere soil, which was greater at the low N compared to the high N rates. However, this was only significant ($P$ = 0.05) for deeper wheat roots (25-50 cm) due to the large variability in the data. A lower $\delta^{13}C$ at the high N level may be due to $^{12}C$ dilution via enhanced photosynthesis or alternatively lower $^{13}C$ uptake due to drought stress, as outlined earlier. The amount of $^{13}C$ in field pea rhizosphere soil was lower than that of wheat. Furthermore, in contrast to that of field pea, the rhizosphere soil of wheat had greater $\delta^{13}C$ values with depth. This indicates a greater contribution of C to soil by wheat than field pea and the allocation of this C to newer root biomass, which would have mostly occurred in the 25-50 cm layer. Part of the $^{13}C$ present within the rhizosphere and bulk soil is expected to be associated with roots that were not recovered at sampling, root fragments (root tips, fine root turnover) and exudates, and also the soil microbial biomass. While $^{13}C$ may also be associated with more stable soil C pools, this is difficult to determine from the current study. Since pulse-labelling was performed ~48 h prior to sampling $^{13}C$ is most likely associated with more labile C pools.

As expected, $^{13}C$ enrichment of the bulk soil was lower than the rhizosphere soil and showed less variability between treatments (Table 2). For field pea bulk soil, the $\delta^{13}C$ value of the 0-10 cm layer was significantly ($P$ = 0.05) higher at 100 mg/kg N than 40 mg/kg N. However, N rate did not affect $\delta^{13}C$ in any other depth and no effect of $CO_2$ concentration was observed. For wheat, both $CO_2$ and N level significantly affected $\delta^{13}C$ of the 25-50 cm bulk soil. These results showed that the increased $^{13}C$ content of wheat biomass and release into the rhizosphere was sufficient to increase $^{13}C$ in the bulk soil. Further, increasing N supply decreased the $\delta^{13}C$ of the bulk soil. The differences in $^{13}C$ in root and soil fractions between treatments were much greater than that for root biomass. The results suggest that $^{13}C$ in the soil pools was not simply due to differences in exudation as a function of root mass. Furthermore, $^{13}C$ enrichment of the bulk soil generally decreased with depth, which is opposite to findings for the rhizosphere soil. It is likely that $^{13}C$ in the surface layer (0-10 cm) of the bulk soil originated from earlier pulse-labelling while $^{13}C$ in the rhizosphere, particularly the deeper layer (25-50 cm), from the final pulse-labelling. However, $^{13}C$ originating from separate pulse events cannot be differentiated in the current study. The $\delta^{13}C$ values therefore represent a complex interaction between the amount and location of exudate release and the subsequent stabilisation or decomposition of these compounds over time. Further studies should use a $^{13}C$ pulse-chase approach to quantify the effects of $CO_2$ concentration and N level on $^{13}C$ allocation in the plant-soil system at different growth stages. Nevertheless, $\delta^{13}C$ of the bulk soil was enriched in all treatments as compared to the natural abundance of this soil (ca. -23.5$^{o}/_{oo}$; data not shown).

## Acknowledgements

This research was supported by an Australian Research Council Linkage Project (LP 100200757). SoilFACE was funded by DPI Victoria and the Grains Research & Development Corporation.

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# Elevated $CO_2$ enhances phosphorus immobilisation in rhizosphere of chickpea and wheat grown in soils with phosphorus fertilizer history

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Knowledge of how crops access the phosphorus (P) pools in soil under elevated atmosphere $CO_2$ ($eCO_2$) will be required for the efficient management of P in cropping systems in future climate-changed environments. This study examined the effect of $eCO_2$ on changes in various P fractions in the rhizosphere of chickpea (*Cicer arietinum* L.) and wheat (*Triticum aestivum*. L.) grown in both a Vertosol and a Calcarosol soil with or without a P fertilizer history. Plants were grown in rhizoboxes for 6 weeks and then the rhizosphere soil was subjected to a sequential inorganic/organic P fractionation procedure. The $NaHCO_3$-extractable inorganic P (Pi) in the rhizosphere was depleted by both plant species but was not significantly affected by $eCO_2$. However, $NaHCO_3$-extractable organic P (Po) accumulated in both soils, especially under $eCO_2$ with a P fertilizer history ($P \leq 0.001$), while NaOH-Po accumulated only in the Vertosol under $eCO_2$ with a P history ($P \leq 0.001$). Elevated $CO_2$ generally did not alter the relative ability of the two crop species to access any P pool in either soil. The exception was wheat, which depleted $NaHCO_3$-Pi and NaOH-Pi in rhizosphere more than did chickpea. Elevated $CO_2$ increased microbial C in the rhizosphere by an average of 21%. The pool size of the Po fraction in the rhizosphere correlated positively with the microbial C, but not with soil phosphatase activity or rhizosphere pH. We conclude that elevated $CO_2$ increased Po fractions in the rhizosphere of chickpea and wheat, particularly in the soils with a history of P fertilization. This P immobilisation under $eCO_2$ appears to have resulted from increased microbial activity in the rhizosphere of plants growing with $eCO_2$.

# Interactions and feedbacks between above and below-ground ecosystems under $eCO_2$ and eT

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Rising atmospheric carbon dioxide ($CO_2$) concentrations and elevated temperatures (eT) are of global concern. Forest soils have been identified as potential C-sinks under future climate forcing, as carbon inputs to soil are generally predicted to increase under $eCO_2$. However, a key determinant of an ecosystems ability to efficiently sequester C is the availability and sustainability of other nutrients including nitrogen (N). This presents great uncertainty in predicting the global capacity of terrestrial ecosystems to sequester C because there are major gaps in our understanding of the interactions and feedbacks between the carbon and nitrogen cycles. This uncertainty is in part driven by limited understanding of the interactions and feedbacks between above- and below-ground ecosystems under climate forcing. Key to this is improving our understanding of microbial-mediated nutrient cycling in response to multiple climate change factors ($eCO_2$ and eT). Using whole tree chambers at the Hawkesbury Forest Experiment, we determined the interactive effect of $eCO_2$ (+200 ppm above ambient) and eT (+3 °C above ambient) on the structure and activity of soil microbial communities in soils growing under Eucalyptus globulus. Total soil respiration varied with time and treatment and was generally more responsive to eT than $eCO_2$. N-mineralisation was largely unaffected by both $eCO_2$ and eT, but a large increase in the microbial C:N ratio under $eCO_2$ and eT treatments, that became stronger over time, indicated increasing N-limitation within these systems. Further, the strong interactive effect of $eCO_2$ and eT, suggests that nutrient poor soils may have limited capacity to sequester C in the long-term.

# Grass leaves emit nitrous oxide.

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Nitrous oxide ($N_2O$) emissions from grazed pastures are considered to be mainly through soil nitrification and denitrification. There are a few studies in the literature that provide evidence that N2O emissions can occur from plant leaves. In this study we investigated whether the leaves of grass species in New Zealand grazed pastures emit significant fluxes of $N_2O$.

First we developed and tested an apparatus to measure $N_2O$ fluxes from leaves and to separate these from soil emissions. We then conducted a number of experiments in a climate controlled cabinet to measure $N_2O$ fluxes from ten different grass species, the C3 species *Pennisetum clandestinum, Poa annua, Lolium perenne, Bromus hordeaceus, Agrostis capillaris, Holcus lanatus, Dactylis glomerata* and *Anthoxanthum odoratum*, and the C4 species *Digitaria sanguinalis, Pennisetum clandestinum* and *Paspalum dilatatum*. The grasses were grown in a potting medium containing sand, peat and a commercial nutrient mixture. The plants were placed in a growth cabinet with a light intensity of 200 µmol/sec/m2, a temperature regime of 15°C night and 20°C day and a 12 h photoperiod. The plants were allowed to adjust to conditions for 2 days before $N_2O$ flux measurement. After the measurements were completed we harvested the leaves and measured leaf areas. Here we report the data from 3 measurement occasions from 3 plants of each species.

$N_2O$ fluxes from leaves ranged from 0.00 to 0.21 mg $N_2O$-N/ $m^2$ leaf area/ hour with mean and median 0.03 and 0.02 mg $N_2O$-N/ $m^2$ leaf area/ hour respectively. The highest mean flux was from Paspalum dilatatum ($0.07 \pm 0.02$, $n = 9$) and the lowest from *Poa annua* ($0.005 \pm 0.001$, $n = 9$). When leaf area index was assumed as one, the fluxes were within the range of those measured from urine patches in dairy and sheep systems.

# Thursday 6 December

## SOIL CARBON AND CLIMATE CHANGE (ELEVATED CO2)

*Presented Posters*

Back to contents

# Spatial heterogeneity of soil-derived $CO_2$ efflux in three tropical Australian savannas

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Savannas consist of a mix of trees and grasses that ultimately leads to local heterogeneity in soil properties. Here we present two years of monthly soil-derived $CO_2$ efflux measurements from three tropical savannas in far north Queensland, undertaken with the aim to quantifying sources of variation in soil-derived $CO_2$ efflux in terms of climate and tree/grass abundance. All three sites are *Eucalyptus* open woodlands with a perennial grassy understorey at two sites and annual grasses at the third. The perennial grass sites have sandy soil on granite and clayey soil on basalt whereas the drier annual grass site has clayey alluvial soil. Measurement locations were stratified based on proximity to trees and for analysis grouped according to tree canopy cover. At the perennial grass sites, fluxes were consistently higher at tree locations, with measurements close to trees ~1.5 – 2 times greater than measurements remote from trees. At the annual grass site, fluxes close to trees were higher during the dry season (~2 times that of the grasses), this also being observed during periods of inundation. Nevertheless, the difference between tree and grass locations decreased or even reversed during the wet season, this corresponding to the period of rapid biomass accumulation by the grass layer. The seasonality of fluxes also depended on location, being highest in the open areas dominated by grasses. Soil-derived $CO_2$ efflux in tropical savannas is thus both temporally and spatially heterogeneous suggesting that reasonably complex models may be required to accurately estimate carbon losses from these systems.

## Soil-atmosphere greenhouse gas exchange in a biofiltration system

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Biofilters have been demonstrated to effectively remove pollutants and reduce the quantity of stormwater. They utilize the biogeochemical processes that occur in soils and plants to capture and transform pollutants and water. A typical biofilter consists of a filter media (0.7-1 m soil +/ sand ± organic material), vegetation, structures for inflow, outflow, ponding and overflow, and optionally under-drains. The biogeochemical processes that remove pollutants from the water (i.e. respiration, denitrification) produce greenhouse gases that diffuse through soil and can be released to the atmosphere. This is the first study to quantify soil greenhouse gas fluxes from a biofilter. $CO_2$, $CH_4$ and $N_2O$ were measured over a year at the Monash University biofilter. This system treats runoff from a multistorey car park. Half of the system has a raised outlet, creating a continuously submerged zone in the soil at depth. Average gas fluxes from the biofilter were $CO_2$ 101.49 mg/m$^2$/h, $CH_4$ -16.42 µg/m$^2$/h and $N_2O$ 13.78 µg/m$^2$/h without a submerged zone and $CO_2$ 92.70 mg/m$^2$/h, $CH_4$ -4.16 µg/m$^2$/h and $N_2O$ 65.62 µg/m$^2$/h with a submerged zone. $CO_2$ fluxes increased as soil temperature increased. $CH_4$ and $N_2O$ were affected by soil water content, with occasional very large emissions in wet conditions contrasting with generally moderate emissions of $N_2O$ and moderate uptake of $CH_4$. The submerged zone consistently created wetter soil conditions. Further studies of C and particularly N cycling in biofilters are required to optimize their design to remove pollutants from stormwater while minimizing emission of greenhouse gases.

# Soil carbon cycling under urine patches

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Soil carbon (C) cycling under urine patches is an important part of agricultural sustainability. This area requires further investigation as urine deposition may lead to decreases in soil C, and whether losses increase with intensity is not known. Soil C solubilisation is a mechanism by which soil C could be lost from the profile by either leaching or priming. In air dried pine and pasture soils, following treatment with urine, we measured soil C solubilisation of 11-28% of soil C concentration. While this was valuable information, we still needed to determine whether the soil C now in the liquid phase was available for loss. We measured priming of soil C decomposition in pine and pasture soils treated with urine; losses were between 4-5% of soil C concentration, whereas leaching of soil C from pasture soil was only 0.4%. The minor losses of soil C by leaching infer that this may not be the predominant pathway under patches, but that priming of soil C decomposition may be. Further investigation using simultaneous measurement of leaching and priming under controlled and field conditions is required.

# Fire modification of the forest floor and surface soil in *Eucalyptus obliqua* forests of the Otway Ranges, Victoria

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Prescribed fire partially oxidizes surface litter and fine fuel fractions, deposits charcoal and ash on the soil surface, and in places, directly alters surface soil organic matter. Incomplete combustion of surface and soil organic matter produces pyrolytic compounds of low to medium molecular weight that have a lasting influence on soil properties and the reactivity of soil organic matter. This study investigated the impact of prescribed fire in lowland forests of the Otway ranges on litter, humus and soil organic matter quantity and quality. Fire-induced mass loss of litter and humus were measured and soil to 10cm depth was analysed for total C and characterized by mid infra red spectroscopic scans, pyrolysis-GC/MS and acid digests to describe fire impacts on soil organic matter quality and quantity. When comparing the spectra of soils before and after fire, aliphatic carbon near the 2930cm$^{-1}$ region reduced after fire and new peaks for aromatic carbon that represent charcoal were found in the fingerprint region of post-fire soil spectra. The chemical modification of the forest floor by fire, as described by these methods, is discussed particularly the application of pyrolysis-GC/MS to detect and characterize the black carbon in forest soils produced by prescribed fire.

# Does cropping rotation and liming affect greenhouse gas emissions from a semi-arid soil?

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We investigated if including a grain legume and/or lime in a cropping rotation altered greenhouse gas emissions from acidic soil. Nitrous oxide ($N_2O$) and methane ($CH_4$) were measured from a rain-fed, cropped soil in a semi-arid region of south-western Australia for two years on a sub-daily basis. The randomised-block design included two cropping rotations (lupin-wheat, wheat-wheat) by two liming treatments (0, 3.5 t ha$^{-1}$) by three replicates. The lupin-wheat rotation only received N fertilizer during the wheat phase (20 kg N ha$^{-1}$), while the wheat-wheat received 125 kg N ha$^{-1}$ in total. Fluxes were measured using automated chambers connected to a gas chromatograph. Nitrous oxide fluxes were low (1.4 to 9.2 g $N_2O$-N ha$^{-1}$ day$^{-1}$), and including a grain legume in the rotation did not enhance emissions. Total $N_2O$ losses were approximately 110 g $N_2O$-N ha$^{-1}$ after two years for both rotations when averaged across liming treatment. Liming decreased $N_2O$ emissions from the wheat-wheat rotation by 31% by lowering emissions following summer/autumn rainfall, but had no effect on $N_2O$ emissions from the lupin-wheat rotation. Daily $CH_4$ fluxes ranged from -14 to 5 g $CH_4$-C ha$^{-1}$ day$^{-1}$. Total $CH_4$ uptake was lower from the wheat-wheat rotation (601 g $CH_4$-C ha$^{-1}$) than from either lupin-wheat rotations (967 g $CH_4$-C ha$^{-1}$); liming the wheat-wheat rotation increased uptake (1078 g $CH_4$-C ha$^{-1}$) to a value similar to the lupin-wheat rotations. Including a grain legume in cropping rotations decreases greenhouse gas emissions from grain production by lowering $CO_2$ emissions from fertilizer production and urea hydrolysis.

# A procedure for assessing the impacts of climate change on soils – case studies from NSW

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Significant anthropogenic climate change (CC) is becoming a demonstrable reality. The latest IPCC Fourth Assessment Report (AR4) presents a disturbing picture of rapidly increasing $CO_2$ levels, temperatures and sealevels, with coincident changes to common climate variables such as rainfall max/mins, averages and extremes. Significant impacts from CC are likely over the mid to long term, and soils are very likely to be affected by a wide range of new climatic situations. The high diversity of soil types in NSW makes it difficult to determine precise impacts of CC in each specific location; however rules of thumb can be used to assess the likely impacts. We present a series of criteria that build to determine likely CC impacts on soils. These include examination of climatic indices such as: a) the absolute value of the change of the climatic measure, and its trajectory, b) the rate of change in the climatic measure, c) likely extremes in the climatic measure and their frequency of occurrence, d) possible cyclic phenomena, e) potential threshold or catastrophic events, and f) potential feedback mechanisms. Soil parameters examined include: a) intrinsic vulnerability of particular soils and/or landscapes to changes in the climatic measures, b) proximity to thresholds, c) interactions between soils and plant communities, and d) interactions between soils and land management. Examples from NSW of possible interactions between projected climate change and soils are presented as case studies.

# Thursday 6 December

## SOIL FERTILITY AND SOIL CONTAMINANTS (NUTRIENT MANAGEMENT)

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# Sustainable management of nutrient returns in excreta on grazed dairy soils

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## Abstract

Nutrient recycling and deposition in dung and urine from grazing cows can contribute to the uneven distribution and accumulation of nutrients on dairy farms. The locations of cows in space and time were documented on ten Australian dairy farms on each of five visits over a year. Information and samples of the diets consumed were collected at the same time. Analysis of this data indicated that cows spend a small proportion of their time in dairy sheds (2%) and yards (8%) where nutrients are likely to be collected for reuse, with similar time spent in feedpads (8%) and holding areas (2%). However, when analysed only for farms with feedpads, cows spent twice as much time in these locations. High nutrient accumulation can occur in these areas, and needs careful management to reduce its impact on nutrient losses. The largest amounts of nutrients (26, 5 and 39 t of N, P and K respectively) were recycled by cows to grazed paddocks, but these were not uniformly returned, with accumulations occurring in specific parts of the landscape. Improved nutrient management will depend on an understanding of the spatial and temporal patterns of animal movement and the implications for nutrient return and accumulation.

**Key Words**
Spatial, temporal, nutrient distribution, heterogeneity, phosphorus, nitrogen.

## Introduction

Dairy production systems are often in net positive nutrient balance (Gourley *et al*. 2012) with potential for losses of nutrients from farms to waterways and the atmosphere (Gourley and Ridley 2005, Monaghan *et al*. 2007). Nutrient losses are often associated with the accumulation of nutrients in parts of the landscape where transport pathways exist (Monaghan *et al*. 2007). Data from grazed dairy farms around Australia indicate considerable heterogeneity in nutrient concentration around farms (Gourley *et al*. 2010). Nutrient application in the form of fertiliser or effluent from dairy sheds and animal management practices such as night paddocks, strip-grazing and holding areas can lead to large amounts of nutrients deposited onto specific paddocks within a farm. In addition to farm management practices, animal behaviour can contribute to nutrient accumulation zones due to the propensity for dairy cows to congregate around water (waterways and troughs), under trees and on slopes.

Within grazed dairy systems better understanding of the spatial and temporal location of dairy cows can lead to a greater awareness of the potential for heterogeneous nutrient distribution and contribute to management practices that minimise nutrient accumulation. An approach was tested on ten dairy farms around Australia where the location of cows in space and time was documented. In conjunction with feed data, nutrient deposition in management units around the farms was estimated.

## Methods

### *Farm locations*

A subset of ten farms participating in the Accounting for Nutrient project was sampled (Gourley *et al*. 2010). The farms were located in Gippsland, the South West and the Northern regions of Victoria, New South Wales, Queensland and Western Australia, and represent different grazing management systems across Australia (Table 1).

### *Data and sample collection*

On each of five visits (February, May, August and November 2008 and February 2009), farmers were asked to identify all locations on their farms that their lactating herds had visited in the previous 24 hours and the time the cows spent in each of these locations. Locations were grouped into the following management units: 'Paddocks', 'Dairyshed' (where cows are milked), 'Yards' (holding areas adjacent to the dairyshed), 'Laneways', 'Feedpad' (areas where cows are fed, which may or may not be concreted) and 'Holding areas' (other areas where cows are confined).

Samples of the diet offered to the lactating herds were collected as well as information about the amounts fed and any wastage. Feed samples were analysed for metabolisable energy and nutrient (crude protein, phosphorus, potassium, sulphur) content. Milk samples collected from the vats were analysed for nutrient, fat and protein content. Feed nutrients were calculated and then subtracted from the milk nutrients produced to estimate the nutrients excreted by the herd.

**Results**

Cows spent on average 2 and 8% of their time in the dairyshed and adjacent yards respectively, both of which are usually concreted and where excreted nutrients are likely to be collected for reuse (Table 2).

| Region | Farm No | Visit | Herd Type§ | No of Cows | Paddocks | Laneways | Yards | Dairyshed | Feedpad | Holding areas |
|---|---|---|---|---|---|---|---|---|---|---|
| NSW | 25 | Feb-08 | 1 | 46 | 79% | 9% | 11% | 1% | | |
| | | May-08 | 1 | 46 | 86% | 6% | 8% | 1% | | |
| | | Aug-08 | 1 | 55 | 84% | 6% | 8% | 2% | | |
| | | Nov-08 | 1 | 45 | 86% | 9% | 5% | 1% | | |
| | | Feb-09 | 1 | 58 | 88% | 4% | 6% | 1% | | |
| | 44 | Feb-08 | 1 | 184 | 57% | 5% | 21% | 2% | 15% | |
| | | May-08 | 1 | 210 | 47% | 8% | 7% | 2% | 35% | |
| | | Aug-08 | 1 | 220 | 73% | 6% | 9% | 1% | 10% | |
| | | Nov-08 | 1 | 212 | 79% | 4% | 8% | 1% | 8% | |
| | | Feb-09 | 1 | 200 | 79% | 4% | 9% | 2% | 7% | |
| SW VIC | 28 | Feb-08 | 1 | 543 | 69% | 16% | 14% | 1% | | |
| | | May-08 | 1 | 253 | 89% | 5% | 5% | 1% | | |
| | | Aug-08 | 1 | 534 | 84% | 6% | 9% | 1% | | |
| | | Nov-08 | 1 | 556 | 81% | 10% | 8% | 1% | | |
| | | Feb-09 | 1 | 528 | 85% | 7% | 7% | 1% | | |
| | 29 | Feb-08 | 1 | 240 | 54% | 0% | 22% | 3% | 21% | |
| | | May-08 | 1 | 249 | 45% | 4% | 7% | 2% | 42% | |
| | | Aug-08 | 1 | 261 | 45% | 3% | 6% | 3% | 43% | |
| | | Nov-08 | 1 | 315 | 72% | 5% | 8% | 2% | 14% | |
| | | Feb-09 | 1 | 292 | 61% | 6% | 6% | 2% | 26% | |
| | | Feb-08 | 2 | 240 | 54% | 2% | 16% | 3% | 25% | |
| | | May-08 | 2 | 249 | 33% | 5% | 13% | 2% | 47% | |
| | | Aug-08 | 2 | 260 | 37% | 5% | 8% | 2% | 49% | |
| | | Nov-08 | 2 | 315 | 55% | 4% | 6% | 2% | 33% | |
| | | Feb-09 | 2 | 291 | 49% | 5% | 14% | 2% | 30% | |
| NIR | 33 | Feb-08 | 1 | 163 | 77% | 4% | 7% | 2% | 11% | |
| | | May-08 | 1 | 148 | 85% | 4% | 6% | 1% | 4% | |
| | | Aug-08 | 1 | 126 | 86% | 5% | 7% | 2% | 0% | |
| | | Nov-08 | 1 | 182 | 69% | 4% | 9% | 1% | 17% | |
| | | Feb-09 | 1 | 143 | 71% | 3% | 8% | 1% | 16% | |
| | 34 | Feb-08 | 1 | 203 | 86% | 6% | 6% | 2% | | |
| | | May-08 | 1 | 240 | 60% | 5% | 11% | 2% | | 22% |
| | | Aug-08 | 1 | 160 | 84% | 7% | 7% | 2% | | |
| | | Nov-08 | 1 | 249 | 83% | 5% | 11% | 2% | | |
| | | Feb-09 | 1 | 221 | 79% | 10% | 10% | 2% | | |
| WA | 40 | Feb-08 | 1 | 192 | 86% | 5% | 7% | 2% | | |
| | | May-08 | 1 | 269 | 84% | 5% | 9% | 2% | | |
| | | Aug-08 | 1 | 172 | 87% | 4% | 7% | 2% | | |
| | | Nov-08 | 1 | 266 | 85% | 4% | 9% | 2% | | |
| | | Feb-09 | 1 | 196 | 85% | 6% | 7% | 2% | | |
| | 43 | Feb-08 | 1 | 430 | 86% | 6% | 7% | 1% | | |
| | | May-08 | 1 | 537 | 76% | 11% | 12% | 1% | | |
| | | Aug-08 | 1 | 388 | 78% | 13% | 7% | 1% | | |
| | | Nov-08 | 1 | 346 | 88% | 5% | 6% | 1% | | |
| | | Nov-08 | 2 | 196 | 80% | 6% | 13% | 1% | | |
| | | Feb-09 | 1 | 175 | 49% | 2% | 14% | 1% | 34% | |
| | | Feb-09 | 2 | 310 | 34% | 6% | 5% | 1% | 54% | |
| QLD | 5 | Feb-08 | 1 | 109 | 43% | 7% | 8% | 1% | 8% | 33% |
| | | May-08 | 1 | 99 | 60% | 6% | 6% | 1% | 19% | 8% |
| | | Aug-08 | 1 | 118 | 62% | 9% | 6% | 1% | 7% | 14% |
| | | Nov-08 | 1 | 113 | 62% | 9% | 6% | 1% | 7% | 14% |
| | | Feb-09 | 1 | 128 | 61% | 9% | 7% | 1% | 7% | 14% |
| Gipps | 11 | Feb-08 | 1 | 550 | 89% | 3% | 7% | 2% | | |
| | | May-08 | 1 | 486 | 81% | 11% | 6% | 1% | | |
| | | Aug-08 | 1 | 695 | 82% | 9% | 8% | 2% | | |
| | | Nov-08 | 1 | 680 | 85% | 6% | 7% | 1% | | |
| | | Feb-09 | 1 | 400 | 90% | 2% | 7% | 1% | | |

**Table 1: Percent time cows spent in different management units on ten commercial Australian dairy farms.**

**Table 2: Mean percent time spent by lactating dairy cows in management units on ten Australian dairy farms, on farms with feedpads and farms without feedpads**

| Management unit | All farms¥ | Feedpads | No feedpads |
|---|---|---|---|
| Paddocks | 74.12% (0.021) | 65.22% (0.030) | 83.01% (0.013) |
| Laneways | 6.24% (0.004) | 5.85% (0.005) | 6.63% (0.006) |
| Yards | 8.38% (0.004) | 8.84% (0.007) | 7.92% (0.004) |
| Dairyshed | 1.59% (0.001) | 1.61% (0.001) | 1.56% (0.001) |
| Feedpad | 7.58% (0.018) | 15.16% (0.029) | 0% (0.000) |
| Holding areas | 2.1% (0.009) | 3.32% (0.016) | 0.89% (0.009) |

¥ data in parenthesis are standard errors of the means

Where feedpads were present, dairy cows spent a mean of 15% of their time on them (Table 2), which would lead to the deposition of large amounts of nutrients in these areas. An estimated 19 and 3 kg nitrogen (N) and phosphorus (P) respectively would be deposited on a feedpad over a 300-day lactation by each cow in a dairy herd. However in only two of these farms were the feedpads concreted to assist with collection of excreta for reuse, suggesting that nutrients that could be recycled in these systems are potentially wasted. Providing supplementary feeding on feedpads or in holding areas has become more prevalent in Australian grazed dairy systems as a means of increasing milk production (Thorrold and Doyle 2007).

On all the dairy farms studied, the greatest proportion of time was spent on grazed pastures, even where feedpads or holding areas were used (Table 2). Based on dietary intake and milk production, the daily nutrients excreted by cows on Farm 28 were calculated (Table 3). Nitrogen, P and potassium (K) returned to pastures in a year were 26, 5 and 39 t respectively, while only 3, 0.6 and 4.5 t were deposited in the dairy shed and yards (Table 4). From an agronomic point of view these amounts are important when applied to the grazing area as a whole, amounting to 90 and 20 kg/ha/year N and P respectively. The N and P imported onto this farm in fertiliser over the year were 74 and 8 kg/ha respectively (data not presented).

**Table 3: Calculated nutrients excreted (g/cow/day) by lactating dairy cows from Farm 28, a commercial dairy farm in south west Victoria.**

| Nutrients excreted (g/cow/day) | | | |
|---|---|---|---|
| N | P | K | S |
| 159.2 | 29.2 | 236.5 | 19.8 |

**Table 4: Annual deposition of nutrients (t) by lactating dairy cows in management units around Farm 28.**

| | Annual nutrients deposited by lactating cows | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | N | | P | | K | | S | |
| | t | t/ha | t | t/ha | t | t/ha | t | t/ha |
| Paddocks | 26.1 | 0.09 | 4.8 | 0.02 | 38.8 | 0.14 | 3.2 | 0.01 |
| Laneways | 2.5 | 0.35 | 0.5 | 0.06 | 3.8 | 0.52 | 0.3 | 0.04 |
| Dairy sheds and yards | 3.0 | 13.3 | 0.6 | 2.4 | 4.5 | 19.8 | 0.4 | 1.7 |

The nutrients returned by animals are seldom deposited uniformly across the grazing areas of the farm. From the annual records of the daily locations of dairy cows on Farm 28, cows returned more frequently to certain paddocks, consequently depositing nutrients unevenly across the farm (Figure 1a). The heterogeneous distribution of nutrients was exacerbated when the movements of other animals such as dry cows, springers and heifers were considered (Figure 1b). The return of these nutrients to paddocks poses a risk to the environment if they accumulate in parts of the landscape where pathways for nutrient movement occur.

### Conclusion

The spatial and temporal movement of dairy cows appears to contribute to the heterogeneous distribution of soil nutrients observed on many dairy farms. Additionally, as the use of supplementary feeding increases, dairy cows are spending an increasing amount of time in areas where the nutrients they excrete are concentrated and not immediately available for recycling. The loss of nutrients from production, the potential for nutrient contamination from feedpads and holding areas, and the accumulation of nutrients in parts of the landscape are substantial factors that may hinder sustainable management of grazed dairy systems.

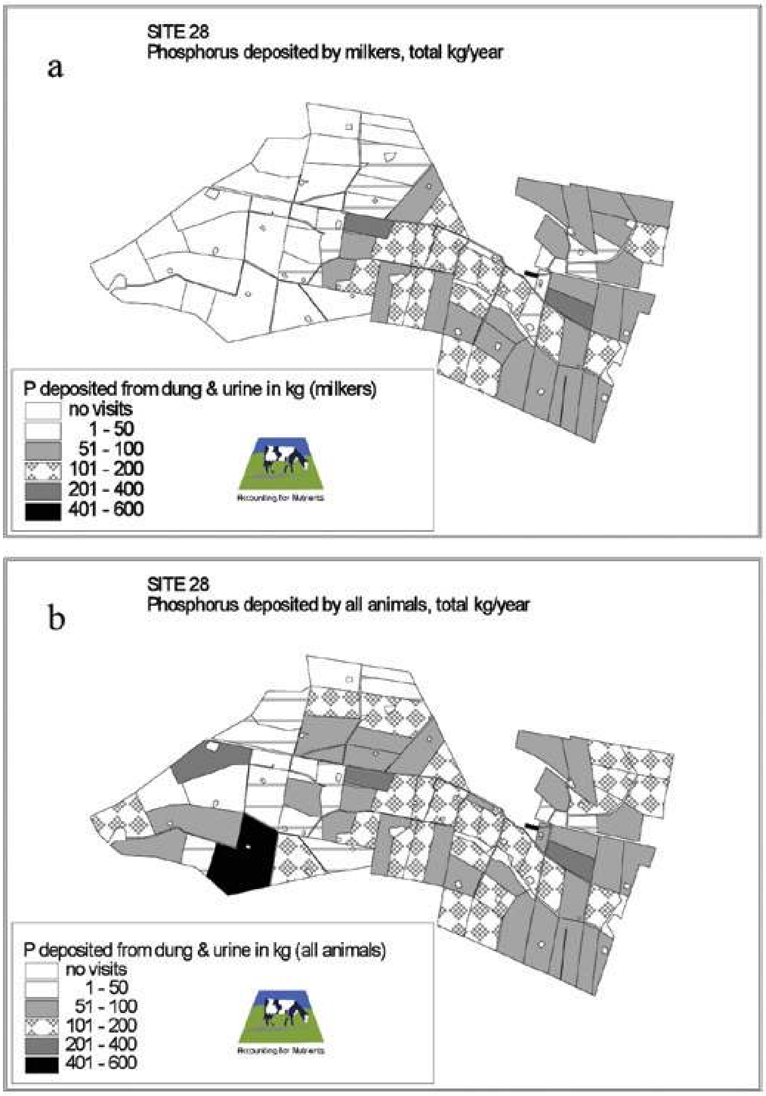


**Figure 1: Phosphorus (kg) deposited in dung around Farm 28 over 421 days by a) lactating dairy cows and by b) all cows (lactating, dry, heifers, springers).**


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# When is enough, enough? Determining production gains from nitrogen, phosphorus and potassium inputs on Australian dairy farms

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**Abstract**

The cost of manufacturing fertilisers, principally associated with the cost of fossil fuels, is expected to rise substantially in real terms in coming decades and consequently fertiliser use will become a larger part of dairy farm operating costs. Many farms have already high soil fertility levels with limited production gains from further fertiliser expenditure. In this study we investigated nitrogen (N), phosphorus (P) and potassium (K) whole-farm balances and existing soil nutrient levels on 41 contrasting dairy operations across Australia. We found that there are large excesses of N, P and K on a broad range of dairy operations across Australia and that nutrient surpluses and within farm nutrient distribution was related to stocking rate and milk production per ha. While there was a positive relationship ($P<0.001$) between milk production and N fertiliser inputs, there was no overall relationship between P and K fertiliser inputs and milk production. We recommend comprehensive soil testing to guide fertiliser decisions on dairy farms and more sophisticated quantification of nutrient flows through feed and manure deposition to determine manure nutrient loading rates across the farm for highly stocked dairy operations.

**Key Words**

Nutrient balance, soil test, fertiliser**.**

**Introduction**

Total N, P and K inputs onto dairy farms, mainly in the form of feed and fertiliser, are usually much greater than the outputs of P, K and N in milk, animals, and crops (VandeHaar and St-Pierre 2006). These surpluses tend to increase as farms intensify and stocking rates increase. Excess P and K on dairy farms can result in increasing soil P and K levels beyond agronomic requirements (Weaver and Reed, 1998). Unlike P and K, N is not significantly buffered by soils, and where N is applied in high concentrations such as in dung, urine or fertiliser, losses through volatilization and leaching can be high (Rotz *et al*., 2005). The challenge of optimising the production potential and profitability of nutrient inputs in animal agriculture while reducing negative environmental effects is faced by most industrialized countries (Steinfeld *et al*. 2006) including Australia. Animal agriculture is now commonly recognised as the dominant contributor of nutrient inputs to water bodies because of its extensiveness (Department of Water 2010).

**Methods**

Forty-one commercial dairy farms representing a broad range of geographic locations, soil types, milk production, herd and farm size, and reliance on irrigation, were selected for this study (Gourley *et al*. 2012). Farm area ranged from 47 to 496 ha, cow numbers per farm ranged from 59 to 1930 cows, and number of paddocks per farm ranged from 14 to 141. Whole-farm nutrient balances were determined for N, P and K as described by Gourley *et al*. (2012). Milk production from home-grown feed was determined for each farm as the difference between total annual milk production and that produced from imported feed (Heard *et al*. 2011). Soil nutrient levels were determined by sampling all paddocks used for pasture and crop production on each farm, as well as areas with high animal densities (holding areas, feeding areas, sick paddocks, bull paddocks). Thirty 0-10 cm soil cores were collected along a diagonal transect across each area, bulked, dried (40ºC) and ground (2 mm sieve) and analysed for pH (0.01M CaCl2), Olsen extractable P, Colwell P and K, and P buffering index (PBI). More than 1860 paddock samples were included in the analysis.

**Results and Discussion**

*Farm-scale nutrient balances*

Whole-farm N surplus (the difference between total nutrient imports and total nutrient exports) ranged from 47 to 601 kg N $ha^{-1}$ $yr^{-1}$ with a median of 226 kg N $ha^{-1}$, and N use efficiency (the ratio of total nutrient exported in product divided by total nutrient imported at the farm scale) ranged from 14 to 50%, with a median of 26%. We also found a strong correlation between total N imported and milk production per ha (Fig. 1a) while N surplus was also strongly related to milk production (Fig. 1b) with the slope of this linear relationship (0.0121; SE = 0.0015) providing an estimate of the productivity N surplus, equivalent to 12.1 g N $litre^{-1}$ milk produced. Whole-farm P surplus ranged from -7 to + 133 kg P $ha^{-1}$ $yr^{-1}$, with a median of 28 kg $ha^{-1}$. Phosphorus use efficiencies ranged from 6 to 158%, with a median of 35%. Potassium balances ranged from 13 to 452 kg K $ha^{-1}$, with a median value of 74 kg K $ha^{-1}$; K use efficiency ranged from 9 to 48%, with a median value of 20%.

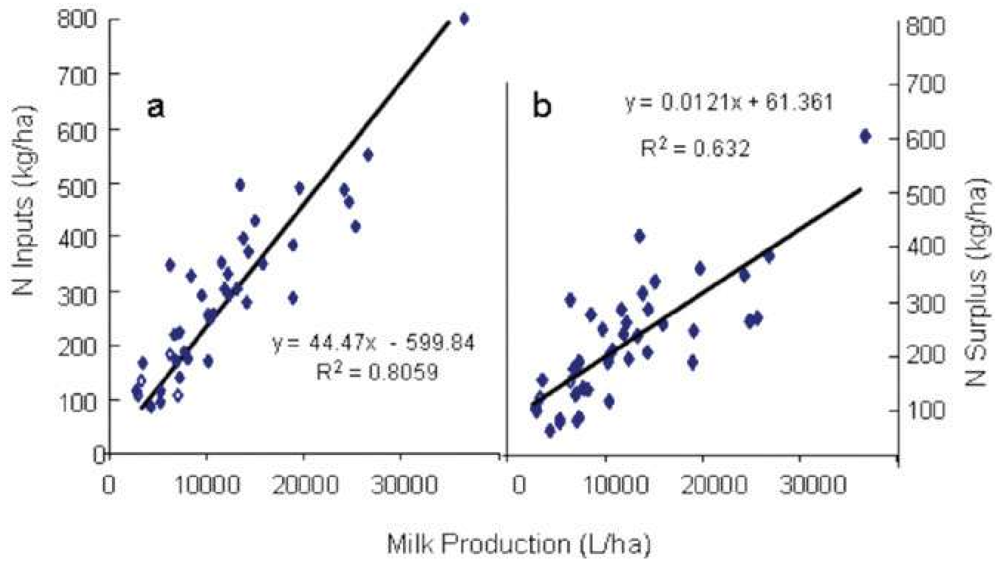


**Figure 1: Relationships between milk production and (a) whole-farm nitrogen inputs and (b) nitrogen surplus for 41 contrasting dairy farms across Australia. Unshaded symbols represent organic dairy farms.**

*Soil P and K levels*

There was a large range in soil P and K levels from grazed pasture paddocks (Figure 2). Olsen P levels ranged between 3 and 189 mg $kg^{-1}$ and the Colwell K levels ranged from 14 to 3400 mg $kg^{-1}$. Only 20% of the paddocks sampled had soil P or K values below the recommended agronomic optimum (Olsen P of 20 mg $kg^{-1}$ and Colwell K of 180 mg $kg^{-1}$), while 50% of the paddocks sampled were at least 1.5 times the recommended agronomic optimum. At the high fertility end, 20% of paddocks sampled had Olsen P or Colwell K levels at least 3 times the agronomic requirements. Areas with high animal densities, such as calving paddocks, feed pads, holding areas and 'hospital' paddocks, had substantially elevated soil nutrient levels when compared to the overall pasture paddocks. In contrast, low intensity areas such as 'other animal' and areas with trees had much lower fertility levels.

*Production gains from fertiliser inputs*

Milk production from home-grown feed increased with increasing N fertiliser input, although there was a high variation and uncertainty around milk production gains (Fig. 3a), suggesting substantial improvements in the utilization of N by grown pasture could be achieved on many farms. The relationship improved slightly when additional N from the fixation of atmospheric nitrogen by pasture legumes was included (Fig. 3b).

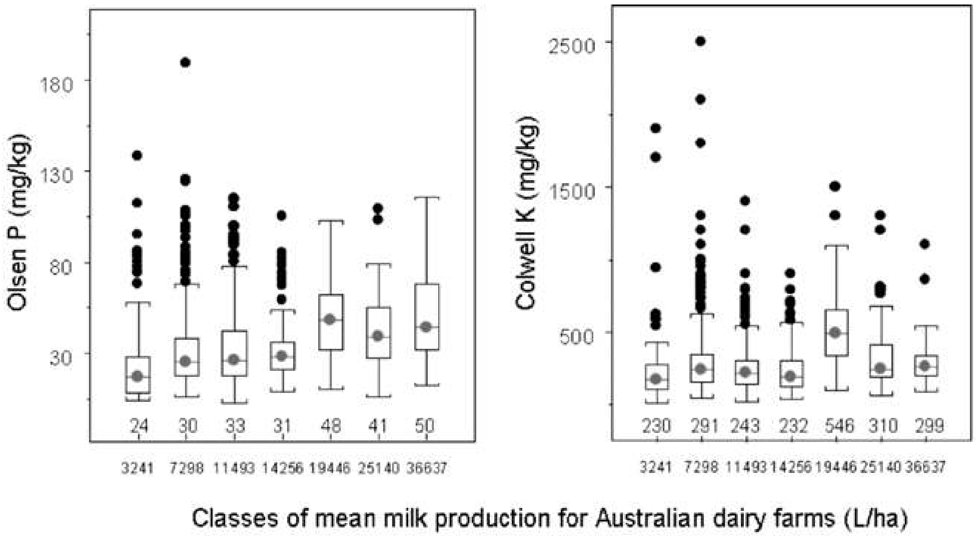


**Figure 2: Box and whisker plots of soil P and K levels for individual paddocks for dairy farms grouped into annual milk production classes (litres ha-1). Mean soil test levels for each production class are below the plots.**

In contrast there was no relationship between P and K fertiliser applications and milk production attributed to home-grown feed (Fig. 3c and d). This lack of relationship is supported by the generally high levels of soil P and K measured. Under these conditions, additional pasture and crop production from the application of P and K fertiliser would not be expected and therefore neither would an associated increase in milk production from home grown feed. Moreover, the milk production from farms with low or no P or K fertiliser inputs but with adequate levels of soil P or K, suggest that these soil reserves can be utilized without a resulting decline in milk production.

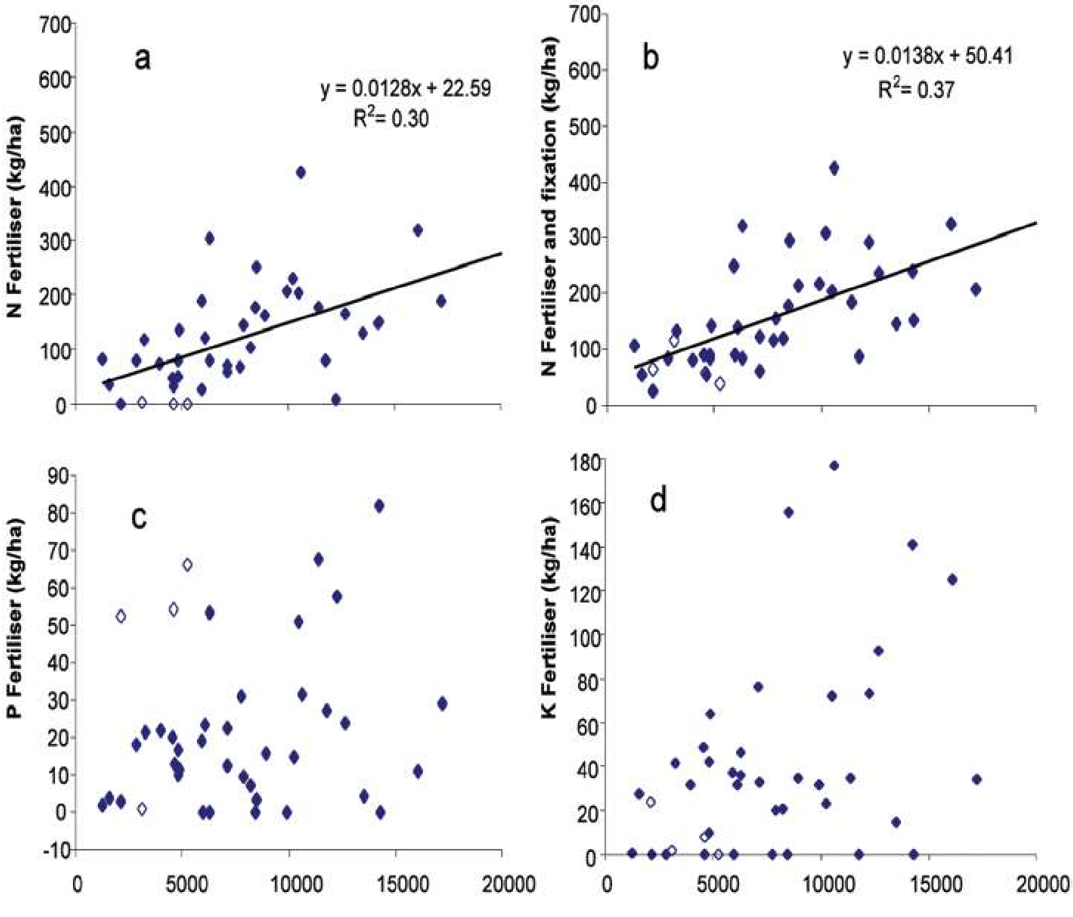


**Figure 4: Relationships between milk production from home-grown feed and (a) N fertiliser input, (b) N fertiliser plus N fixation, (c) P fertiliser input and (d) K fertiliser input.**

## Conclusions

The quantity of nutrients imported onto these diverse dairy operations varied markedly, with feed and fertiliser generally the most substantial imports. Elevated N, P and K surpluses on dairy farms were related to milk production and stocking rate per ha. While N fertiliser input was a driver of milk production from home-grown pasture and crops, there was a high degree of uncertainty around production gains. In contrast, the lack of a relationship between P and K fertiliser inputs and milk production from home-grown pasture and crops reflected the high soil P and K levels measured on these farms.

Within farm soil nutrient heterogeneity is substantial, irrespective of the intensity of the dairy operation. Higher soil nutrient levels of P, K (and N) are driven by paddock stocking density, proximity to the dairy, frequency of effluent applications, and feeding strategies. These results support recent assessments in Australia that suggest that excess nutrients may be cycling in some agricultural systems and placing considerable pressure on the environment (OECD 2008). Soil testing to determine available soil P and K is therefore an important tool to manage P and K build-up and maintenance.

These results suggest that there are opportunities on many dairy farms to reduce or exclude fertiliser inputs. The relatively small costs associated with a strategic and on-going soil sampling program are likely to be returned many times over through the potential savings in unnecessary fertiliser expenditure. Where applications are warranted, appropriate rates and blends of fertiliser should ensure profitable increases in pasture and crop productivity. In the future, more sophisticated approaches will likely be required which quantify nutrient flows through the continuum of feed, milk production and manure; increases the capture and storage of excreted manure; determines nutrient loading rates in areas across the farm; and ensures that fertiliser and manure are applied under favourable soil and climatic conditions for optimum plant uptake.

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# Potassium cycling and the practice of standing cows off: Risks and opportunities

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## Abstract

As dairy farmers strive to reduce their environmental footprint while maintaining or improving productivity, the frequency and duration of standing cows off paddocks is increasing. While many of the benefits of standing cows on feedpads are understood, the challenges that this practice present to nutrient management have not been comprehensively researched. Standing cows on feedpads impacts on nutrient cycles in a number of inter-related ways. The major effects of feedpad use on nutrient dynamics relate to the reduction in excreta deposition to paddocks, and consequently, the capture and management of this excreta as slurry and liquid effluent.While nitrogen has received some attention, there have been very few studies of the impacts of standing cows on feedpads on the potassium (K) cycle.

The effects of standing cows off paddocks on the K cycle within dairy pastures have been investigated over 3 years at Massey University's No 4 Dairy Farm as part of a much larger investigation of duration-controlled grazing (*DC*).As far as K is concerned, the *DC* treatment represented a form of standing cows on feedpads between grazings and returning effluent from a bunker at the end of the pad. *DC* grazing was compared with standard grazing (*SG*) on a series of 14 plots. A range of soil, plant and water parameters were monitored.

Typically, feedpad bunker slurry has low K levels relative to its total solids content. The use of feedpad bunker slurry on the *DC* plots resulted in a net loss of 137 kg K/ha/yr from these areas compared with the *SG* plots. Consequently, herbage K concentrations and soil QT K were significantly lower in the *DC* plots by the second season.Also, the cumulative annual losses of K in drainage were lower from *DC* plots compared with *SG*. This may have been not only due to less total K return, but also more even return of K, compared to K deposited in urine spots on the *SG* plots.

If *DC* grazing is to be implemented at a farm scale, it is important that the K content of all effluent sources is known and is considered in farm nutrient management plans.

## Introduction

Many New Zealand dairy farmers stand their cows on feedpads for prolonged periods in order to protect soils and pasture from treading damage. Standing cows off paddocks also reduces nutrient losses to water ways. This practice is becoming more common and more important as it is a good fit with the New Zealand dairy industry's objective of increasing productivity while reducing its environmental footprint, with particular reference to nutrient loss to waterways (PCE 2012).

The management of effluent and/or slurry generated on pads is challenging. Typically, the excreta deposited on feedpads is scraped into a bunker at the end of the pad. Rainwater on the pad also enters the bunker. In order to minimise the size of the bunker, the liquid component of this slurry is sent to the farm's effluent ponds. A weeping wall will often be used to separate the phases.

Standing cows on feedpads is likely to impact on nutrient cycles and nutrient management. While nitrogen usually receives the most attention(Christensen *et al.* 2012), standing cows on feedpads may also affect potassium (K). Standing cows off paddocks will affect the K cycle in numerous ways; most obviously, there will be less excreta (particularly urine) deposited in the paddocks. While, in theory, it is possible to return the K in the excreta placed on the feedpad back to the paddocks, this is often not practical, not least of all because of the draining of much of the urine from the bunker. Also, different systems used for land application of effluent, bunker manure and pond sludges will impact on the K loading to different areas of the farm.

This paper briefly examines the effects of standing cows off (i.e. the *DC* grazing) on herbage, soil and drainage water K concentrations, the total amount of K lost as leaching (kg/ha), and compares these to a standard grazed (*SG*) system. Practical implications of these effects are discussed.

## Materials and Methods

### *Trial site*

The experiment was conducted on Massey University's No. 4 dairy farm near Palmerston North, Manawatu, New Zealand (40° 23' 46.79" S; 175° 36' 35.77" E). The trial site was located in a flat landscape (<3% slope) which receives an average annual rainfall of approximately 1000 mm. The site had a mixed pasture sward of predominantly perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*). The trial was established on a mole-pipe drained Tokomaru silt loam soil, which is classified as a Fragic Perch-gley Pallic Soil (Hewitt 1998).

The research area consisted of fourteen plots (~850 m$^2$/plot), each with an isolated mole and pipe drain system. Mole channels, ~ 40 m long, were installed at a depth of 0.45 m with an interval between moles of 2 m. Drainage from the mole channels was intercepted by a perforated pipe drain (0.11 m diameter) installed perpendicular to the moles at a depth of 0.60 m. Further description of the topography and soil properties of the site can be found in Houlbrooke *et al.* (2004).

### *Experimental design*

The trial consisted of two treatments. The standard grazing (*SG*) treatment involved a grazing duration of ~7 hours for day grazings and ~12 hours for night grazings. The other treatment was duration-controlled grazing (*DC*) which involved a grazing duration of ~ 4 hours for both day and night grazings. All plots were grazed on the same day with the same average stocking rate, which was set according to pasture cover (as estimated using a rising-plate pasture height meter). Grazings alternated between 'day' and 'night' regimes to simulate standard farm practice. There were 8-10 grazings per year. The trial site was established during the summer of 2008. Fertiliser applications, drainage dates and cow grazing targets are given by Christensen *et al.*(2012).

### *Estimated pasture dry matter accumulation, herbage elemental analysis and dung deposition*

A rising-plate pasture height meter was used pre- and post-grazing to estimate the quantity of pasture on all plots. These 60 measurements per plot were used to estimate pasture accumulation between grazings, the stocking rate required at each grazing and cow intakes at each grazing. Prior to each grazing from January 2009, a total of 40 hand-plucked samples of herbage were taken from each plot, avoiding herbage directly surrounding dung pats and urine patches. This herbage was oven dried and ground, and underwent basic elemental analysis (Hill Laboratories Ltd, Hamilton, New Zealand).

The dung pats deposited on each plot were counted to give an indication of the total amount of excreta returned to the plots (results not presented here). The difference in average dung deposition between the two treatments was used to estimate the quantity of excreta collected on the feed pad and, therefore, the amount of slurry to be applied to the *DC* treatment, based on N return. Slurry, derived from a feedpad bunker with a weeping wall, was first applied (5 to 10 mm) to the *DC* plots in mid-December 2008 (Christensen *et al.* 2012). This was the only application of slurry in the 2008/09 season. No slurry was applied during the second lactation season, while four dilute applications of slurry were applied to *DC* plots in the third season.

### *Drainage water volume measurements and water analysis*

Drainage water collection methods can be found in Christensen *et al.*(2012). Drainage water samples were filtered through a 0.45 µm filter. The filtered samples were analysed for K using atomic absorption spectroscopy on a GBC Avanta ∑ spectrometer (Walsh 1955).

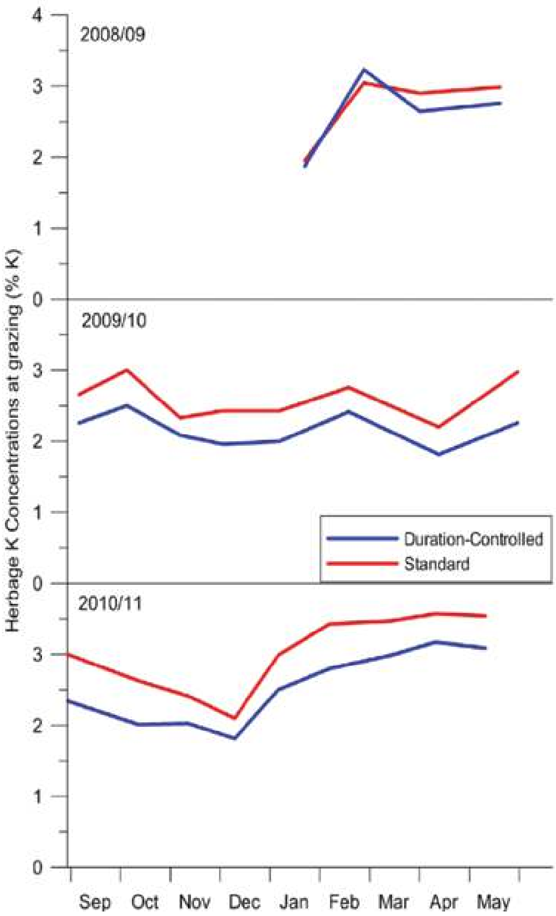
### *Statistical analysis*

Herbage K concentrations at each grazing were analysed using repeated measures in the generalised linear model procedure (PROCGLM), using the statistical package SAS v9.3 (SAS Institute Inc., Cary NC, USA) (SAS 2012).

## Results and Discussion

Herbage K concentrations were similar in both treatments at all grazings in 2008/09 (albeit herbage analysis for this season was only measured from January). In 2009/10, the K concentration in *SG* herbage was higher (P<0.05) than in *DC* herbage in all but the first and third grazings. This trend continued in 2010/11, where herbage K in *SG* plots was greater than *DC* (P<0.05) at all grazings (Figure 1).

In the 2008 drainage season, before treatments were applied, K concentrations in drainage were similar for both grazing treatments (data not shown). The K concentrations in drainage remained relatively constant each drainage season (majority of plots leaching < 4 mg/kg K). This contrasts with the pattern of $NO_3^-$-N concentrations in drainage (Christensen *et al.* 2011), which shows a general trend of being higher (8-20 mg/kg N) at the start of the drainage season and steadily decreases to lower concentrations (<1 mg/kg N) over the drainage season. Low and constant concentrations of K over the drainage season reflect the slow release of exchangeable soil K, whilst the lower concentrations in drainage from *DC* plots than *SG* plots resulted from less urine return. The total amount of K (kg/ha) lost in drainage water varied over the 3 drainage years after treatments were applied. On average, *DC* plots lost 42% less K than *SG* plots via drainage water, with the difference in losses increasing over time (Figure 2).



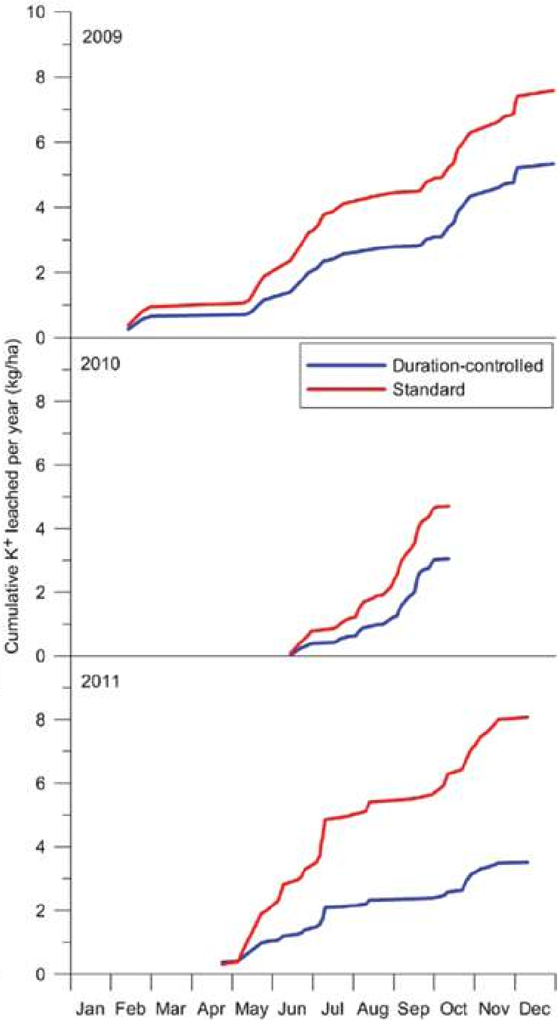




**Figure 1: (Left): Herbage K concentrations at each grazing for each treatment**

**Figure 2: (Right): Cumulative K leached in drainage water over 3 drainage seasons for each treatment**

The widening gap in the quantity of K leached from treatments over time indicates that there was less available (leachable) K in the grazing system on the *DC* plots as time progressed. This trend was also evident in the herbage analyses. In addition, QT K soil test results in May 2010 were on average 4.6 for *SG* plots, while they were only 2.9 for *DC* plots. Soil tests between treatments were similar at the commencement of the study.

In the 2008/09 season, an average of 66 kg K/ha was applied in slurry in the single application to the *DC* plots. In 2010/11, using the same K:N ratios as the first season for feedpad scrapings and estimates for the K concentrations in pond effluent, it is estimated that 72 kg K/ha was applied in slurry. Using figures for the average % K in the pasture herbage and silage fed on the feedpad, and the total feed mass ingested, the K transfer model of Salazar *et al*. (2010) was used to estimate the amounts of K removed from the plots and transferred to other areas of the farm. Based on the model and calculations from first principles, it was estimated that ~183 kg K/ha/yr more K was transferred to the effluent system from the *DC* plots, compared to *SG* plots. The average return of K via slurry to the *DC* plots was 46 kg K/ha/yr, a shortage of ~137 kg K/ha/yr compared with the *SG* plots. This net negative return to *DC* plots was similar to that modelled in OVERSEER® Nutrient Budgeting software; however this software over-predicted the three year average amount of K lost in drainage water from both the *SG* and *DC* plots by ~200% and 100%, respectively. Moreover, the deficit in K returned helps to explain the decline in soil tests on the *DC* plots.

It is clear that the slurry applications from the feedpad bunker did not make up the difference (~183 kg K/ha) in the net K budget between the *SG* and *DC* grazing systems. The feedpad slurry was low in K because the liquid stream, containing urine K, was drained away from feedpad slurry to the dairy effluent storage pond. If cows are stood off paddocks regularly, the applied slurry must contain the urine collected from the feedpad if the available K balance in the soil is to be maintained. If it does not, the practice of standing cows off will create a greater K fertiliser requirement. However, areas that have received storage pond effluent in the past, hence with excessive K soil levels, could benefit from less grazing time (associated with standing off) in combination with the addition of only feedpad slurry.If *DC* grazing is to be implemented at a farm scale, it is important that the K content of all effluent streams is known and is considered in the farm's nutrient management plan.


## Acknowledgements

This study was conducted through the Pastoral 21 Environment Programme (C10X0603) funded by the Foundation for Research, Science and Technology, DairyNZ, Fonterra and Meat & Wool New Zealand. Thank you to the Fertilizer & Lime Research Centre technical team for their assistance with field and laboratory work and to the staff of No. 4 Dairy Farm.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# Environmental limits to land use intensification on New Zealand stony soils

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Stony alluvial soils are an extensive landscape in eastern regions of the North and South Islands of New Zealand. These stony soils vary both in the depth of fines to gravels, and in the texture of the fines in the matrix between gravels. Over the last 40 years these soils have been subject to land use intensification (particularly dairy farming) associated with irrigation development. We hypothesise that in New Zealand, stony soils are the most sensitive soil-landscape for nitrogen and phosphorus leaching, and are the most widespread and vulnerable landscape for leaching of organic contaminants under effluent irrigation. Across the wide range of stony soils, there is limited research on the environmental effects of intensification. This paper presents a methodology to evaluate opportunities and threats to land resources, and identify what knowledge is required for land use and management on stony soils to develop within environmental limits.

# Re-defined soil test–grain yield response relationships can give better fertiliser decisions for P, K and S with wheat, canola and lupin in the cropping systems of Western Australia

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## Abstract

Soil test P, K and S – response relationships for wheat, lupin and canola can be used to make better fertiliser decisions for the cropping systems of Western Australia. A database of N, P, K and S experiments conducted in Western Australia was compiled. Relationships were then defined between soil test value and per cent maximum yield. Data sets were partitioned according to soil properties and sampling depths to improve the accuracy of the fitted regression equations. Sampling of the surface 0-10 cm soil layer provided suitable predictions of P relationships when separated into soil types with different P sorption capacities. In contrast, soil sampling to 30 cm depth improved the relationship for K soil tests for wheat and canola when the impact of soil acidity and potential yield on root distribution was taken into account. In contrast, deeper soil sampling to 30 cm improved the relationship for S for canola but not for wheat where rate of S leaching and potential yield appear to be important. The N soil test relationships for wheat and canola were not significant and critical levels could not be defined.

## Introduction

Phosphorus (P), potassium (K), sulphur (S) and nitrogen (N) fertilisers are key inputs to maintain a profitable cropping industry in Western Australia (WA). However, excessive fertiliser use or inappropriate application practices can lead to nutrient pollution of land, water and air. Adoption of best management practices (BMP) is required to achieve high agronomic effectiveness with low environmental impact. This paper is directed at defining fertiliser nutrient requirements for wheat, canola and lupin as determined from the soil test results and crop grain yield response. This is achieved by soil sampling to various depths, analysis of the soil for nutrient concentrations using standard extraction techniques and defining a relationship between soil test measurement and crop response to applied nutrient, referred to here as the soil test – crop response relationship. These relationships need to be calibrated to account for differences in crop demand, soil types and climatic conditions. Once defined, these relationships can be used to determine whether to apply or to not apply a nutrient and assist with calculating the rate of nutrient required. The aim of this paper is to define the soil test– grain yield response relationships for P, K, S and N with wheat, canola and lupin grown in WA.

## Materials and Methods

### *Database*

A database of 1824 fertiliser experiments conducted by the Department of Agriculture and Food (DAFWA) was developed. All data accepted in the database met rigorous quality assurance criteria. Since minimum tillage practices are currently used in WA, only trials conducted post 1990 were selected to define the soil test – crop response relations. The exception was when the data base was small, for example K soil test – lupin response where all years were included in the analysis. This limited the analysis to 1237 experiments. Soil types were separated into grey sands, coloured (yellow, brown and red) sands, gravels, duplex soils and loams. The data are available on the web as an interactive data base "Making Better Fertiliser Decisions for Cropping Systems in Australia Interrogator" (www.bfdc.com.au).

*Soil test measurements*

Techniques used to measure plant available nutrients in Western Australia are as follows:

- P soil test or bicarbonate extractable P (Colwell 1963),
- K soil test or bicarbonate extractable K (Wong *et al*. 2000),
- S soil test or KCl-40 extractable S (Blair *et al*. 1991),
- N soil test or KCl extractable nitrate and ammonium (Rayment and Lyons 2010).

Other soil measurements undertaken to improve nutrient assessment of the soils include soil pH, total carbon content, clay content and P sorption capacity. Various P sorption measurements have been conducted and include reactive iron, phosphorus retention index (PRI) and phosphate buffer index (PBI).

*Sampling layer*

Soil samples were collected prior to conducting the experiments. The surface sampling layer was to 10 cm depth, except for N experiments where the surface sampling layer was to 15 cm. Additional soil samples were collected down the soil profile for many sites. The most commonly used sub-soil sampling depth was to 30 cm for K, S and some P experiments and to 45 cm for N experiments. Nutrient content of the soil can be expressed in units of mg of nutrient/kg of soil (mg/kg). Alternatively, nutrient content can be expressed in units of kg of nutrient/ha to a specified sampling depth. This value is obtained by multiplying nutrient concentration (mg/kg) by depth of soil layer (mm) and by bulk density divided by 100. As bulk densities were not measured for most experiments, they were assumed to be related to soil texture with sands having bulk density of 1.5 g/cm$^3$ and loams having bulk density of 1.3 g/cm$^3$. The results from gravelly soils were not presented using this approach because gravel content of the soil reduces the nutrient content of soils and unfortunately gravel content was not reported in most experiments. Soil N supply was calculated as amount of N extracted by soil test to depth of 45 cm plus predicted growing season N mineralisation based on levels of soil organic N (Anderson, unpublished data). The root distribution weighting approach of Wong *et al*. (2000) was used to examine the impact of root distribution on availability of soil K to wheat and canola to a depth of 30 cm. The c coefficient in the exponential root distribution with depth equation was initially set to 0.065. However, most wheat and canola experiments examined were affected by surface and/or sub-soil acidity. The c coefficient was hence increased to 0.100 when the 0-10 cm soil layer pH was less than 5.5 and 10-20 cm layer pH was less than 4.5. Also when the wheat grain yield was less than 1.5 t/ha the c coefficient was increased to 0.150.

*Soil test – crop response relationships*

Nutrient rate experiments were conducted to derive crop grain yields (t/ha). Nutrient rates used and the number of rates applied varied widely among the experiments. The database mainly contains multiple nutrient rate treatments which generally produced a yield plateau to provide the most accurate assessment of maximum yield. Percentage of maximum grain yield (PMY) was calculated by dividing the yield obtained for the nil nutrient rate treatment by the maximum yield observed among the nutrient rate treatments. In some experiments, the maximum yield was observed for the nil rate treatment. By definition, for each trial maximum percent yield is 100. This approach allows comparison of experiments on different soil types and under different seasonal conditions that produce differences in maximum crop grain yield.

Soil test values were plotted on the x axis and PMY was plotted on the y axis. A modified Mitscherlich equation, PMY=a-b exp ($-c^d$), where; a, b, and c are coefficients; and d is the soil test value, was fitted. The *a* coefficient was set to equal 100. The *b* coefficient was derived from the fitted equation or estimated so that the predicted critical levels are consistent with the approach used by the BFDC Interrogator (Chris Dyson, personal communication). Critical soil test levels were calculated to correspond to 95% *PMY.* The confidence interval around the critical value was calculated from the standard errors associated in defining the 95% PMY using the statistical package R version 2.15.0 (http://www.r-project.org/ downloaded 17/04/2012).

## Results

The P soil test for crops (wheat, canola and lupin) differed among soil types (Table 1) due to differences in P sorption properties of the soil types. The critical wheat P soil test value for grey sands was 15 mg/kg (confidence interval 12-17 mg/kg) and for other soils 29 mg/kg (confidence interval 28-31 mg/kg). The critical canola P soil test value was defined as 20 mg/kg (confidence interval 17-25 mg/kg) across a wide range of soil types. The critical lupin P soil test for grey sands was 14 mg/kg (confidence interval 11-16 mg/kg) and for yellow sands with PRI=1 was 24 mg/kg (confidence interval 18-30 mg/kg).

When using the 0-10 cm sampling depth (mg/kg), the K soil test - wheat grain yield response relationship was poor, when data was pooled across all soil types (Table 1). In contrast, the K soil test – canola yield response relationship defined a critical value of 57 mg/kg (confidence interval 53-61 mg/kg). The relationship for wheat was improved when soils were separated into different soil types. Grey sands had lower critical values of 32 mg/kg (confidence interval 24-39 mg/kg) compared to yellow sands, gravels and loams, with defined critical value of 59 mg/kg (confidence interval 51-67 mg/kg). Duplex soils had a poor relationship and neither the critical value nor the confidence interval could be defined. The relationship for lupin grown on grey sands defined a critical value of 27 mg/kg (confidence interval 23-30 mg/kg).

When using the 0-30 cm sampling depth (kg/ha), the K soil test - wheat grain yield response relationship was also poor for soil sampled in the 0-30 cm layer (Table 1). In contrast, the K soil test – canola yield response relationship defined a critical value of 55 kg K/ha (confidence interval 35-75 kg/ha). Application of the Wong *et al*. (2000) root distribution approach improved the relationships when the c coefficient was increased to account for reduced root growth due to soil acidity and low yield potential. The weighted critical soil K test for wheat (0-30 cm) was defined as 32 mg/kg (confidence interval 29-35 mg/kg) and for canola was defined as 40 mg/kg (confidence interval 36-44 mg/kg).

**Table 1: Critical values, confidence intervals and regression coefficients for soil test – crop response relationships by nutrient, crop, soil type and soil depth.**

| Nutrient | Crop | Soil depth | Soil type | No of experiments | Critical values[G] | Confidence intervals[H] | $r^2$ |
|---|---|---|---|---|---|---|---|
| P | Wheat | 0-10 cm[A] | Grey sands | 22 | 15 | 12-17 | 0.63 |
| | | 0-10 cm[A] | Other soils[D] | 67 | 29 | 28-31 | 0.86 |
| | Canola | 0-10 cm[A] | All | 31 | 20 | 17-25 | 0.72 |
| | Lupin | 0-10 cm[A] | Grey sands | 22 | 14 | 11-16 | 0.37 |
| | | 0-10 cm[A] | Yellow sands[E] | 46 | 24 | 18-30 | 0.89 |
| K | Wheat | 0-10 cm[A] | All | 139 | na | na | 0.05 |
| | | 0-10 cm[A] | Grey sands | 13 | 32 | 24-39 | 0.42 |
| | | 0-10 cm[A] | Other soils[F] | 76 | 59 | 51-67 | 0.63 |
| | | 0-10 cm[A] | Duplex | 48 | na | na | 0.08 |
| | | 0-30 cm[B] | All | 62 | na | na | 0.02 |
| | | 0-30 cm[B] | Yellow sands, Duplex | 59 | na | na | 0.33 |
| | | 0-30 cm[C] | Yellow sands, Duplex | 33 | 32 | 29-35 | 0.57 |
| | Canola | 0-10 cm[A] | All | 182 | 52 | 48-56 | 0.67 |
| | | 0-30 cm[B] | All | 91 | 55 | 35-75 | 0.51 |
| | | 0-30 cm[C] | All | 182 | 40 | 36-44 | 0.72 |
| | Lupin | 0-10 cm[A] | Grey sands | 22 | 27 | 23-30 | 0.82 |
| S | Wheat | 0-10 cm[A] | All | 61 | na | na | 0.00 |
| | Wheat | 0-30 cm[B] | All | 55 | na | na | 0.00 |
| | Canola | 0-10 cm[A] | All | 126 | na | na | 0.15 |
| | Canola | 0-30 cm[B] | All | 126 | 43 | 39-46 | 0.57 |

[A] with units in mg/kg [B] with units in kg/ha, [C] weighted for root distribution with units in mg/kg, [D]other soils - yellow, red and brown sands, loams, clays and duplex soils, [E]yellow sands with PRI=1, [F]yellow sands, gravels and loams, [G]soil test value (mg/kg) at 95% of predicted maximum grain yield, [H]95% chance that this range covers the critical soil test value, na not available.

The soil S test - grain yield response relationship was poor for both wheat and canola and critical levels could not be defined when the sampling depth was 0-10 cm. Better fits for the soil S test values for canola were obtained by summing the extractable soil S content to 30 cm depth. The critical level for canola was 38 kg S/ha (confidence interval 35-40 kg/ha). There are only 7 lupin fertiliser experiments in the database with only one responsive site which had an extractable S level of 3.7 mg/kg (data not presented).

The inorganic N soil tests using both the 0-10 and 0-45 cm sampling layers– for both wheat and canola grain yield response relationships - were poor (data not presented). Soil N supply or soil profile N plus predicted N growing season mineralisation better predicted wheat grain yield response when sites were separated by rainfall zones and soil types. However, the regression coefficients for soil N – crop yield relationships were all less than 0.25. Nevertheless, some sites with N supply of 102-106 kg/ha were able to produce 4.0 t wheat/ha within the 275 to 375 mm rainfall zone.

## Discussion

Phosphorus sorption in known to have a large impact on the availability of soil P to crops particularly when P is extracted using the sodium bicarbonate solution (Helyar and Spencer 1977). Within the wheat and canola data base, there was limited availability of P sorption data and only PRI was recorded in the lupin data base. As a result, soil type was used as a surrogate for P sorption to separate of the P data. This approach resulted in wheat and lupins grown on grey sands having a lower critical value of 14-15 mg/kg compared to higher critical levels of 29 mg/kg for wheat on grown on all other soils. A critical value of 24 mg/kg was determined for lupins grown on yellow sands with PRI=1 (Table 1). For soils with PRI values greater than 1, P sorption reduced soil P availability to lupins but it was not possible to define a soil test – crop response relationship (data not presented). Canola appeared to be less sensitive to soil type, with the soil test values correlated to canola yield response across a wide range of soil types, giving a critical value of 20 mg/kg (confidence interval 17-25 mg/kg). The P soil test increases with higher fertiliser applications and this differs in relation to PBI, production history and the rate of P input (Weaver and Wong 2011). The impact of P fertilisation history on defined P soil test – crop response relationships needs to be investigated.

Sulfur soil test better predicted canola grain yield responses when using the soil sampling depth of 0-30 cm compared to the sampling depth of 0-10 cm. The improvement compared to the 0-10 cm soil sampling layer is attributed to the utilisation of sub-soil S. For wheat, sampling to a depth of 30 cm did not improve the relationship as rates of S leaching and potential yield appeared to have an impact on the relationship.

Potassium soil test better predicted canola grain yield responses when using the soil sampling depth of 0-30 cm compared to the sampling depth of 0-10 cm. However, the approach required root distribution as affected by soil acidity and potential yield to be taken into account.

Nitrogen soil testing both for the 0-15 cm or 0-45 cm soil layers, even when the contribution of growing season mineralisation was estimated, had limited predictive capacity for crop yield response, presumably because of the large impact of nitrate leaching on soil inorganic N availability in WA (Anderson *et al*. 1998).

## Acknowledgments

The project was jointly funded by Department of Agriculture and Food Western Australia, Murdoch University and Grains Research and Development Corporation (UMU00030). The project collaborated closely with the national Better Fertiliser Decision Crop project managed by DPI NSW.

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# Thursday 6 December

## SOIL FERTILITY AND SOIL CONTAMINANTS (NUTRIENT MANAGEMENT)

*Presented Posters*

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# Integrated soil fertility management: Residue quality effects on soil N stabilization

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Soil organic matter is important to improve and sustain soil fertility in tropical agroecosystems. The combined use of organic residue and fertilizer inputs is advocated for its positive effects on short-term nutrient supply, but the effect of the integrated use on soil organic N stabilization is unknown. Furthermore, this effect may be controlled by residue quality. We conducted a 545-d soil incubation experiment with three different residue qualities (*Zea mays*, *Calliandra calothyrsus*, and *Tithonia diversifolia*) with and without the addition of urea fertilizer in a Kenyan Humic Nitisol. The inputs were enriched in $^{15}N$ in a mirror-labelling design to trace the fate of residue- and fertilizer-derived N in soil aggregate fractions. We hypothesized that combining residue and fertilizer inputs would enhance the stabilization of N relative to either input alone and this interactive effect would increase with decreasing residue quality. While combining low quality *Zea mays* residue and fertilizer decreased the amount of residue-N stabilized within soil aggregates (10% less) it increased the stabilization of fertilizer-N (23% more), indicating an interactive increase in the stabilization of new N. The higher quality residues stabilized a lower amount of fertilizer-N and did not result in a significant interactive effect when combined with fertilizer. Our results indicate that residue quality and N input type control the stabilization of N in soil aggregates. Interactive effects from combining low quality residue with fertilizer inputs can increase the short-term stabilization of N, which has the potential to improve soil fertility.

# Comparative effects of urea plus gypsum and urea plus ammonium sulphate on growth, yield and nutrition in canola cropping system in calcarosol of south-eastern Australia

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The importance of nitrogen (N) and sulphur (S) in canola cropping systems is well established. However, in the past, urea and gypsum were the commonly used strategies for N and S nutrition. We hypothesised that the use of ammonium sulphate (AS) along with urea may enhance both N and S efficiency in a calcarosol growing canola. We conducted a field experiment to investigate the effects of combining urea and AS on canola growth, yield and nutrient uptake in a calcarosol at Walpeup in south eastern Australia. Irrespective of sources, nitrogen significantly ($p \leq 0.05$) increased above ground biomass at flowering stage and grain maturity stage. Similarly, this was also reflected on a range of nutrient uptake. However, nutrient uptake (N,P,K, S, Ca, Mg and micro nutrients) at flowering stage also higher ($p \geq 0.05$) in the urea plus AS treatment compared to that observed in the urea plus gypsum treatments. Although a response to S was not seen at flowering stage, a significant S response occurred at grain maturity stage in urea plus AS compared to urea plus gypsum. Urea plus AS significantly ($p \leq 0.05$) increased agronomic N and S efficiency by 3.65% and 35.58% respectively, compared to urea plus gypsum. We hypothesise that the higher efficiency may be associated with lower ammonia volatilization and high acidification of the root zone in the alkaline calcareous soil which enhanced nutrient uptake. Further results on N and S uptake at maturity and 15N studies of plant and soil under field micro-plots will be used to test our hypothesis.

# Relative contribution of nitrogen sources to irrigated rice under contrasting rotations, soil types and N inputs

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In Uruguay rice rotates with perennial pastures and country productivity has reached 8 Mg ha$^{-1}$ with relatively low N fertilization inputs (70 kg N ha$^{-1}$). Our objective was to determine the relative contribution of different N sources (soil, fertilizer, biological fixation) in rice seeded in contrasting soil types, rotations systems and N fertilization rates. Twelve rice trials were located in three soils types and two contrasting rice predecessor (legume vs. no-legume pastures) during two growing seasons. Each experiment evaluated two N rates (0 kg ha$^{-1}$ and 64 kg ha$^{-1}$) split at planting, tillering and panicle initiation in a complete randomized block design with four replications. Determinations included $\delta^{15}N$, total N, organic C and $\delta^{13}C$ in soils and plants, rice biomass and yield. High variability in $\delta^{15}N$ was observed among soils; the value was correlated with soil sand content, r=0.90 and 0.71 in year one and two, respectively. In the first season, rice yield was correlated with soil N mineralization potential (r= 0.85) and productivity with N addition was 8% higher than without N (9.81 Mg ha$^{-1}$). In the second season, rice yield with N was 15% higher than without N (7.86 Mg ha$^{-1}$). Plant $\delta^{15}N$ suggested that N biological contribution from pastures or during the rice growing cycle was not significant to crop nutrition. Grain and plant $\delta^{15}N$ in all trials were very similar to soil $\delta^{15}N$ and no major differences in grain $\delta^{15}N$ and plant $\delta^{15}N$ was observed between N treatments, indicating that fertilizer contribution to grain N was very poor. Results suggested that soil is a major source of N in Uruguayan rice systems and that that isotopic fractionating process were not important, probably due to high crop demand of mineralizated N from soil.

**Keywords:** $^{15}N$ natural abundance, isotopic techniques, N cycle, flooded soils.

# Nitrogen supplying capacity assessment in temperate pasture soils: A guide to better prediction of fertiliser N response

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Nitrogen (N) is one of the key nutrients applied as fertiliser to meet the production goals of intensively grazed temperate pasture systems. However, injudicious use of N on pastoral soils decreases the N efficiency of the production system, with increased risk of loss of N to water and to air. There are currently no field-calibrated N tests in use for pasture soils to predict N responses, therefore, farmers tend to apply a uniform rate of N across the farm without realising that areas of their farms could have significant variation in N response. This "hit and miss" approach by farmers can result in a significant waste of money and risk of environmental damage. AgResearch has conducted a number of laboratory and field evaluations and tested various pools of soil N for use as better predictors of N fertiliser response by pastures.

Total N in New Zealand pasture soils at 0-7.5 cm depth varies between 0.3 and 1.5% (w/w). Approximately 97-99% of the total N in soils is present in organic forms as part of the soil organic matter. Approximately 8 to 10% of the organic N in these soils is present in an easily mineralisable form ("extractable organic N"). The rate of release of N from the easily mineralisable pools is dependent on factors that affect microbial activity in soils such as temperature, moisture, soil nutrients and C:N ratio. A number of replicated plot trials across the country have been conducted over the last 10-15 years where various pools of soil N (2 *M* KCl extractable mineral N, hot-water extractable total N, total N, and microbial N) were evaluated as indicators to predict yield responses to added fertiliser N. The most promising indicator of dry matter response was obtained by measuring the total N pool in soils. There was an inverse relationship between total N in soils and N fertiliser response (extra kg DM produced per kg N applied). Generally, the greater the total N, the lower the response per unit N applied. Even so, application of fertiliser N at higher levels of soil total N can be economically profitable. Since total N and organic matter content in pasture are highly correlated, greater total N may also mean better soil physical and biological condition for plant growth e.g. better water retention in soil with higher total N contents.

The use of soil total N as an indicator of DM response offers the potential to increase N fertiliser use efficiency by better targeting of fertiliser to more responsive paddocks; especially where there is spatial variation in soil total N across a farm - or where N leaching is capped through environmental regulation and there is a benefit from achieving highest productivity for a constrained level of N fertiliser input.

**Key words:** yield response, extractable N, total N, microbial N

# Monitoring soil chemical properties in grazed dairy systems

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Effective monitoring of grazed dairy soils must integrate spatial variability at many scales. Animal management practices can lead to nutrient redistribution between fields within a farm, while animal behaviour can result in within-field variability. Meta-analysis of soil data collected from 127 grazed dairy farms in three dairy regions of south eastern Australia was undertaken, to (i) identify properties that can be used for on-going monitoring of soil condition, (ii) quantify spatial and temporal variability in soil properties and (iii) develop sampling strategies that account for variability at different scales. Chemical properties that predominantly characterise the fertility status of soils (e.g., pH, extractable soil P and K) were analysed in the majority (99%) of soil samples, while measures of the C cycling function of soils were only analysed in 6% (Total C) and 13% (soil organic C) of samples. Significant differences were observed in median soil pH between the three dairy regions, with soils from the northern Victorian region having the highest pH. Temporal trend analysis in the data collected between 1995 and 2008 in all three regions indicated that median soil pH has not declined, while median extractable soil P and K have increased. Variance component analysis of soils collected in the south west Victorian region showed that the variance for soil pH (in CaCl2) was lowest between soil types and was highest between farms, while Total N was highest between soil types. The implications of these results for sampling strategies for grazed dairy soils are discussed.

# Rain with a regular 'pulse': Explaining soil water and nitrogen accumulation under summer fallow conditions

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The use of a fallow period to conserve rainfall as stored soil water and build up soil mineral nitrogen is a well-established practice in many dryland agricultural systems. However, the magnitude of soil water and nitrogen accumulation has proven highly variable and difficult to predict with rainfall amount and distribution being key factors of influence. Inspired by similarities in the response of primary productivity to rainfall variability in the field of arid land ecology, we recently adopted their 'pulse paradigm' to explain fallow management impacts on soil water accumulation. Rainfall variability is viewed in terms of different sequences of rainfall events, which create pulses of soil water. The depth and longevity of these pulses are a function of rainfall amount and frequency as well as evaporative demand. When pulses overlap they may reach beyond the evaporation zone and lead to more permanent storage of soil water. Closely spaced rainfall events and/or low evaporative demand create conditions favouring overlap and were found to increase soil water accumulation. Here we build on this concept to study the impact of rainfall variability on soil nitrogen mineralisation under fallow conditions. Analysing the results of simulations using historical weather data from Wagga Wagga, NSW we find that during summer frequent rewetting is critical for mineralisation to occur and that considering pulses of soil moisture can be a useful way to explain the amount of mineralised nitrogen at sowing. In other words, rain with a regular pulse is the key to nitrogen accumulation under summer fallow conditions.

# Thursday 6 December

## THE PHYSICAL SOIL AND SOIL WATER

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# Catchment-wide modelling of land use impacts on the Ruataniwha Plains.

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The Hawkes Bay Regional Council (HBRC) is currently investigating a water storage project for the Ruataniwha Plains. They are seeking information to define the potential volumes of water that are needed, now and in the future, to irrigate a range of agricultural and horticultural enterprises. In addition, the Council are also seeking information to help assess the potential effects of land use change on surface and groundwater quality as a result of irrigating from stored water. In this paper we will discuss a modelling approach that is being used to simulate irrigation demand plus nitrogen (N) and phosphorous (P) loads from a range of land use activities. The modelling is being carried out in two stages. Firstly, at the enterprise scale, we are using Plant and Food Research's SPASMO model (Soil Plant Atmosphere System Model) to simulate the daily water and nutrient balance for a range of land uses, soil types and microclimates. Then we are using AgResearch's GIS landscape modelling tools to aggregate the water and nutrient balances across a number of sub-catchments (irrigation zones). The task of the modelling is to assess the impacts of land use intensification on the quality of the ground and surface water resource. Some preliminary results will be presented for selected farming enterprises.

# Methodological challenges in agricultural product water footprinting: A case study of New Zealand wine

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Globally, the food industries are confronting serious challenges associated with freshwater use. The water footprint (WF) has been proposed as an indicator of the impacts of production systems on freshwater resources. A number of methods for water footprinting have been proposed. These methods must capture the inherent variability of agricultural production systems and ensure that the local hydrology is rationally accounted for. The WF of wine was assessed for the Marlborough and Gisborne growing regions. Central to our analysis was the use of the hydrological water-balance which was simulated for 29 soil types spread across 19 different climatic locations. Results indicate that the major impact occurs in the grape growing stage. The hydrological method shows large variation in impacts since the local hydrology is driven by the variability in rainfall and the wide differences in soil properties. On average, in both regions, groundwater resources are recharged under viticulture which is reflected in a negative blue-water footprint. The grey-water footprint is calculated as the water needed to dilute the leached nitrate to the required drinking-water standard. This was higher for the Gisborne region (248 L/kg grapes) than for Marlborough (42L/kg), due to the higher rainfall plus the greater nitrogen mineralization in Gisborne soils. We conclude that the WF based on hydrological method provides meaningful information to a number of end-users including growers, regulators and marketing personnel about the impact of production on local water resources. It also enables meaningful comparison between products from different locations as the WF is assessed within the context of local hydrology. These results are also compared with two other WF methods.

# Development and initial testing of a surface runoff simulation model based on Cellular Automata

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The long-term stability of rehabilitated landforms is often threatened by surface runoff, which may cause severe soil erosion, revegetation failure, waterways pollution and contaminants transport to surrounding areas. Therefore, a good understanding and prediction of surface runoff is essential for sustainable rehabilitation practice. Most existing runoff models, however, are limited by complex conditions and their failure to describe the dynamic hydrologic behaviours. In this study, an innovative runoff prediction model has been developed based on Cellular Automata (CA) in C++. CA is a discrete dynamic system composed of a set of cells, and its global behaviour is determined by evolution of states of all the cells. Thus CA is very effective in simulating complex phenomenon from local to global according to simple transition rules. For this CA runoff model, the cell state (height) consists of both altitude and water depth, which is determined by rainfall, interception, evapotranspiration and infiltration, where five widely used infiltration equations have been integrated. The water flows between these cells could be simulated based on minimisation-of-differences algorithm and multiple-flow-direction algorithm. This developed model has been initially tested using rainfall simulation experiments on small runoff plots with different infiltration capacities and slope gradients. Results showed that the simulated runoff curves were very close to the observed data, and total runoff volumes could also be well predicted, with the average accuracy above 95%, indicating that this model was effective in this study. Nevertheless, further tests are required for application at larger scales and more complex situations.

# Attenuating the differential water capacity for different soils and plants

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The differential water capacity of a soil is integrated between various limits to estimate the maximum so-called 'plant available water'. We know, of course, that much of this water is in fact NOT available to plants because adverse soil physical conditions limit normal physiological growth of plants (e.g. salinity, poor aeration, high soil strength, excessively large or small hydraulic conductivity, etc). To quantify the magnitude of these adverse soil conditions on water availability, we can attenuate the differential water capacity for different soil properties prior to integration. The problem is that no universal attenuation function exists for a given limiting soil factor applicable to all plants under all conditions. There are, however, some general functions that can be adapted for different plants, soils and environmental conditions. For example, an attenuating function for high soil resistance to root penetration, $\varpi_{SR}(h)$ takes the form:

$$\varpi_{SR}(h) = \left( \frac{\left( \frac{SR(h_f)}{SR(h_i)} \right)}{\left( \frac{SR(h_f)}{SR(h)} \right)} \right)^{\beta} .$$

*SR(h)* is the penetrometer resistance as a function of the matric head (modelled from soil measurements), $h_i$ is the soil matric head at which the penetrometer resistance starts restricting root penetration (e.g. 0.5 MPa, but can be varied for different plants), $h_f$ is the soil matric head at which the penetrometer resistance stops root growth altogether (e.g. 2.5 MPa, but can be varied for different plants).

The parameter β accounts for the ability of different plants to extract water from high-strength soils by relying on the soil hydraulic conductivity or by reducing water demand. We present other such functions for the effects of soil aeration, salinity and hydraulic conductivity.

# The effect of sheep and cattle treading on soil water holding capacity and predicted surface water run-off

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With the recent introduction of irrigation schemes into the North Otago Rolling Downlands (NORD), land use has intensified including a gradual shift from dry land sheep farming to dairying. Treading damage under livestock grazing has a detrimental effect on these soils by causing compaction, a process that reduces soil pore size and/or overall porosity.

At a trial site in the NORD, soil compaction in response to land-use was assessed by monitoring pore size distribution under dairy cattle and sheep grazing pasture. For each animal class, treatments were divided into irrigated and dryland systems. Following 6 months of rotational grazing by sheep or cattle, a significant reduction in porosity was evident in the irrigated treatments; however this effect was even greater where cows had been grazed. In the case of cattle grazing, a shift in pore size distribution towards microporosity was also observed. Reduction in total porosity lowered air space when soils were at field capacity while a decline in pore size reduced plant available water at lower matric potential. This indicates that under wet conditions, compacted soils will reach a point of saturation more rapidly than non-compacted soils yet under dry conditions drainage will be slower.

At locations along the hill slope where soil depth is shallow, such as hill crests and upper slopes, changes in water holding capacity are predicted to increase the incidence of saturation excess run-off during rainfall events thereby increasing the likelihood of nutrient loss from land to surface waters. During dry summers, a decline in plant available water may in turn limit plant growth. This change in soil water content and matric potential is likely to limit the rate of natural soil structural recovery thereby increasing the risk of repeated soil damage with on-going grazing, particularly under cattle, where the initial damage is more severe.

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## SOIL CARBON

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# Can subtropical perennial pastures effectively sequester soil carbon?

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Summer active perennial grasses are receiving growing interest within the Australian agricultural community as an alternative to winter active annual grass based pasture systems due to their potential agronomic and soil carbon benefits. This study has focused on examining changes in soil organic carbon (SOC) where C4 subtropical perennial grasses, kikuyu (*Pennisetum clandestinum*) or a mix of Gatton panic (*Panicum maximum*) and Rhodes (*Chloris gayana*) grasses, have been sown into formerly C3 annual grass dominated pastures. In total, 46 paired perennial and annual pastures were sampled in four agricultural regions to 30 cm depth and analysed for total SOC, stable carbon isotope composition ($\delta^{13}C$), and the distribution of SOC and $\delta^{13}C$ into fractions of soil carbon. Of the two main perennial grass types included in this study, only kikuyu was found to be effective at changing SOC levels with a median sequestration rate (defined as perennial SOC – paired annual SOC divided by number of years since vegetation shift) of 0.5 Mg C $ha^{-1}$ $yr^{-1}$. Within pastures planted to kikuyu, the greatest relative gains were found in sandy soils with low initial SOC levels. Importantly, at the sites with the greatest SOC gains, the carbon fraction and isotope data indicated that almost all of the new SOC has accumulated in less protected carbon pools suggesting that if these sites were to revert to a lower C input system, much of this new SOC would be rapidly lost.

# Digital soil mapping of available water content using proximal and remotely sensed data and a hierarchical spatial regression model

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Two thirds of all irrigated agriculture in Australia is undertaken within the Murray-Darling Basin. However climate change predictions for this region suggest rainfall will decrease. In addition, environmental concerns and new industries are competing for water resources. In order to maintain profitability, more will need to be done by irrigators with less water. In this regard, irrigators need to be aware of the spatial distribution of the available water content (AWC) in the root-zone (i.e. 0.0-0.90 m). Owing to the expense of traditional soil survey methods, digital soil mapping techniques are being used with increasing frequency to map soil properties. This includes, soil properties related to AWC such as clay content and mineralogy. Here we create a digital soil map of the AWC at the district scale. This is achieved by determining AWC by the difference between the laboratory measured permanent wilting point (PWP) and field capacity (FC) using a pressure plate apparatus. The PWP and FC is coupled with remote (i.e. gamma-ray spectrometry) and proximal (i.e. EM38 and EM34) data and two trend surface parameters using a multiple linear regression model (e.g. stepwise). This information is used as the basis of developing a hierarchical spatial regression (HSR) model to predict AWC in the irrigation areas of Warren and Trangie, in the lower Macquarie valley. The reliability of the models were compared using prediction precision (RMSE – root mean square error) and bias (ME–mean error). It was found that using EM38-v, EM34-10, eU, and eTh provided the best results ($r^2$=0.55). The DSM maps are consistent with the known pedoderms and soil types and provide a basis for irrigation management.

## Sensors and sequences for soil biological function: Embracing variability from microbes and aggregates to landscapes

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Understanding the relationships between soil microbial community structure and soil function remains a major challenge: "*who* is doing *what*, and *when*? Furthermore, soil properties can vary significantly at different spatial and temporal scales, from millimetres to landscapes and from within a day to between years. This variability has major implications for the assessment of rates of particular functions, such as carbon sequestration, and for the prediction of the impacts of land use changes and land degradation. To address this issue, we are using an inter-disciplinary approach, where real-time monitoring of the soil environment will be related to microbial communities, aggregates, and biogeochemistry dynamics. Basic information on environmental spatial and temporal variability will be derived from a network of environmental sensors deployed across three paired pasture and native woodland sites. The sensor data will be integrated with regular sampling for the characterization of microbial communities and expressed genes, aggregate dynamics, and soil biogeochemistry using the latest techniques in molecular microbial ecology, environmental genomics, and compound specific isotopic analyses to directly link specific soil microorganisms and functional genes with carbon and nutrient dynamics and trace gas emissions. The role of invertebrates in soil ecosystem function will also be assessed, and further studies will be carried out in mesocosm experiments to characterize specific soil processes in relation to these ecosystem engineers. Here we highlight the sensor network, molecular and isotope techniques and soil aggregate approaches that will be used to link the active soil microbes to selected soil functions across spatial and temporal scales.

# Evaluation of an accurate sampling methodology for determining soil organic carbon in clay delved sandy texture-contrast soils

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Soils are identified as a major sink for carbon and increasing soil carbon stocks may help offset greenhouse gas emissions. This is recognised within National Carbon programs such as the Australian Carbon Farming Initiative which will provide an opportunity for landholders to participate in carbon trading schemes.

Many studies have reported a positive correlation between clay content and soil organic carbon (SOC) concentration. Therefore adding clay to sandy soils may increase SOC concentrations. In sandy texture-contrast soils, this could be done by clay delving, which incorporates subsoil clay into the surface soil and is quite distinct from clay spreading. Clay delving mixes the sand and clay layers to approximately 60cm along the delve line but in the areas between the delve line, mixing only occurs to 10-15cm. Increased SOC concentrations after clay addition could be due to two factors: (i) increased plant growth and therefore C input, and (ii) decreased decomposition due to protection of SOC by clay.

Carbon trading schemes require accurate determination of carbon stocks and a national soil sampling methodology is currently being prepared. However, the highly heterogeneous environment created by clay delving makes sampling these soils with confidence problematic and a modified methodology may be required.

A number of sampling strategies were evaluated comparing clay distribution and organic carbon stocks from a delved (on delve and between delve lines) and an undelved site. Clay distribution, organic carbon and bulk density varied considerably on and off the delve line. A new sampling methodology for the delved sites was developed.

# Optimising the use of mid-infrared spectroscopy for predicting soil carbon contents

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The Carbon Farming Initiative legislation provides Australian landowners with the opportunity to earn and sell non-Kyoto Australian Carbon Credit Units for storing and maintaining additional carbon in their soils. Development of cost-effective and rapid soil carbon measurement methodologies is required to allow landowners to quantify and monitor soil carbon stocks. The Soil Carbon Research Program has collected and analysed in excess of 20,000 soil samples collected throughout Australia's agricultural lands. The total, organic and inorganic carbon content of all of these samples has been measured using automated dry combustion carbon analysers. Diffuse reflectance mid-infrared (MIR) spectra were also acquired for these soil samples and used in a partial least squares (PLS) statistical analysis to assess the ability of the MIR/PLS analyses to quantify total, organic and inorganic carbon and how this capability varies across different soils. With adequate calibration, the MIR/PLS analysis successfully predicted the content of soil organic and inorganic carbon and prediction accuracy could be optimised through the derivation of soil type specific prediction algorithms. This capability will reduce the analytical time and cost associated with the determination of soil organic carbon contents, particularly for soils containing carbonates where traditional laboratory methods require application of an acid pretreatment to remove inorganic carbon prior to dry combustion analyses.

# Dynamics of inorganic and organic carbon in acidic limed soil

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Lime is a common amendment to overcome soil acidification in agricultural production systems, however, its impact on inorganic and organic soil carbon dynamics is largely unknown. In a glasshouse experiment, we monitored rhizosphere effects on lime dissolution and $CO_2$ effluxes in an acidic Kandosol. The experiment consisted of four replicates of treatments viz: soil only (control), soil + lime, soil + wheat and soil + lime + wheat. We measured $CO_2$-C effluxes on 7, 43 and 98 days after planting (DAP) and leachate was collected on 56 and 101 DAP. The soil $CO_2$-C efflux increased significantly with lime addition on 7 and 43 DAP compared to control. On 43 DAP, the largest increase in $CO_2$-C effluxes was observed in lime + wheat treatment. However, on 98 DAP similar $CO_2$-C effluxes were observed from wheat and lime + wheat treatments, which suggest that most of the lime has dissolved in the lime + wheat treatment. Both dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC) concentrations in the leachate increased significantly with lime and in the presence of wheat only (cf control). In contrast to DOC, there was an increase in DIC concentration in the soil solution from lime + wheat columns on 101 DAP (significant wheat*lime interaction), thus, accentuating the pronounced role of wheat roots. The data show that in limed acidic soil, lime is successively utilized in the presence of plants and plant mediated dissolution of lime increased the concentration of DIC in the soil leachate.

**Key Words**

Liming; rhizosphere; carbon effluxes; dissolved inorganic carbon; organic carbon; acid soils.

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## NEW TECHNIQUES IN SOIL SPATIAL ANALYSIS

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# High-resolution digital soil assessment – facilitated by proximal soil sensing

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There is a need for high-resolution spatial ( <20 m) soil information. This demand usually occurs in high-value operations where soil is an integral component such as contaminated site assessment and remediation, precision horticulture and agriculture, and sites of special scientific or cultural interest, e.g., archaeological sites and classical field experiments. The methodology for obtaining the data for such assessments is probably best obtained by proximal soil sensing (Viscarra Rossel & McBratney, 1998) and discussed in detail in Viscarra Rossel et al. (2010). High-resolution digital soil assessment is described and compared and contrasted with medium-resolution (10 – 250 m) digital soil assessments.

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# Digital mapping of a soil drainage index in the Lower Hunter Valley, NSW.

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Soil colour is often used as a general purpose indicator of internal soil drainage. In this digital soil mapping study we take advantage of a rich soil point dataset from the Lower Hunter Valley (LHV), NSW to derive and map a continuous index of internal soil drainage based on the primary soil colour of the master or in most cases the B2 horizons. From built up knowledge of the soils in the LHV, the sequence of well draining → imperfectly draining → poorly draining soils closely follows the colour sequence of red → brown → yellow → grey → black for the master soil horizons. It is with this knowledge that an empirical drainage index was derived. First, for each soil profile in the database we extracted the wet Munsell colour information for the master horizon which was then converted to the CIELAB colour space. The L-a-b colour coordinates were then spatially interpolated across the mapping domain using Cubist models, where the explanatory covariates included various primary and secondary terrain information derived from a digital elevation model. For each pixel, we then determined the Euclidean distance to centroid colour coordinates, which were derived from the soil database, to each of the colours in our knowledge driven soil-colour/drainage sequence. Fuzzy memberships to each centroid colour were then derived from which a continuous drainage index value was calculated which ranged from 5 (well draining) to 1 (poorly draining). Prediction of the L-a-b colour coordinates was acceptable based on an external validation of randomly withheld data points. This also implies acceptable prediction of soil drainage. The resulting map of soil drainage is intuitively acceptable given knowledge of soil variation in the Hunter Valley and the prevalence of less well-draining soils in areas lower in the landscape where water tends to accumulate.

# Spatial disaggregation of soil map units and continuous soil property mapping

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There is an ever-increasing need for continuous soil property maps for modelling purposes. Unfortunately it is difficult to derive such information from traditional choropleth soil maps. In such maps, polygon map units may contain a number of soil classes whose spatial distribution is essentially unknown. Spatial disaggregation techniques attempt to reduce the uncertainty around soil class distribution by mapping the soil classes individually. Our spatial disaggregation algorithm attempts to achieve this level of mapping via an iterative procedure that builds decision trees to predict the soil class distribution across the landscape. For each iteration, a new decision tree is built; the calibration data are randomly-sampled points within each map unit that are allocated a soil class based on the multinomial distribution of soil classes within the map unit. After *n* iterations, the probability of occurrence of each soil class at each cell of a raster map grid across the landscape is determined. Continuous soil property maps can then be created using the probabilities as weights in a weighted-mean soil property calculation at each cell of the map grid. We illustrate the use of the algorithm with a case study in the Burdekin catchment in Queensland.

# Field-scale soil spatial variability and implications for sampling

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There is little quantitative information on field-scale spatial variation of soil properties in Australia. However the nature and extent of that variation dictates appropriate sampling and monitoring design. Soil characterisation is usually predicated on: (i) a soil unit or individual being representative of a land mapping unit (e.g. soil type), and (ii) the variation within the soil individual being less than the variation within the land mapping unit. Here we quantify the horizontal field-scale variation in soil properties, examine the nature of the variation, and explore the consequences for sampling and monitoring design. An intensive field sampling was conducted within an area of one hectare from near Temora, New South Wales, Australia. The 100 m ×100 m study area was subdivided into 16 square cells (25 m × 25 m). Samples were collected from the top of the B horizon following an approximately exponential sample spacing with 7 sampling points along a linear transect of 20 m. The transects were laid in different directions. A number of soil physical, hydrological and chemical properties were measured following standard laboratory procedures and their spatial variability within the field was examined using geostatistical analysis. Relative variance in soil properties at different scales was examined. Consequences for sampling design are discussed, including for the monitoring of soil condition.

# Application of on-the-go radiometric mapping for investigating processes of soil genesis in the Hunter Valley, NSW, Australia

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Proximal soil sensing techniques have been used to map spatial variations of soil properties and therefore to make inferences about soil quality in agricultural landscapes. These techniques offer the advantage of surveying sites at a high resolution reasonably quickly at low costs and minimal landscape disturbance. Recently, these on-the-go soil sensing instruments, i.e. passive gamma-ray spectrometers, have also been applied to explore patterns of soil erosion and soil deposition in the landscape through estimating the distribution of artificially produced radionuclides ($^{137}Cs$) in the soil. $^{137}Cs$ has been used widely as environmental tracer to estimate soil erosion in agricultural landscapes. Here, we conducted a soil survey at two catchments in Pokolbin, NSW, to investigate short-term processes of soil genesis in this region with a long history of wine production. We employed a vehicle-borne gamma-ray spectrometer to map at a high resolution the occurrence of naturally ($^{40}K$, $^{238}U$-series, $^{232}Th$-series) and artificially ($^{137}Cs$) produced radioactive isotopes in the soil. Subsequently, soil samples along toposequences were taken to determine the concentration of $^{137}Cs$ and $^{40}K$ in the laboratory, to validate the produced radiometric maps and therefore to better assess the derived soil erosion and soil deposition patterns. Furthermore, the landscape evolution model SIBERIA was used to predict soil erosion rates for the study sites using a high resolution digital elevation model, to see how well model predictions correlate with real-world data. Overall, these observations will also give us a clear picture on the function and response of managed soil systems to a changing environment.

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# Finding an efficient way to measure soil organic carbon pools in soil carbon turnover models

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Soil carbon (SC) models are based on several conceptual SC pools with different turnover rates. There are various methods have been proposed to separate soil samples into soil carbon fractions with distinct chemical and physical characteristics providing data to populate the pools. However, these conventional fractionation procedures are costly and time-consuming to produce the model outputs, which may make them less attractive than just analysing total soil carbon. Therefore, this study focuses on testing the efficacy of near-IR (NIR) and mid-IR (MIR) data taken from whole soil which can be used to populate soil carbon models. Fifty soil samples have been collected from three major bioregions of New South Wales, and totally dispersed and disaggregated into aggregate sizes (<63, 63-250, >250 µm). Total carbon, organic carbon and inorganic carbon will be measured in whole, dispersed and aggregated soils by CHN analyser, NIR and MIR spectroscopy. Since, SC is closely related to the formation and stability of soil aggregates. The relationship between aggregate formation and the modelling of carbon may prove useful to predict and numerate the associated soil structural changes. From this study it will be possible to establish the degree of soil aggregation can be significantly correlated with NIR- and MIR-measured pools from whole soil; and if data are meaningful for carbon models linked to soil aggregation, such as Struc-C, which describes how the SC influences the dynamics of soil structure, and consequently, soil physical behaviour.

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# OzSoilML: An Australian standard for exchange of soil data

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CSIRO have developed a soil information model (OzSoilML) to meet Australian Collaborative Land Evaluation Program (ACLEP) requirements for the exchange, collation and delivery of nationally consistent soil data for Australia. OzSoilML enables delivery of the data stored in the Australian Soil Information System (ASRIS) based on the field methods specified in the 'Australian Soil and Land Survey Field Handbook'. Being based on Open Geospatial Consortium (OGC) and International Organization for Standardization (ISO) standards, OzSoilML follows other successful international GML-based standards, such as GeoSciML, EarthResourceML and GroundWaterML. Using these standards enables the discovery, query and delivery of the soil and landscape data via standard OGC Web Feature Services. The ACLEP implementation of these services will include well-governed model and service registries, along with vocabulary bindings to ensure the soil data services are interoperable at both the schematic and semantic levels. These data services will incorporate Australian data within broader international soil data standardisation and delivery initiatives, such as the GlobalSoilMap.net project.

The International Union of Soil Sciences (IUSS) established the IUSS Working Group on Soil Information Standards (WG-SIS) to develop, promote and maintain a standard to facilitate the exchange of soils data and information. Developing an international exchange standard that is compatible with existing and emerging national and regional standards is a considerable challenge. Although OzSoilML specifically meets Australian requirements, it also attempts to harmonise with the various emerging soil data standards from around the world.  As such, OzSoilML is being considered as a profile of the more generalised SoilML model being progressed through the IUSS Working Group.

# Use of mid-infrared spectroscopy to characterise and predict particulate organic carbon and nitrogen in soils after land-use change from agriculture to plantation forestry

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## Abstract

The expansion of hardwood forest plantations in south-western Australia has been mostly on agricultural lands. Land-use change can significantly affect the quantity and quality of soil organic matter (SOM), and notably that of particulate organic matter (POM). Mid-infrared spectroscopy (MIRS) is a well established analytical methodology for qualitative and quantitative analysis of SOM. The present study comprised soils (0-10 cm) from 31 paired sites affected by land-use change (from pastures to *Eucalyptus globulus* plantation). Composition of SOM was assessed using specific mid-infrared spectral peaks and vibrations in soil spectra. MIRS calibration models for particulate organic carbon (POC) and nitrogen (PON) were developed using partial least squares regression (PLSR). The MIRS-PLSR calibration models developed from a smaller subset gave precise POC and PON predictions with coefficients of determination $R^2$ = 0.97 and 0.94 respectively. There was no significant difference between predicted POC and PON concentrations between land-uses.

## Key Words

Land-use change, soil organic matter (SOM), particulate organic carbon (POC), particulate organic nitrogen (PON), mid-infrared reflectance spectroscopy (MIRS)

## Introduction

Soil organic matter (SOM) is central to soil fertility, affecting for example, N stores and cycling, and water holding capacity. It follows that an understanding of the composition and decomposition of SOM should underpin management for sustainable production and the provision of ecosystem services (e.g. C storage). The quantity, quality and spatiotemporal distribution of SOM may be affected by land-use change. One approach to study the quality of SOM is to separate fractions on the basis of particle size (Christensen, 2001). Particulate organic matter (POM) is generally enriched in plant-derived material (lignins, waxes and amino compounds), which is included with the sand-size materials and protected C in aggregates (Calderón *et al.*, 2011). POM is a sensitive indicator which often responds more rapidly than total organic matter to changes in vegetation (Cambardella and Elliott, 1992). Mid-infrared spectroscopy (MIRS) is a rapid and effective analytical methodology to measure and characterise SOM (e.g. organic functional groups) and its fractions (e.g., Janik *et al.*, 2007; Viscarra Rossel *et al.*, 2008; Calderón *et al.*, 2011; Demyan *et al.*, 2012).

In south-western Australia, establishment of *Eucalyptus globulus* plantations since the mid 1990s has been mostly on land previously managed for agriculture (usually pastures). After the first rotation, N availability declined in these soils, probably due to lower N inputs due to removal of clover and the wider C to N ratio and reduced decomposability of eucalypt litter compared with pasture litter (Mendham *et al.*, 2004). The objectives of the present study were a) to identify specific SOM functional group vibrations in soil MIR spectra and thereby the composition of SOM; b) to develop preliminary MIRS and partial least squares regression (PLSR) calibrations to predict the C and N contents of POM (respectively POC and PON); and c) to determine differences between pasture and plantation soils in predicted POC and PON.

## Methods

### *Sites and soils*

Soils from 31 paired sites (0–10 cm depth, 4 replicates per site, $n$ = 248) comparing pasture and *E. globulus* plantations in south-western Western Australia (see O'Connell *et al.*, 2003) were further analysed. Soils were sampled in the first rotation at 6 to 11 years of age. The plantations were established on long-term pasture lands, which were previously natural forests. The sites represented a range of soil textures (sandy to clay loam) and climate (e.g. rainfall 600 to 1300 mm $yr^{-1}$). Soils from a representative subset of ten sites had been analysed for POC (>45 µm) and PON (Mendham *et al.*, 2004) and were used to develop MIRS-PLSR calibration models.

*MIRS and PLSR*

Soil samples (<2 mm) were dried (< 40°C) and finely ground in a vibrating steel puck mill for one minute (Rocklabs, Auckland, New Zealand) for MIRS.

Mid-infrared diffuse reflectance spectra were collected using a PerkinElmer Spectrum One FT-IR spectrometer from 4000 to 450 $cm^{-1}$ at 8 $cm^{-1}$ resolution. Scans were co-added for one minute. A reference background spectrum was recorded every thirty minutes or every 16 samples. Matlab (version R2010a) (The MathWorks, USA) and PLS_Toolbox 4.2 (Eigenvector Research Inc., WA, USA) were used to fit PLSR calibration with leave-one-out cross-validations. Data were first transformed using a square root to normalise the distributions. Spectra were pre-processed using multiplicative scatter correction (MSC) followed by mean-centering. Predicted concentrations of POC and PON were obtained by back-transforming (squaring) the model predicted data.

## Results and Discussion

*Spectral features*

Figure 1 illustrates characteristic absorption features and functional groups in soil spectra from two example paired pasture and plantation sites. These spectral intensities were consistent with spectral features reported in other soil MIRS studies (e.g. Janik *et al.*, 2007; Viscarra Rossel *et al.*, 2008; Bornemann *et al.*, 2010; Calderón *et al.*, 2011; Demyan *et al.*, 2012).

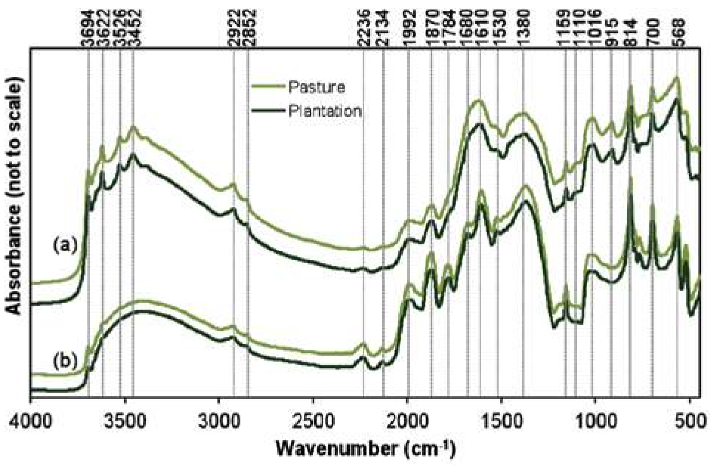


**Figure 1. Mid-infrared spectra (4000-450 $cm^{-1}$) of 0-10 cm soils from two example pasture and plantation paired sites with different soil texture and POC contents (averages of pasture and plantation): (a) sandy loam, POC 62.8 g $kg^{-1}$ soil, and (b) sand, POC 13.9 g $kg^{-1}$ soil.**

In general across all sites, the soil spectra for both pastures and plantations within each site showed similar peak patterns, however distinct differences were observed between the sites owing to organic carbon concentrations and mineralogy (clay mineral O-H stretching vibrations at 3694 and 3622 $cm^{-1}$, Si-O stretching near 2000, 1870 and 1790 $cm^{-1}$, intense silicate band inversion near 1080 $cm^{-1}$, and clay mineral and quartz vibrations below 900 $cm^{-1}$). Distinct bands representing alkyl C-H stretching modes of the SOM were evident at 2922 and 2852 $cm^{-1}$ since these aliphatic $CH_2$ vibrations are free from overlaps or masking from other intense vibrations. The bands between 2200 and 2000 $cm^{-1}$ could be related to silicate overtones. Carbonyl C=O stretching from amide-I was found at 1680 $cm^{-1}$. Strong peaks at 1630-1610 $cm^{-1}$ were due to water –HOH deformation, and $COO^-$ symmetric stretching of carboxylates. Peaks near 1600 were identified as C=C stretching of aromatic compounds. Small bands at 1540-1510 $cm^{-1}$ could be associated with N-H deformation and lignin vibrations, C=N stretching of amide-II, and aromatic C=C stretching. Major intensities between 1380 and 1360 $cm^{-1}$ represented C-O stretching and O-H deformation of phenols originating from lignin compounds. Peaks between 1170 and 950 $cm^{-1}$ were attributed to both carbohydrates (C-O and C-O-C stretching) and alumina-silicate lattice vibrations (Al-O-Si, Si-O-Si and Al-

O-Al) of clay minerals. Peak at 915 $cm^{-1}$ was formed due to C-H deformation and organic bands below 900 $cm^{-1}$ were dominated by aromatic deformations.

Although spectral differentiation between land-uses was not evident, soils with higher clay content were associated with higher organic C and N contents (data not shown) and increased peaks of aliphatic C bands (2922, 2852 and 1460 $cm^{-1}$) probably indicating demethylation of lignins in the decomposition process. On the other hand, sandy soils displayed strong peaks ascribed to amides (1680 and 1540 $cm^{-1}$), aromatic and carboxylic C (1610, 1540 and 1390 $cm^{-1}$) from proteins and lignins, together with lower aliphatic C peaks; these functional groups are indicative of plant materials at slow rates of breakdown.

*Predictive models for POC and PON*

Although based on relatively few data, the PLSR calibrations for POC and PON were precise and unbiased (Figure 2). The cross-validation for POC showed excellent results with $R^2$=0.97, RMSEC=0.26, RMSECV=0.45, and calibration bias= -0.01. Five latent variables (LV) were found to be optimal for POC model, although the first three accounted for 94% of the explained variance (Figure 3a).

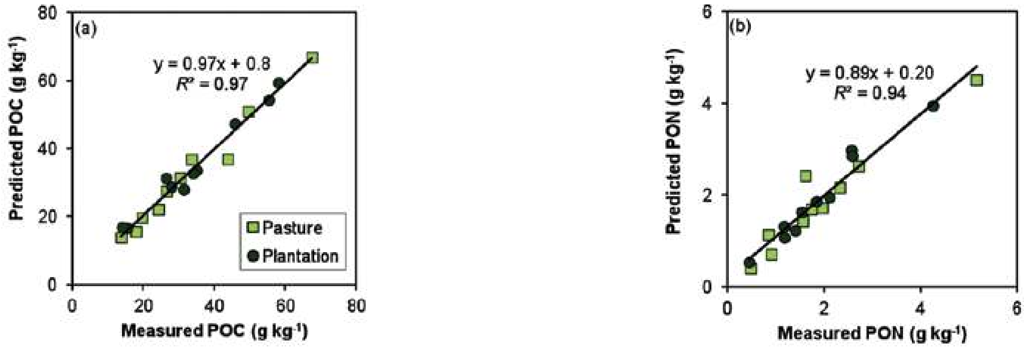


**Figure 2. Relationship between measured and MIRS-PLSR predicted (a) POC and (b) PON in pasture and plantation soils.**

Dominant peaks in the loading weights were characteristic of aliphatic C-H stretching modes (2922 and 2852 $cm^{-1}$), carbohydrate C-O vibrations (1110-1080 $cm^{-1}$ and 1016 $cm^{-1}$), proteins from amide bands (1660 and 1552 $cm^{-1}$), aromatic intensities (1610 $cm^{-1}$), aliphatic C-H deformation (1450 $cm^{-1}$), carboxylic acid and aromatic plane deformation (1226 $cm^{-1}$) and aromatic structures below 900 $cm^{-1}$. Intensities < 1400 $cm^{-1}$ suggested the presence of cellulose although lignin can also contribute to this region (Janik *et al.*, 2007). Bornemann *et al*. (2010) observed similar functional groups contributing to POM-C models and assigned them mostly to cellulose and unaltered lignin in fresh undecomposed plant debris in the 2000-250 μm fraction and to aliphatic $CH_2$ and $CH_3$ groups as decomposition products in 250-53 μm fraction.

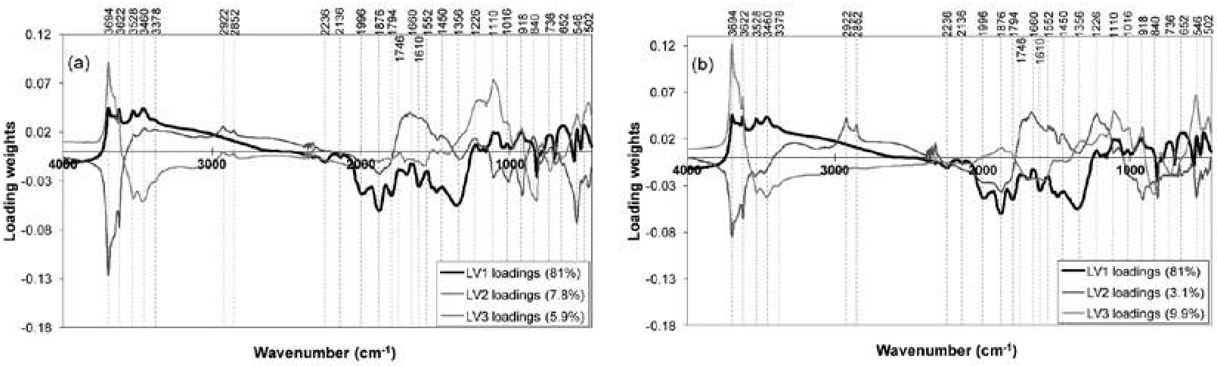


**Figure 3. Loading weights of the first three latent variables showing principal spectral features contributing to (a) POC and (b) PON predictions in the calibration model.**

The PON cross-validation was also excellent with $R^2$=0.94, RMSEC=0.10, RMSECV=0.18, and calibration bias= -0.02. Although the PON prediction model used 6 LVs, first three LVs explained 94% of the variances (Figure 3b). The first loading weight indicated vibrations from O-H groups, phenols, carbohydrates and aromatic moieties; while the second and third loading weights featured intense aliphatic C-H stretching modes (2922 and 2852 $cm^{-1}$), carboxylic acid (1746 $cm^{-1}$), strong amide peaks (1660 and 1552 $cm^{-1}$), aliphatic C-H (1450 $cm^{-1}$), carboxylic acid and aromatic deformation (1226 $cm^{-1}$) and carbohydrates (1110 and 1016 $cm^{-1}$).

In both POC and PON models, loading weights from second and third LVs contributed to differentiate the dominance of carbohydrate and lignin features on POC, and amides and carboxylic acids on PON. Additionally, positive relationships of mineral absorptions (clay minerals, silica and quartz) with predicted POC and PON were evident in first and third loading weights, suggesting close association of occluded organic fractions with clay minerals or present as thin coating on sand particles.

*Differences in POC and PON between land-uses*

Across the 31 sites, there was no significant difference in the concentrations of predicted POC and PON between land uses (Figure 4), although the averages were slightly higher in plantation soils (38.0 and 2.15 g $kg^{-1}$ respectively) compared to pasture soils (36.4 and 2.08 g $kg^{-1}$). This is consistent with that found by Mendham *et al*. (2004) at 10 of the sites where POC was directly measured and averaged 74.6% of whole-soil C in plantations and 61.8% in pastures and where total C did not differ between land uses. The relatively small difference in POC and PON between the land-uses in the present study probably reflects the short time (6 to 11 years) since reforestation.

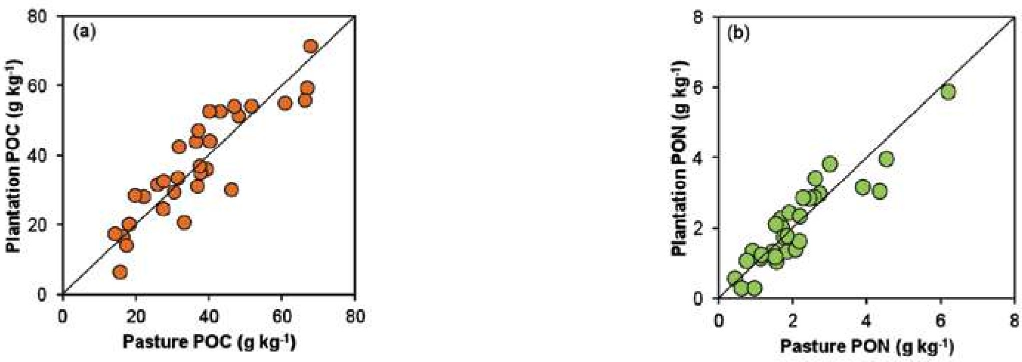


**Figure 4: Variation in predicted a) POC and b) PON in plantation and pasture soils across 31 sites. Values are means of n = 4. The diagonal line in each graph represents 1:1 ratio.**


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# Image analysis and physical study to characterize claypan properties, in Fadak park- Isfahan – Iran

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**Abstract**

Concern about the claypan properties in claypan soils is very important for agricultural and environmental efficiency. The study was conducted on the physical and micromorphological properties of clay pans in Vertisols. The field experiment was conducted in Fadak Park, an 80 ha area that is located north of Isfahan city in Iran. The clay content (48 to 60, 44 to 66.4 and 45.6 to 64.8 %), bulk density (1.4 g $cm^{-3}$ to 1.9 g $cm^{-3}$) and soil saturation percentage (47.21% to 79.17%) increased and porosity percentage (45.3% to 28.6%) decreased relative to the overlying soil. Thin sections produced from the resulting clods/aggregates from the claypan for micromorphological study indicated the lower porosity in the claypans as a result of compaction. $Fe^{+2}$ and $Mn^{+2}$ nodules and intercalation, showed reduction condition in these layers. Also, the infiltration rate in 3 areas in Fadak Park with the double ring method showed an increase from profile number 1 to profile number 3 (0.2 cm $min^{-1}$ to 3.7 cm $min^{-1}$). Large vertical desiccation cracking and existing roots are most important factors for increasing the infiltration rate in profile number 3 than in the others. The purpose of this study is to understand the properties of claypan layer for better recognizing this layer in agricultural systems and increasing plant yields. All of these parameters which affect soil compaction may decrease plant growth due to decreased water and nutrition availability, aeration and limitation in root growth in undesirable growth areas. Therefore, efforts should be made to minimize soil compaction for improving the agricultural and landscape qualities in the claypan which is restricted in subsoil layers to increase the potential for optimal conditions for crop production.

**Keywords**

Claypan soil, Physical properties, Micromorphology, Vertisol

## Introduction

Soil compaction causes damage to physical properties of soil and it is the cause of crop yield limitation in agroecosystems (HyeMin *et al*. 2010). These soils are described by silt loam surface horizon with an abrupt argillic horizon within 10 to 50 cm of soil surface (Motavalli *et al*. 2003 and HyeMin et al. 2010). Claypan layers usually have a considerable amount of smectitic clay minerals (40 to 60%) which reduce vertical water drainage and have a unique hydrology controlled by a slow water movement in the soil matrix of the limiting clay layer (HyeMin et al. 2010). Soil moisture contents have a great effect on infiltration rates (Jamison *et al*. 1961). Desiccation soil cracking is usually observed in claypan soils. As these soils dry, there is a soil matrix volume decrease caused by collapsing layers of the clay micelles. The fast transfer of water and dissolved material through desiccation soils cracks can go to crop water and nutrient stress as well as ground and surface water contamination (Baer et al. 2009). The harmful effects of compaction on the physical properties of soil for crop production include higher soil bulk density and lower total porosity (Motavalli *et al*. 2003). Compaction may unfavourably influence crop growth by reduced Ksat (Pengthamkeerati *et al*. 2006), decrease drainage and water movement in soils and limit soil aeration (Baer *et al*. 2009). Climate and time are two factors that affect compaction on crop yield. During the dry years, moderate compaction can increase growth, but with severe compaction, limited root growth can lead to increased moisture stress and decreased produce. In moist years, decreased soil aeration and water infiltration caused by soil compaction can limit nutrient accessibility and increase run-off (HyeMin 2010). The objective of this study was to study the physical and micromorphological properties of claypan layers in Vertisols for increasing the crop yields in these types of soils.

## Materials and Methods

### *Soil and field sampling*

This study was conducted during 2008 at Fadak Park. The study area is located on the river alluvial plain of ancient Zayandeh Rood path north of Isfahan-Iran. The elevation of the site relative to the sea site was

1573 m, being about 85 ha in area. The soil is classified as a fine clayey, mixed, semiactive, thermic, vertic haplocalcids in Soil Taxonomy. Mean annual precipitation from 1975 to 2003 (Metrological Organization) was 108 mm and the mean annual minimum and maximum temperatures were 5.9 and 23.8°C, respectively. For studying the soils, 6 profiles were dug. The soil was collected from 3 of them. Soil samples were taken from the surface to a depth of about 160 cm (claypan layer) to analyse physical and micromorphological properties. The soil cores were used to determine soil bulk density. Morphological properties such as boundaries, sand stone, CaCO3, porosity, roots, mottling and structure of soil horizons were described. The amount of total porosity (E) was calculated via the relation between bulk density (ρb) and particle density (ρs) as: $E= \{1- (\rho b / \rho s)\}\ 100$

*Preparation and description of thin section*

In the laboratory, the clods taken from the claypan were placed in containers and after that resin mixture (include: 100 cc Resin, 25 cc Acetone, 0.5 cc Estearic acid and 2 drop Cobalts catalyst) was added to completely cover the samples. They were left for 2 months. The resin was hardened slowly in an oven at laboratory temperature. From these resin blocks, sections were cut and mounted on slides and polished to 30 Microns thickness. These were examined using Lytz Microscope under Plane Polarised Light (PPL) and cross polarized (XPL) light at 40X magnification. Micromorphological properties of claypan include pedofitures, microstructure, b-fabric, C/F related distribution pattern, organic matter and voids described with Stoops Method (Stoops 2003).

**Results and Discussion**

*Morphological properties*

The morphological study indicated the massive structure without roots and sand stone in the depth. The amount of porosity and $CaCO_3$ decreased as a function of depth compared to the surface depth zones in all profiles. Baer et al. (2009) explained that compaction and water logging in the claypan layers caused the reduction condition in these layers. This condition causes a significant mottling effect. The most common soil structure to the surface depth zone were granular in the profiles which changed to fine angular blocky, medium prismatic and medium angular blocky and then massive structure in the claypan layers. Hussein and Adey (1998) noted that formation of a granular structure in the surface soil showed rapid desiccation, while a compound prismatic-angular, sub angular blocky structure in subsoil is thought to be due to a combination of stress and subsequent slow drying.

*Physical properties*

The summary of the physical properties are shown in Table 1. The amount of the clay, silt percentage, the bulk density and the saturation percentage increased while, amount of sand stone and porosity decreased (HyeMin 2010; Motovalli 2003).

**Table 1: Physical properties of profiles number 1, 2 and 3**

| | Depth | | | % | | Bulk | % | |
|---|---|---|---|---|---|---|---|---|
| Horizon | (cm) | Texture | Clay | Silt | Sand | Density | Porosity | Saturation |
| | | | | *Profile No.1* | | | | |
| C | 0-13 | Clay | 48 | 22.68 | 29.32 | 1.51 | 43 | 42.98 |
| 2A | 13-22 | Clay | 54.4 | 26.84 | 18.78 | 1.63 | 38 | 49.40 |
| 2BW1 | 22-34 | Clay | 60.8 | 21.4 | 17.8 | 1.94 | 26 | 52.34 |
| 2BW2 | 34-50 | Clay | 61.6 | 20.72 | 17.68 | 1.57 | 40 | 56.86 |
| 2Bd | 50-89 | Clay | 60 | 25.56 | 14.44 | 1.9 | 30 | 57.51 |
| 3Ab | 89-130 | Clay | 48 | 21.96 | 30.04 | 1.93 | 27 | 78.57 |
| 3Bdk | 130-160 | Clay | 70.4 | 25.44 | 4.16 | 1.82 | 21 | 82.42 |
| 4 Bdk | 160-190 | Clay | 70.4 | 25.5 | 2.1 | 1.89 | 21 | 82.4 |
| | | | | *Profile No.2* | | | | |
| A | 0-10 | Clay | 44 | 37.36 | 18.64 | 1.13 | 57.3 | 49.74 |
| B | 10-18 | Clay | 48.8 | 33.84 | 17.36 | 1.73 | 34.7 | 47.78 |
| Bk1 | 18-43 | Clay | 52 | 35.76 | 12.24 | 1.69 | 36.2 | 49.14 |
| Bk2 | 43-80 | Silty | 50.4 | 47.84 | 1.76 | 1.8 | 32 | 60.24 |
| Bk3 | 80-140 | Clay | 62.4 | 37.04 | 0.56 | 1.88 | 29 | 73.20 |
| Bd | 140-160 | Clay | 66.4 | 31.84 | 1.76 | 1.89 | 28.6 | 77.92 |
| 2Bd2 | 160-190 | Clay | 66.4 | 32.5 | 1.1 | 1.9 | 28 | 77.8 |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| *Profile No.3* | | | | | | | | |
| A | 0-10 | Silty | 45.6 | 42.72 | 11.68 | 1.55 | 36.6 | 48.95 |
| B | 10-35 | Clay | 55.2 | 32.32 | 12.48 | 1.86 | 40.3 | 49.00 |
| Bk1 | 35-65 | Silty | 56 | 41.16 | 2.84 | 1.67 | 31.6 | 62.29 |
| Bd1 | 65-120 | Clay | 64.8 | 33.92 | 1.28 | 1.68 | 30.9 | 67.70 |
| 2Bd2 | 120-200 | Clay | 64.8 | 34.2 | 1 | 1.92 | 31.6 | 77.21 |

*Micromorphological study of claypan layer*

The structure was moderately separated and subangular blocky. Subangular blocky was reported by Hussein (1998) for the Vertisols microstructure and is shown in Figure 1.

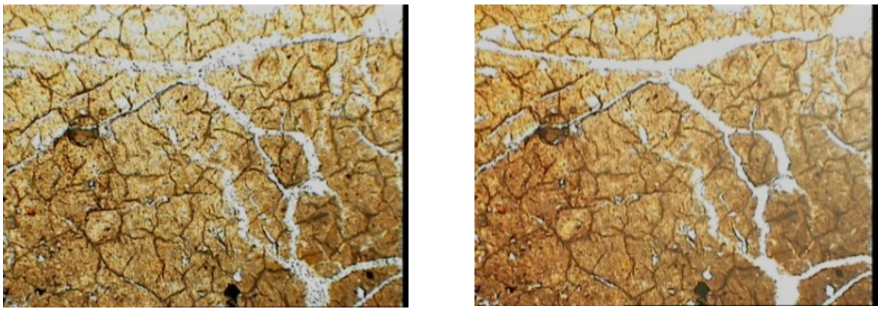

**Figure 1: Subangular blocky microstructure, channel and vugh voids; PPL (Plane Polarised Light-left) and XPL (cross polarized -right), Bd horizon (130-160 cm)**

**Table 2: Micromorphological properties of claypan layers**

| | |
|---|---|
| Pedofitures : Intercalation | Serrated Intercalation |
| Pedofitures : Nodules | Typic nodules(Fe & Mn oxides) intrusive pedofiture |
| Microstructure | Moderately separated subangular blocky microstructure |
| b-fabric | Crystallitic b-fabric |
| C/F related distribution pattern | Fine Monic |
| Organic matter | Organic Pigment – Amorphous organic fine material |

The soils had planar voids dominant and few vughs, channels, chamber voids. Hussein (1998) reported the variation between voids in topsoils at the Vertisols from vughy, packing voids, planar, whereas subsoils seem to be dominated by planar voids.

**Table 3. Type of voids in claypan layers**

| Horizon Depth(cm) | Voids: Type | Shape | Size (Micron) | Abundance: Of the total thin section | Abundance: Of the total pores | Accommodation | Roughness & Smoothness |
|---|---|---|---|---|---|---|---|
| Bdm | Chamber | | 100-400 & 200-400 | 7 | 15 | | Rough |
| | Plane | Zigzag | 15 | 5 | 70 | Accommodated | Smooth |
| | Chamber | - | 100-300 | 10 | 10 | | Smooth |
| | Channels | Arched | 200 | 5 | 15 | | Smooth |
| | Vughs | Stare shaped & regular | 100-300 & 100-400 | 10 | 5 | | Rough to Smooth |

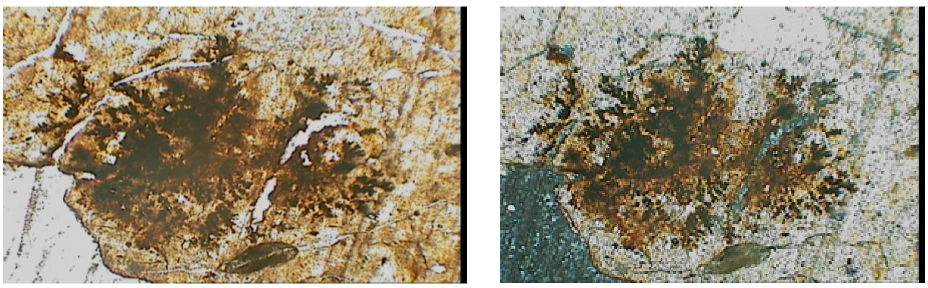

**Figure2: Fe2+and Mn2+ pedofitures; PPL (left) and XPL (right) in Bd horizon (130-160 cm)**

*Infiltration Rate*

Infiltration rates are shown in Figure 3. In comparison, between these profiles, soil compaction was greater in the 50-89 cm depth in profile number 1 (1.9 g cm$^{-3}$) than profile number 3. Also, large vertical cracks in the profile number 3 and existence roots to 120 cm depth compared with profile number 1 indicated higher infiltration rate in this profile. Large cracks (from millimetres to centimetres wide), may induce preferential flow. Large cracking occurs in clayey soil (Fig. 3), because the larger cracks are vertical, it caused higher infiltration (Lee 2004; Chertkov 2012).


**Figure 3: Deep fissure caused by contraction in July in Fadak Park-Isfahan-Iran (profile number 3, left). Infiltration rate in Fadak Park (right)**

## Conclusion

In all the claypan layers, the clay contents and the bulk density increased and the total porosity decreased. In fact, excessive compaction in the claypan was measured. In the topsoil, the clay content is usually lower due to erosion and eluviation which causes a lateral or downward movement of clay fraction. This compaction might create restrictions for plant growth; therefore, it is important to understand the soil properties prior to farming in agricultural and environmental ecosystems to improve results. Addition of organic matter and deep ploughing may improve these soils, as discussed by Nyakatawa (2001). Organic matter decreased bulk density and increased cation exchange capacity, soil moisture content and infiltration rate. Therefore, it creates a better environment for root systems and promotes the growth of green plants in this type of soils.

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# Piloting an early warning wind erosion threat mapping method for the Victorian Mallee

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An early warning wind erosion threat mapping method has been piloted in the Mallee region of north-west Victoria. Fine-scale landform mapping was combined with satellite-derived, seasonal crop data to produce a wind erosion threat assessment for agricultural lands for 2012.

Agricultural land in the Victorian Mallee supports approximately 1.8 million hectares of dryland cropping. This is a low rainfall environment (MAR 270-370 mm) with aeolian landforms that are highly susceptible to wind erosion. Ground cover is critical to protect the soil from erosion, especially for the period between harvest and establishment of the following season's crop.

Paddock-scale landform components were assigned wind erosion susceptibility ratings based upon soil surface texture and topographic profile. Time series MODIS (Moderate Resolution Imaging Spectro-radiometer) pixels (nominally 250 m) were classified for land cover type and relative biomass using the Enhanced Vegetation Index (EVI) for the 2010 and 2011 cropping seasons. Different cover types, e.g. cereal or legumes, were weighted according to the potential of their residues to provide protection of the soil surface. The weighted land cover was combined with the relative biomass data to produce a classification of potential post-harvest ground cover provided by crop residue. The landform susceptibility and potential ground cover data were combined in a 'threat matrix' to map relative wind erosion threat for 2012.

The method is a cost-effective, repeatable and timely way to annually determine areas most threatened by wind erosion.

# Evaluating contact angles from measured soil-water retention curves under controlled wetting and drying cycles in water-repellent soils

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Soil organic matter can modify the surface properties of the soil mineral phase by changing the surface tension of the mineral surfaces and influences the rate of wetting and soil-water retention properties. The surface contact angle (*CA*) of water-repellent soil gradually decreases during continuous contact with water droplet, subsequently allowing water imbibition into the media. However, the time dependence in *CA* with soil-water contact time on water retention characteristics is not fully understood. In this study, the water retention curves under controlled wetting and drying cycles for volcanic ash soil (VAS) and hydrophobized sand samples were measured using hanging water column equipped mini T-TDR coil probe. The contact angles ($CA_{\_SWCR}$) were evaluated based on the inflection points ($h_{ip}$) on the main imbibing curves assuming the reference soil $CA_{\_SWRC}$=0, where the water imbibition in porous media was introduced either by step-wise increment of water pressure ( from -80 cm to +5 cm $H_20$) or a constant positive pressure (+15 cm $H_20$). Degrees of hydrophobicity greatly affect the soil-water retention properties of WR soil. The calculated $h_{ip}$ gradually decreased with increasing surface *CA* for all the samples. $CA_{\_SWRC}$ values ranged from 20$^o$ to 48$^o$ for VAS and from 40$^o$ to 66$^o$ for hydrophobized sand. The calculated $CA_{\_SWRC}$ were smaller than the measured surface *CA*, indicating the reduction of contact angle with soil-water contact time. Additionally, difference in water-filled pore distributions under controlled wetting and drying cycles were examined by using soil-water capacity and pore-size density as a function of ψ.

# Thursday 6 December

## SOIL CARBON AND CLIMATE CHANGE (BIOCHAR/COMPOST)

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# Macadamia biochar as a growth stimulus for *Eucalyptus nitens* forestry nurseries

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**Abstract**

The use of Biochar as a potting mix amendment to reduce fertiliser rates and aid the growth of *Eucalyptus nitens* seedlings grown for Tasmanian commercial forest plantations was investigated. The early growth of *E. nitens* seedlings (from seed to nine-month-old seedling) was analysed under controlled conditions (glasshouse) in a pot trial. The treatments combined two fertilizer levels (100% and 50% of the commercial rate) with eight biochar doses (0, 2, 5, 10, 20, 50, 80 and 100 t/ha). Seedling height was measured weekly during the experiment, while agronomic response data was collected at four destructive harvesting periods at 135, 177, 219 and 268 days after planting. The results of the agronomic response (height, biomass) showed a strong dependence on fertilizer level but no significant differences in reference to biochar levels. Biochar was found to influence the level of water content in leaves and soil.

**Key Words**

Biochar, *Eucalyptus*, growth

## Introduction

Biochar as a soil amendment is a potential new technology that may assist both agriculture and forestry to meet economic and environmental goals while concomitantly sequestering atmospheric carbon in the soil (Klein *et al*. 2007, Milne *et al*. 2007, McHenry 2009). While much research in Australia is dedicated to biochar used as a soil amendment for agricultural crops on Australia's mainland, there is limited data available concerning its use in forestry, or on its use in the soils and climate of Tasmania.

The beneficial effects of biochar on soil characteristics and crop yield have been reported both in the laboratory and at a field scale (Chan *et al*. 2007, Lehmann and Joseph 2009). However, these responses are variable and the literature reports a wide range of biochar application rates, and an equally wide range of plant responses. The effect of biochar on forestry species has not been investigated on a wide scale or from a commercial point of view.

Forestry is a significant industry in Tasmania, with large-scale plantations of Radiata pine (*Pinus radiata* D. Don) and *Eucalyptus* (*E. globulus* and *E. nitens* H.Deane & Maiden) which play an increasingly important role in supplying national and international demand for timber. Propagating robust seedlings for planting in the field is an important part of plantation establishment and influences potential yield, while also being a significant budgetary component. In this research we investigate the hypothesis that biochar added to nursery potting mix will increase fertilizer use efficiency and enable nursery managers to reduce fertilizer inputs, ensuring at the same time that seedling quality remains similar to that of a full fertilizer treatment without the addition of biochar.

## Materials and Methods

### *Experimental design and layout*

The glasshouse pot trial was established on 09/05/2011 and the final destructive harvest took place on 02/02/2012. Four destructive sampling harvests (H1-H4) took place on 135, 177, 219 and 268 days after planting (DAP). This study was a factorial combination of eight biochar and two fertilizer rates, arranged in a randomised complete block with three replicates of four sample plants. At each harvest one plant from each replicate was destructively sampled for soil and plant analysis. Plants were raised from seed in 4L pots at the Horticultural Research Centre on the Sandy Bay campus of the University of Tasmania. Day/night temperatures were approximately 24/20ºC in winter and 26/20ºC in summer and the plants were irrigated to meet the seasonal evapotranspiration requirements. Seedlings were watered to field capacity on the morning prior to harvest, approximately 24 hours prior to data collection of the potting mixes wet weight data.

*Biochar*

Biochar used in the pot study was provided by the Rainbow Bee Eater Project Pty Ltd and produced from macadamia nut shells at a HTT (highest temperature treatment) of 450-480ºC. Some chemical characteristics of the product are presented in Table 1. Biochar was evenly incorporated within the potting mix at 0, 2, 5, 10, 20, 50, 80 and 100 t/ha (B0, B5, B10, B20, B50, B80 and B100) calculated on a volumetric basis.

**Table 1: Chemical and physical characteristics of the biochar incorporated into the potting mix. This biochar was produced from macadamia shells at a HTT of 450-480ºC. PAH –Polycyclic aromatic hydrocarbons DM –Dry matter.**

| Electrical conductivity [dS/m] | pH ($H_2O$) | pH ($CaCl_2$) | Nitrogen [%] | Carbon [%] | Mineral N [mg/L] | PAHs [mg/kg DM] | CEC [meq/100g] |
|---|---|---|---|---|---|---|---|
| 0.59 | 8.76 | 8.07 | 0.57 | 78.03 | 3.6 | <10 | 2.38 |

*Soil and fertilizer*

A seed raising mix was prepared by staff at the Forest Nursery (Forestry Tasmania) to emulate the industry standard. The mix consisted of pine bark (72%), washed sand (18%) and peat moss (10%). The fertilizer used was a mix of Osmocote Exact® 3-4 mth (Everris), Osmocote Plus® 8-9 mth (Everris), dolomite lime, ferrous sulphate, Micromax® (Everris) and rock gypsum. Fertiliser was applied to the pots at 100% and 50% (F50 and F100) of the prescribed dose used by forestry nurseries in Tasmania.

*Plant material*

The seed line originated from a single open pollinated family harvested in 2008 from a native stand in the central Victorian highlands. Nine seeds were planted into each pot at a depth of 5-10 mm. Six weeks after germination, at which point 60% of the seedlings had germinated, seedlings were thinned to the three strongest in each pot, and after another ten days to one seedling per pot.

*Measurements*

For each harvested plant leaf number, leaf area, stem fresh and dry weight, root fresh weight and leaf fresh and dry weight was measured. Biomass was air-dried at 60°C for 72 hours. Leaf area was calculated via image analysis using ImageJ® ver.1.45. Beginning on the 10th August 2011, seedling height was measured weekly until the end of pot trial.

*Statistics*

Data was analyzed using the IBM SPSS Statistics ver. 19, using ANOVA and an LSD probability of $P$=0.05.

**Results**

*Growth*

The growth rate of the seedlings was not dependent on biochar application rates but was dependent on fertilizer level (*P=0.05)*, with a higher growth rate at F100.

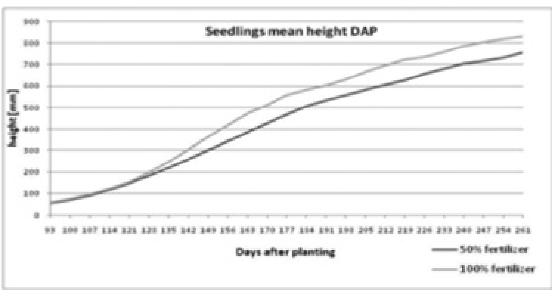


**Fig.1: Mean height (mm) of *E.nitens* seedlings fertilised at 50% and 100% of commercial rates reported from 93 to 260 DAP**

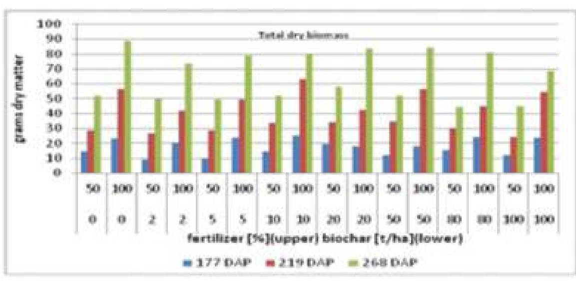


**Fig.2: Dry weight (g) of *E. nitens* seedlings at 177, 219 and 268 DAP in response to fertilisation at two levels (50% and 100% of the commercial rate) in factorial combination with 8 Biochar application rates (0-100 t/ha).**

Height increase between fertilizer levels was similar at 4 mm/day until 128 DAP (*P*=0.013), and after this mean growth rate under F100 was 1.5mm/day, greater than that under F50 (Fig.1). This effect on seedling growth rate continued until approximately 177 DAP, at which time growth rate at F100 again declined to a rate similar to that under F50 (*P*≤0.01). The gain in total height under F100 was maintained until the end of the experiment. Total dry biomass was also greater under F100 (*P*=0.000) at 177, 219 and 268 DAP (Fig. 2).

*Agronomic response*

Leaf number, leaf area, leaf wet and dry weights, stem weight, soil water and root mass were dependent on fertilizer level at all four harvests (*P≤0.05*). At the first harvest (135 DAP), both leaf and soil moisture were influenced by fertilizer and biochar application. Leaf moisture content increased in response to increased fertilizer (*P*=0.001) while the response to biochar application rates was mixed (*P*=0.003) (Fig. 3). Gravimetric soil moisture content decreased under F100 (*P*=0.004) and in response to all levels of biochar (Fig 4). These effects were not evident beyond 135 DAP.

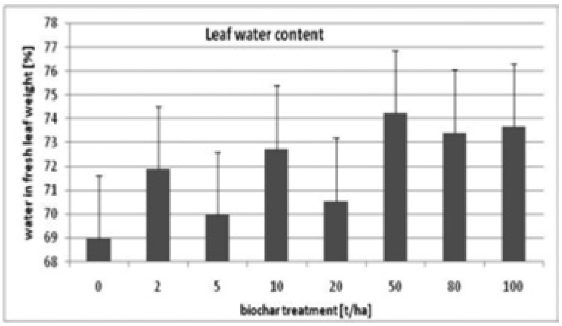


**Fig.3: Leaf moisture content (%) in potted *Eucalyptus* seedlings at 135 DAP in response to biochar applications increasing from 0 to 100 t/ha. Error bars indicate the LSD (P=0.05)**

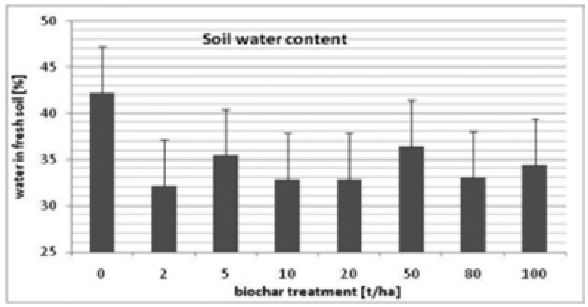


**Fig.4: Soil moisture content (%) in potting mix at 135 DAP in response to biochar applications increasing from 0 to 100 t/ha. Error bars indicate the LSD (P=0.05)**

Seedlings under B0 treatments had the lowest leaf moisture content at 135 DAP, although this was not statistically different from B5 and B20 treatments. The highest water content in leaves was noticed under B50-100 biochar treatments and the lowest water content was found with no biochar treatment. The treatments in between (B2, B5, B10 and B20) do not show a clear pattern of biochar effect on leaf water content. In contrast, soil water content was highest in soil not amended with biochar (B0, *P*=0.025), while its application at all rates reduced gravimetric soil moisture.

## Discussion and Summary

The results presented here show that the addition of biochar to potting mix used for growing *E. nitens* seedlings did not stimulate seedling growth when measured as plant height or biomass accumulation. As would be expected, seedling height was strongly dependent on fertilizer. Different types of biochar have been reported to bring positive agronomic response in various crops (Van Zwieten *et al*. 2010, Yamato *et al*. 2006, Major *et al*. 2010, Zhang *et al*. 2012). The mechanisms by which biochar renders edaphic conditions more favorable for plant growth are still being discussed. Potential explanations for biochar's positive effect include its ability to a) raise soil pH, b) increase soil porosity and therefore increase water holding capacity of the soil, and c) to stimulate nutrient exchange processes (Chen *et al*. 2010). Early in the experiment, and prior to the appearance of differences in growth rate, biochar application at high rates increased the moisture content of the *Eucalyptus* leaves while at all rates its application reduced soil moisture content. This effect, however, was noticed only at 135 DAP. Leaf water content was found to reach higher levels in seedlings under full fertilizer treatments (data not presented) which might be explained by greater specific leaf area in F100 seedlings with the increased growth generated by increased cell volume per unit of dry matter assimilated into cell walls. Levels of biochar have been found to significantly affect soil water content but as shown in Fig. 4 we found no clear trend between biochar level and soil water content. This might be because the watering regime did not allow for significant water deficits to occur. Soil under B0 treatment accumulated significantly more water than any other treatment. This effect might be explained by pore size distribution enabling soil to hold more water in comparison to soil amended with biochar, however, it was only noticeable until 135 DAP. The lack of biochar effect on seedling growth might be the result of the soil mix used for the pot experiment. The soil, a mix of sand, bark and peat moss had adequate levels of organic matter, and was well aerated. Thus the addition of biochar might not have improved the desirable physiochemical features of the soil in a longer term, explaining the lack of a positive effect which was reported for some poor quality soils (Chan *et al.* 2007, Van Zwieten *et al.* 2010). Another possible explanation might be that the availability of water was not limited in this experiment, therefore biochar benefits related to increased water holding capacity (Van Zwieten *et al.* 2010, Atkinson *et al.* 2010) were not observed. Results show that seedling growth was strongly influenced by fertilizer levels. Biochar used in this study was produced from macadamia nut shells and, being a typical wood-based biochar, did not contain significant amounts of nutrients and therefore could not serve as a nutrient source for the seedlings. Given that Eucalypt seedlings showed a strong dependence on fertilizer rates, amending soil/potting mix used for growing seedlings with biochar containing more nutrients (e.g. additive of chicken litter) could bring interesting results (Chan *et al.* 2008). The positive effect of biochar on soil condition and plant growth is sometimes related to the effect of biochar aging in soil (Nguyen *et al.* 2009, Atkinson *et al.* 2010). As biochar ages in the soil its overall chemical and physical characteristics are modified, including nutrient availability and bulk density. The total time of the experiment (nine months) provided little time to allow significant changes in the soil as a result of biochar application.

The results indicate that the early growth of *E. nitens* seedlings is strongly dependent on fertilizer level and macadamia biochar added to soil does not bring positive effects on seedling height or biomass in a controlled environment under adequate irrigation. Seedlings did exhibit higher leaf moisture contents when higher rates of biochar were applied, which might be beneficial in water-limited environments such as new plantings in the field. However, the effect was transient. In this situation, where competition for soil water might be expected, biochar may provide seedlings with a competitive advantage. Despite the limited agronomic response to fertiliser application in a nursery setting, the benefits of biochar connected to carbon sequestration in the soil should not be ignored (Nguyen et al. 2009). More detailed studies, including the dynamics of chemical compounds in soil and plant material are required to investigate the effect of biochar on water-related issues in soil and plant material in Eucalypt plantations.

### Acknowledgements

The authors would like to thank Forestry Tasmania, BHP Billiton TEMCO Community Foundation, Centre for Renewable Energy and Power Systems (CREPS) and Rainbow Bee Eater Project for their support.

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# The role of compost in increasing soil carbon stocks in in the Sydney Basin, NSW

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### Abstract

Applying carbon-rich organic amendments to soil can enhance soil structure, increase productivity and increase soil carbon sequestration. Farming soils within the Hawkesbury-Nepean Catchment are characterised by low organic carbon concentrations, poor structure and problematic levels of available phosphorus and sulphur, a result of excessive and repeated fertiliser applications. The elevated nutrients pose a threat to the water quality of the Hawkesbury-Nepean River system through nutrient runoff. This research applied commercial compost to sites covering over 400 ha of agricultural land (grazing, orchards, vegetable production and turf) and documents changes in organic carbon and nitrogen, phosphorus and sulphur concentrations in soil. Sixty two sites in the Sydney Basin were sampled pre-application and 12 months post-compost application. Our results demonstrate increases in the concentration of organic carbon in the surface 0.10 m in 54 out of the 62 sites treated with 40 t/ha (dry weight) of compost. Significant increases in organic carbon and total nitrogen concentration ($P<0.05$) occurred without significant increases in either available phosphorus or sulphur. Sites where the concentration of carbon in soil decreased by 7 to 20% were under intensive vegetable production. This research demonstrates that compost can be applied to increase carbon sequestration and without increasing the concentration of nutrients such as phosphorus which may pose environmental risks.

### Key Words

Organic amendments, carbon sequestration, annual horticulture

### Introduction

Applying carbon-rich organic amendments can enhance soil structure, increase agricultural productivity and increase soil resilience. The use of well-decomposed, mature compost supplies both labile and stable carbon (C) fractions to the soil and thus can increase the sequestration of C in soil. Compost application may be the most effective method for increasing C concentrations in soil (Binner *et al*., 2008), especially those under intensive horticultural production. Compost has the additional benefit of not only supplying stable, more permanent C; but also supplying active forms of C that are critically important to maintain essential soil biological functions and ultimately soil health.

By nature Australian soils are challenging and the soils of the Hawkesbury Nepean catchment reflect this; with inherently low fertility and low organic carbon concentration. Farming in this region has been continuous and intensive for over 200 years with farming practices varying with changing technology; the soils are degraded and thus would benefit from applications of organic matter (OM) to improve soil fertility, soil structure and productivity.

Organic C is removed from the soil through decomposition of OM by micro-organisms which is exacerbated by regular cultivation, soil erosion and product removal. To increase the amount of carbon in soil, the carbon inputs need to be greater than the losses. Soil is sequestering C when there is a net gain in the mass of C in soil. Commercial composts are a potential source of C for agricultural soils. The use of compost as a part of a systems-approach to improved soil management would provide an external source of permanent organic carbon and has the potential to reverse some of the C losses in NSW soils.

This paper details results of compost application to 62 sites across a range of agricultural systems in the Hawkesbury Nepean catchment and the impact on C concentration in soil. Currently, inputs in annual horticultural systems and turf farms are largely derived from the application of poultry litter. This has resulted in soils with considerably elevated levels of available nitrogen, phosphorus and sulphur. These nutrients are found in agricultural runoff and pose a threat to the water quality of the Hawkesbury-Nepean River. The project was oriented to shifting farmers towards the use of mature composts as an alternative to excessive use of poultry litter without exacerbating elevated available P or S levels in soil. This paper also reports on more detailed replicated trials at seven sites where changes in C stocks (Mg C $ha^{-1}$) are reported for particular farming systems.

## Methods

### *Site location*

This research was conducted in the Hawkesbury- Nepean River catchment in western Sydney, NSW as a component of the Nutrient Smart Farms project conducted by NSW DPI in 2009-2011. Sampling was conducted on 115 sites. 90 of which were vegetable, 7 turf, 10 dairy/grazing and 8 orchards totalling 407 ha. Only the repeat sampling conducted on 62 sites at 12 months is included in the analysis. Major soil types included Chromosols, Brown Kurosols, Tenosols and Grey Sodosols (Isbell, 2002).

### *Sampling and analytical methods*

Composite samples for chemical analysis were made from 25 30 mm cores at 0-10 cm depth. Soil samples were prepared for chemical analysis as described by Rayment and Higginson (1992; Method 1B1). Soil organic carbon (SOC) was determined on all samples (0-0.10 m) using the acid wet oxidation method (Rayment and Higginson, 1992; Walkley, 1947). Results for SOC are reported as g/100g.

Colwell phosphorus (P) and extractable sulfur (S) were determined (Rayment and Higginson, 1992; Method 9B1, Rayment and Lyons, 2011; Method 10D1). Results for P (Colwell) and S ($KCl_{40}$) are reported as mg/kg. Total nitrogen was determined for all samples by Method 7A2b (i.e. semi-micro Kjeldahl - automated colour) and total phosphorus by Method 9A3a (i.e. Kjeldahl -automated colour). Units are % P or % N on an oven dried basis.

Results for this paper are also reported as carbon stock in Mg C ha-1 calculated by:

Carbon stock (Mg C $ha^{-1}$) = Carbon concentration (g/100g) x bulk density (g/$cm^3$) x depth (cm). An average bulk density (BD) per site was calculated from four samples (McKenzie *et al*., 2002) as described by Dane and Topp (2002). Results were calculated as BD in Mg/$m^3$ on an oven-dry basis to the nearest 0.01 Mg/$m^3$.

### *Statistical analysis*

Statistical analyses were performed using GENSTAT v.8 (VSN International Ltd, UK) software. Differences at $P$=0.05 between means pre- and post-compost applications for major nutrients for comparison of sites were assessed using one- and two-way ANOVA.

### *Compost characterisation*

Australian Standard AS4454-2003 for compliant greenwaste compost was used in this trial. Table 1 provides chemical and physical characterisation of the compost. Compost was incorporated on the vegetable and turf production sites and surface applied in grazing and orchard sites.

**Table 1: Chemical and physical characteristics of compost.**

| Analyte | Range | Analyte | Range |
|---|---|---|---|
| Grading mm | <12 | C:N ratio | 33-55 |
| pH | 7.4-7.8 | Glass, metal, rigid plastics >2mm % w/v | 0.3-0.3 |
| EC dS/m | 2.30-5.54 | Plastic-light, flexible or film >5 mm % w/v | nil found |
| Moisture % dry wt. | 44-56 | Stones, lumps of clay >5mm % w/v | 4.7 |
| Total nitrogen (LECO) % | 0.92-1.5 | Toxicity mm | 55- 78 |
| NO3 + NH4 mg/L | <1-<5 | Arsenic mg/kg | <1.0 -7.6 |
| Total phosphorus % | 0.28 | Cadmium mg/kg | <0.5-<1.0 |
| Soluble phosphorus mg/L | 4.6-4.7 | Chromium mg/kg | 12-19 |
| Potassium mg/kg | 7226-9087 | Copper mg/kg | 41-55 |
| Calcium mg/kg | 3367-3607 | Lead mg/kg | 53-67 |
| Magnesium mg/kg | 1347-1367 | Mercury mg/kg | <0.2-<1.0 |
| Sulphur mg/kg | 40-53 | Nickel mg/kg | 5.5-15 |
| Manganese mg/kg | 260 | zinc mg/kg | 160-190 |
| Organic matter % | 48-54 | Organic contaminants | <0.02 |

**Results and Discussion**

Pre- and post sampling was conducted on sites where compost was applied at 40 t/ha. Post sampling was conducted 12 months after compost application. There was a significant increase in the concentration of C in soil at most (54 out of 62) sites following the application of compost 12 months before. The number of sites with C concentrations of more than 3 g/100 g increased following compost application (Figure 1). Significant increases in C and N concentrations were demonstrated (Table 2). The increases in N are larger than can be attributed directly to the N content of compost. We propose that is in response to the increased microbial activity stimulated by the addition of organic matter, increasing the fixation of $N_2$.

**Table 2: Range of values pre- and post- compost application**.

| | Pre-compost Mean (range) | Post-compost Mean (range) | Significant (P <0.05) |
|---|---|---|---|
| C (g/100g) | 1.94 (0.27-5.6) | 2.54 (0.53-9.3) | Y |
| TN (g/100g) | 0.18 (0.04-0.41) | 0.24 (0.04-0.72) | Y |
| TP (g/100g) | 0.19 (0.03-0.66) | 0.22 (0.04-0.90) | N |
| Colwell P (mg/kg) | 417 (17-1200) | 408 (24-1300) | N |
| S (mg/kg) | 53.1 (2.7-460) | 44.6 (1.3-340) | N |

Importantly for the health and water quality of the Hawkesbury-Nepean River applications of compost did not significantly increase total P, extractable P or extractable S in soil. Chan *et al*. (2008) warned that compost application rates based on the N requirements of crops may supply excessive amounts of P and therefore be an environmental risk. Soil surveys prior to compost application indicated high background levels of P in some of the soils, particularly the intensive production (horticulture and turf) soils. This presented a risk of exacerbating an already over-loaded P system. For these reasons, this project considered the average N and P content of the compost selected and chose the conservative application rate of 40 t/ha which would supply N at a rate well below that recommended for most vegetable crops (NSW Agriculture 1997) and supply lower amounts of P compared with industry practice (poultry litter and mineral fertiliser). Soil tests 12 months after compost application support this application rate as there was no significant increase in extractable P or S.

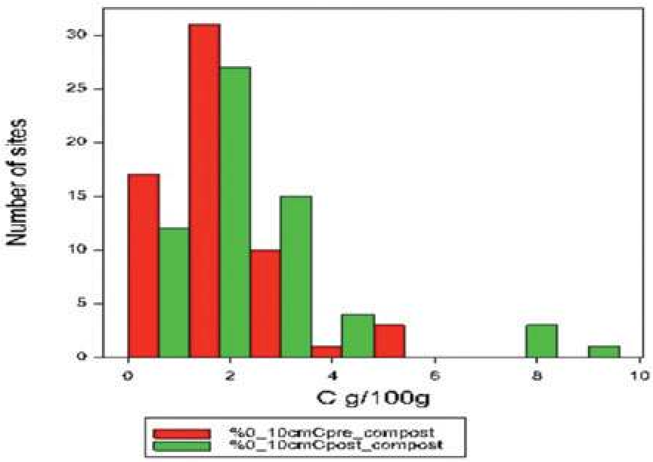


**Figure 1: Frequency distribution of carbon (0-10 cm) at sites before amendment with compost (red) and 12 months after compost application (green).**

**Conclusion**

This research demonstrates that the application of high C:N composts can positively impact accumulation of C stores in soils. The beneficial impact of compost on C stores has been largely attributed to the contribution of stable, well-decomposed (i.e. highly humified) organic mater and also its effect of increasing the activity of soil microbes and the consequential conversion of C to humic substances. Intensive agricultural systems such as vegetable production and cropping often involve high inputs of mineral N fertiliser and low inputs of organic matter. This research demonstrates that high C:N composts such as source separated AS4454 compliant green waste compost, if applied at an optimal rate, are able to stimulate the biological activity and increase the conversion of labile C in compost to more permanent C fractions and provide valuable N. It is concluded that significant increases in C can be achieved without necessarily significantly increasing bioavailable P and S. Future research is required to monitor the permanence of the C sequestered through compost application at these sites.

**Acknowledgements**

This research was part of the Nutrient Smart Farms project. –The project is funded through the Australian Governments Water for the Future Program and is part of the Hawkesbury Nepean River Recovery Program. Nutrient Smart Farms is a joint initiative between the NSW Department of Primary Industries (NSW DPI) and the Hawkesbury-Nepean Catchment Management Authority.

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# An isotopic study to determine the stability of biochar carbon in Australian agricultural soils

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Biochar carbon (C) sequestration in soils may potentially play an important role in reducing CO2 emission. However, to fully evaluate the capacity of soils to store biochar C, an understanding of the mechanisms for biochar C stabilisation in different soils is required. We used a depleted-13C labelling approach to assess the influence of pyrolysis temperature, soil types and soil temperature on biochar C mineralisation. Two wood biochars (450 and 550°C; *Eucalyptus saligna*; d13C ~-36‰) were incubated with four soils (Tenosol, Calcarosol, Ferrosol, Vertosol) at 20, 40 and 60°C.

After one year, we observed that 0.3-7.1% of the added biochar C was converted to CO2. The biochar C mineralisation was 1.5-3.0 times higher for the 450 °C biochar than the 550 °C biochar. The biochar C mineralisation increased significantly with increasing incubation temperature, with 0.3-1.1%, 1.0-2.7%, and 1.0-7.1% of added biochar C was mineralised at 20, 40 and 60 °C, respectively. Biochar C mineralisation rate was high initially, then decreased exponentially with time. The biochar C mineralisation was not significantly different among the soils at 20 °C. At 40 °C, the mineralisation of 450 °C biochar was the highest in the Vertosol, and was significantly higher than observed in the Ferrosol. The C mineralisation of 550 °C biochar at 40 °C was similar among the four soils. At 60 °C, both biochars had the highest biochar C mineralisation in the Vertosol and the lowest in the Ferrosol. The biochar application in the Tenosol caused positive priming of native soil C at all incubation temperatures; whereas in Vertosol, Ferrosol and Calcarosol with higher clay content, biochar presence suppressed the mineralisation of native soil C over time. The C mineralisation and stabilisation data suggested that biochar may be a good option for long-term storage of biomass C in agricultural soils.

# Pesticides management and biochar: Looking beyond soil carbon sequestration and greenhouse gas mitigation

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Biochar is increasingly being recognised by scientists and policy makers around the globe for its potential role in carbon sequestration, reducing greenhouse gas emissions, renewable energy, waste mitigation, and as a soil amendment. Consequently, biochar as a technology may have a role to play in climate change mitigation strategy. While agronomic benefits of biochar amendment to soil are increasingly being recognised, the potential implications (positive or negative) of this practice on pesticide management are less understood. We evaluated wheat straw biochar for its ability to influence bioavailability and persistence of two commonly used herbicides (atrazine and trifluralin) with different mode of actions in two contrasting soils. The biochar was added to soils at 0, 0.5 and 1.0% (w/w) and the herbicides (atrazine and trifluralin) were applied to those soil-biochar mixes at 0, 0.5, 1, 2 and 4 times the recommended dosage. Annual ryegrass (*Lolium rigidum*) was grown in biochar amended soils for one month. The dose-response analysis showed that in the presence of 1% biochar in soil, the dose required to reduce weed biomass by 50% increased by 3.5 times for atrazine, and only by 1.6 times for trifluralin. The combination of the chemical properties and the mode of action governed the extent of biochar-induced effect on herbicide efficacy (Nag et al. 2011)[1]. This presentation would include CSIRO's recent research on pesticide interactions with biochar in soils and especially on bioavailability of pesticides including herbicides and potential implications for pest management and weed control. The consequences of biochar application in soil on weed/pest management as well as management of pesticide residues in soil would be discussed and some of the unanswered questions would be identified.

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# Fresh pasture litter-fall during grazing induces nitrous oxide emissions

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Our previous study showed that significant quantities of litter-fall (harvested but unconsumed plant material dropped during grazing) can be deposited onto the soil surface during a grazing event. However, the contribution of *in situ* decomposition of this litter-fall to nitrous oxide ($N_2O$) emissions is unknown. This current study reports on soil inorganic N dynamics and $N_2O$ emissions following $^{15}N$-labelled ryegrass litter placement on the surface of a pastoral soil in litterbags. Approximately 70% of the total $N_2O$ originated from the surface-applied litter treatment with 38–75% of this occurring within 4–10 d of treatment application. After 66 d, dry matter loss from the litterbags equated to 46–82% of the pasture dry matter applied. Emissions of $N_2O$ likely resulted from ammonification followed by a coupling of nitrification and denitrification during litter decomposition. The litter contributed to both the $^{15}N$ enrichment of the soil $NO_3^-$–N and $N_2O$–N pools. The emission factor (EF) of the *in situ* placed litter was 1.2 ± 0.2%; similar to the IPCC default EF value of 1% for crop residues. Further *in situ* studies using different pasture species and litter-fall rates are required to understand the microbial processes responsible for litter-induced $N_2O$ emissions.

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## SOIL FERTILITY AND SOIL CONTAMINANTS

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# The distribution of soil acidity varies with depth between agricultural regions in the SW of Western Australia with implications for management in relation to lime quality and availability

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We examined geo-referenced soil pH data collected from the Western Australian wheatbelt over four years (2009–2012). Soil was sampled and pH analysed (0.01 M $CaCl_2$) from 0–10 cm (n=53 173), 10–20 cm (n=24 451) and 20–30 cm (n=18 149).

The proportion of topsoil (0–10 cm) samples under target (pH <5.5) varied markedly between the regions (Northern Agricultural (NAR) 52%, Avon 71%, South West (SW) 91% and South Coast (SC) 85%). The proportion of samples under target (pH<4.8) in the10–20 cm layer was relatively similar (NAR 45%, Avon 50%, SW 56% and SC 40%) while in the 20–30 cm layer a lower proportion had pH<4.8 in the South Coast (NAR 50%, Avon 45%, SW 43% and SC 25%). The observed soil pH profiles in the different regions reflect both dominant soil types and dominant farming systems.

Management in the NAR needs to focus on treating acidity at depth, the availability of high quality fine lime products (average NV 90%, fineness 95%<0.5 mm) and relatively short transport distance will assist this goal. Liming is progressively more costly as distance to coastal lime sources increases, e.g. Avon in the central WA wheatbelt, and as the quality of lime decreases further south (average NV 71% and fineness 57%<0.5mm).

Information needs should be targeted towards understanding of the soil pH profile in all regions and calculation of the appropriate application rates with respect to deeper acidity in the north and lime quality in the south.

## Improving the prediction of plant-stress factors, including salinity, nutrient deficiency, and metal toxicity

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Total concentrations of ions (including nutrients, toxic metals, or salts in saline soils) in soils or in soil solutions are poor indicators of environmental impact, other soil properties (e.g., pH) affecting the interpretation of ion effects. Indeed, when considering interactions between plants and ions, plant performance has been almost exclusively related to the concentration or activity of the ion in the bulk soil solution. However, these relationships are often poor restricting our ability to accurately predict plant performance. The root plasma membrane (PM) surface carries negative charges, thus creating a negative surface potential. The negative charges influence the distribution of ions at the PM surface, increasing the activity of cations but decreasing that of anions. We have evidence that the predicted activity of an ion at the outer surface of the root cell PM is a better measure than the activity in the bulk solution in systems that include salinity, nutrient deficiency, and metal toxicity. The data presented here regarding the role of the electrostatic properties of the PM question commonly-held assumptions regarding plant-ion interactions. Additional studies are required to confirm the benefits of this approach.

# How does placement influence the efficacy of zinc oxide and zinc sulfate fertilisers?

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Zinc (Zn) is an important nutrient in cropping systems for both improved productivity and nutritional value of the grain harvest. The usefulness of a Zn compound as a fertiliser will depend on its solubility, bioavailability and placement in the soil profile. Various sources of Zn fertiliser were characterised for nutrient content, morphology and solubility. In glasshouse experiments we measured the plant response to, and the plant recovery of, Zn fertilisers in oxide and sulfate form using three placement treatments - banded, mixed throughout the soil volume, and applied to the soil surface. In addition we measured the plant recovery of Zn from pure Zn phosphates and carbonates, which are likely reaction products of banded Zn fertilisers in soil. The recovery of Zn from Zn phosphates, carbonates and oxides was lower than that of Zn sulfates when banded. Placement was the major factor controlling plant responsiveness to Zn. Zinc oxide fertilisers had very low water solubility and slow dissolution rates in water, which is related to the high pH when the oxide is dissolved in water. There was significantly less recovery of fertiliser Zn by plants from Zn oxides than from Zn sulfates when the fertiliser was banded near the seed, likely due to higher pH in the band as the oxide dissolved. However, the Zn supply to plants was the same for both sources of Zn when fertilisers were mixed through the soil, as the soil likely buffered any dissolution-induced pH increases. This means that Zn oxides, despite their low water solubility, can be effective fertilisers if uniformly mixed throughout the crop rooting depth but Zn sulfate is preferable for banding or surface application.

Acknowledgements; Thanks to the International Zinc Association for funding this work. Thanks also to Caroline Johnston and Maria Manjarrez for technical assistance.

# A slow release boron phosphate to mitigate boron deficiency in high rainfall environments

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Agricultural lands in high-rainfall areas can be deficient in boron (B). Currently, sodium borates (borax) are the most commonly used B fertiliser source. However, sodium borates are highly soluble and, as a result, large leaching losses often occur. We investigated the use of boron phosphate as a slow-release B source. Boron phosphate compounds were synthesized by mixing boric acid and phosphoric acid (1:1 molar ratio) and heating at temperatures of 25 to 1000 °C for 1 or 24 h. X-ray diffraction patterns and chemical analysis results showed formation of boron phosphate compounds at low temperature. These compounds gradually crystallized with increasing temperature and heating time. The compounds synthesized at 300 °C or less were hygroscopic, while those synthesized at 500 to 1000 °C were non-hygroscopic and free-flowing. The solubility of all compounds decreased with increasing synthesis temperature. The solubility of compounds synthesized at 500 and 800 oC increased with increasing pH. The compound synthesized at 1000 °C was nearly insoluble and the solubility was not pH dependent. The release of B from the boron phosphate synthesized at 800 °C for 1 hour was approximately 70% after 150 days, which confirmed its slow release. The characteristics of the compounds synthesized at temperatures 500 and 800 °C showed that it can be used as raw materials for co-granulation with macronutrient fertilisers such as mono-ammonium phosphate. Such co-granulated products are currently being tested.

# Soil modification effects on clay, CEC and organic carbon content using delving and spading on a sandy texture-contrast soil in the South Australian Mallee

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### Abstract

The landscape of the South Australian Mallee region is dominated by dunes and swales, with the most frequent soil types being sands and sandy loams overlying clays at variable depth. These sandy topsoils are frequently low in organic carbon, fertility and water holding capacity, and may be water-repellent (McCord 1995). Adding clay to these soils may increase organic carbon holding potential, increase water infiltration and water holding capacity, and increase cation exchange capacity (CEC), leading to increased nutrient availability and efficiency for crops and pastures.

Methods used to add clay to sandy soils include clay sourced from a pit and spread onto the soil surface, delving clay from below the sand layer up to the surface using a heavy tined implement, and spading clay either from below the sandy layer or from clay spread onto the surface (Bailey *et al.* 2010; Davenport *et al.* 2011) . The deep tillage inherent in delving and spading can also have beneficial effects from breaking up hard pans and mixing topsoil into deeper layers in the soil that may improve root growth.

This experiment examined the effects that delving and spading had on the properties of a typical shallow sand over clay soil in the South Australian Mallee. After treatment the soil profile was completely changed from that of the original soil. Soil texture was altered significantly by the mixing of clay through the sandy topsoil, with a significant increase in CEC. Organic carbon and nutrient levels changed through the soil profile. Differences were also apparent between the delving and spading treatments.

### Key Words

Soil modification, subsoil, clay, sand, delving, spading, soil texture, cation exchange capacity, organic carbon, carbon sequestration potential

### Materials and Methods

The trial was established near Karoonda, South Australia (-35.045246 S,140.090618 E). Soil type was a coarse, sandy topsoil with a bleached sandy A2 horizon to approximately 30 cm, changing abruptly to a sandy clay B horizon. The Australian soil type was a Bleached-Mottled Calcic Brown Chromosol; medium, non-gravelly, sandy/clay loamy, deep (Isbell 2002). The paddock has a history of cereal crop/legume pasture rotations.

The experiment was a randomised block design with 4 treatments (control, delved only, spaded only, and delved + spaded) and 3 replicates. Each plot measured 12m by 100m, with buffers between treatments and replicates. Delving for the delve only and delve + spade treatments was undertaken on Feb 24, 2010. The delved treatment used a purpose built delving machine with four delver tines at 90cm spacing, with the delving depth adjusted to approximately 50 cm (working depth varies with the clay depth). Spading for the spade only and delve + spade treatments was undertaken on the 26$^{th}$ March 2010. The spaded treatments used a 4m wide Farmax spading machine, working to approximately 30-40 cm. The site was then planted to a wheat crop as per district practice and the crop monitored throughout the season.

Soil samples were taken in March 2011 (1 year after modification) using a trailer-mounted soil corer. There were 10 sample sites per plot, randomly located, with the cores divided into increments of 0-10, 10-20, 20-30 and 30-40 cm. In the delved treatments, 10 sampling locations were randomly allocated in proportion to the area modified by delving (30%) and in between the delve lines (70%), designated delve-RP. An additional 10 samples each were taken from along the delve lines (delve-IN) and between the delve lines (delve-OUT). These samples were analysed separately to compare the differences between the results from proportional sampling and targeted sampling in the delved plots. This procedure could not be used in the delve + spade plots due to difficulty locating the delve lines.

Samples were bulked from each plot and analysed for clay, silt and sand content (particle size analysis), organic carbon (Walkley-Black), pH, phosphorus (Colwell), potassium (Colwell), exchangeable cations, carbonates, sodium, chloride and boron. A split-plot analysis of variance was used, comparing the control, spaded, proportionately sampled delved and delve + spade plots. The analysis was re-run using the delve-IN (in delve lines) and delve-OUT (out of delve lines) samples to compare the results.

## Results and Discussion

*Clay content*

Clay and sand distribution through the soil profile was significantly altered through soil modification. When the results were analysed using the proportionately sampled soils from the delved plots, the analysis showed a significant interaction between spading and depth, and between delving and spading. In the control plots, clay content was low in the 0-10 and 10-20 cm layers (6-7 %), and increased at 20-30 cm (20.3%) and 30-40 cm (26.4%) (see Table 1). In the spaded plots, clay content was higher through the 0-10 and 10-20 cm layers, demonstrating that spading had distributed clay from the working depth (30cm) throughout the sandy soil. Delve-RP increased mean clay content in the 0-20 cm layers compared to the control, but had little effect below 20cm. Spading and delving + spading were similar to each other. When the analysis was re-run substituting the delve-OUT samples for the delve-RP, similar results were obtained except that there was no delve by spade interaction.

In contrast, when the analysis was run using the delve-IN samples, there was a clear 3-way interaction between soil depth, delving and spading (see Table 1). Clay content increased significantly in the 0-10 and 10-20 cm layers when any soil modification was present. There was a decrease in clay content in the delve + spade 20-30 cm treatment (probably due to mixing of sand from the upper layers), but otherwise there tended to be less changes in clay content in the 20-30 and 30-40 cm layers.

**Table 1: Clay content (%) of modified soils (all means).**

| Depth (cm) | Control | Spade | Delve-In | Delve-Out | Delve-RP | Delve+Spade |
|---|---|---|---|---|---|---|
| 0-10 | 6.3 | 13.1 | 12.5 | 7.0 | 8.7 | 15.0 |
| 10-20 | 7.4 | 13.2 | 22.4 | 9.3 | 15.5 | 12.9 |
| 20-30 | 20.3 | 19.7 | 22.7 | 22.0 | 25.7 | 15.9 |
| 30-40 | 26.4 | 24.9 | 25.1 | 29.8 | 27.8 | 25.7 |

*LSD (P=0.05) value is 5.47 when comparing values with Delve-IN, except when comparing values with the same soil modification treatments, when LSD = 5.22. LSD value is 5.78 when comparing values with Delve-OUT, except when comparing the same levels of soil modification (6.03). LSD value is 5.60 when comparing values with Delve-RP, except when comparing the same levels of soil modification (5.76).*

The Delve-IN and Delve-OUT measurements represent the effects of the soil modification within the specific areas in the delve lines and between the delve lines.

A number of soil characteristics measured correlated strongly with clay content. Sand content was strongly and negatively correlated with clay % ($R^2$ = -0.99) as expected. Silt was minimal in this soil. Other attributes with high positive correlation with clay content were CEC ($R^2$ = 0.93), boron ($R^2$ = 0.91), potassium (Colwell) ($R^2$ = 0.82), electrical conductivity ($R^2$ = 0.77), and pH ($CaCl_2$) ($R^2$ = 0.75).

The increase in clay content in the 0-10 and 10-20 cm layers of the soil has important implications for soil use. Water holding capacity, infiltration rate and potential water use efficiency of crops/pastures will be affected. Erosion risk will be reduced as the texture class changes from sand to sandy loam or loam.

*Cation exchange capacity*

CEC levels showed a 2-way interaction between depth and spade and a separate single effect of delving when the data were analysed using the proportionately sampled delve plots. Overall, CEC followed the clay content of the soil, with CEC low in the sandier 0-10 and 10-20 cm, and higher in the 20-30 and 30-40 cm where more clay was present. Spading increased CEC in the 0-20 cm layers by mixing clay from the lower layers up through the sandy topsoil. Delving increased CEC throughout (see Table 2). The effects of the different measurements in the delved plots can also be seen in Table 2.

The changes in CEC that can occur through the use of these technologies may have implications for nutrient management in crops and pastures. Elevating the CEC of sands in the 0-10 and 10-20 layers could be expected to enhance nutrient holding capacity with potential for increased crop/pasture production.

**Table 2: Cation Exchange Capacity (cmol/kg) of modified soils (all means).**

| Depth (cm) | Control | Spade | Delve-In | Delve-Out | Delve-RP | Delve+Spade |
|---|---|---|---|---|---|---|
| 0-10 | 3.55 | 5.92 | 5.98 | 3.79 | 4.12 | 8.21 |
| 10-20 | 3.36 | 4.91 | 11.33 | 3.64 | 7.69 | 7.74 |
| 20-30 | 8.64 | 8.76 | 12.32 | 8.47 | 11.95 | 8.55 |
| 30-40 | 13.89 | 13.42 | 13.56 | 14.56 | 15.15 | 13.31 |

*LSD (P=0.05) value is 2.85 when comparing values with Delve-IN, except when comparing same levels of soil modification (2.67). LSD value is 2.85 when comparing values with Delve-OUT, except when comparing same levels of soil modification (2.62). LSD value is 3.24 when comparing values with Delve-RP, except when comparing same levels of soil modification (2.94).*

*Organic Carbon*

When analysed using the proportional samples from the delved plots, there was a significant interaction between delving and depth, and between spading and depth. When analysed using samples taken from in delve lines or out delve lines, OC content showed a 3-way interaction between delving, spading and sample depth. The means are shown below in Table 3.

OC content dropped well below the control level in the 0-10 cm layer in the modified soils, but was increased significantly in the 10-20, 20-30 and 30-40 cm layers (refer Tables 3 and 4). This appears related to the degree of mixing of the A1 and A2 soil layers, with the largest decrease in spaded soil. This is probably due to the strong mixing action of the spader which dilutes the organic carbon from the top 0-10 cm through the deeper soil layers. The 10-20, 20-30 and 30-40 cm layers increased in OC, whether delved or spaded (see Tables 3 and 4). OC concentration decreased with depth in all treatments but soil modification reduced the amount by which it was decreased compared to the control (Table 5).

**Table 3: Mean OC concentration (% of soil sample) for all treatments.**

| Depth (cm) | Control | Spade | Delve-In | Delve-Out | Delve-RP | Delve+Spade |
|---|---|---|---|---|---|---|
| 0-10 | 0.54 | 0.37 | 0.42 | 0.38 | 0.45 | 0.32 |
| 10-20 | 0.21 | 0.28 | 0.25 | 0.22 | 0.25 | 0.30 |
| 20-30 | 0.18 | 0.20 | 0.25 | 0.20 | 0.19 | 0.21 |
| 30-40 | 0.15 | 0.17 | 0.20 | 0.20 | 0.18 | 0.18 |

*LSD (P=0.05) value is 0.049 when comparing values with Delve-IN, except when comparing same levels of soil modification (0.048). LSD value is 0.056 when comparing values with Delve-OUT, except when comparing same levels of soil modification (0.055). LSD value is 0.053 when comparing values with Delve-RP, except when comparing same levels of soil modification (0.047).*

**Table 4: Mean OC content as % of control values at each depth.**

| Depth (cm) | Control | Spade | Delve-In | Delve-Out | Delve-RP | Delve+Spade |
|---|---|---|---|---|---|---|
| 0-10 | 100 | 68 | 78 | 71 | 83 | 59 |
| 10-20 | 100 | 135 | 122 | 108 | 122 | 145 |
| 20-30 | 100 | 113 | 141 | 113 | 107 | 117 |
| 30-40 | 100 | 111 | 133 | 131 | 118 | 118 |

Table 5 shows the OC content as a percentage of the 0-10 cm control value. The majority of the carbon in the unmodified soil is in the 0-10 cm layer with much lower values in the 10-20 cm layer. This is in contrast to the modified soils where the carbon in the 0-10 cm layer appears to have been distributed through the 10-20 and 20-30 cm layers.

**Table 5: Mean OC content as % of 0-10 cm control.**

| Depth (cm) | Control | Spade | Delve-In | Delve-Out | Delve-RP | Delve+Spade |
|---|---|---|---|---|---|---|
| 0-10 | 100 | 68 | 78 | 71 | 83 | 59 |
| 10-20 | 38 | 52 | 47 | 41 | 47 | 55 |
| 20-30 | 33 | 37 | 46 | 37 | 35 | 38 |
| 30-40 | 28 | 31 | 37 | 36 | 33 | 33 |

These measurements show what has happened 1 year after soil modification. Theoretically OC content is positively correlated to soil clay content (Baldock and Skjemstad 1999; Sanderman *et al.* 2010) and therefore higher clay content can result in higher OC levels in soils. An analysis of South Australian soil data by Schapel (2010) has demonstrated this correlation exists across the agricultural soils of South Australia. There is potential for the OC content in clay modified soils to increase further over time as the crop/pasture system stabilises with the new soil. Other measurements taken from this site indicate that root density, dry matter production, and crop yield are significantly increased in the modified plots (Tonkin 2011) due to the changes in water and nutrient availability. This in turn results in higher crop and root residues in the soil, adding to the organic matter in the soil that will increase the potential for carbon retention until a new equilibrium is reached.

## Conclusions

Modifying the sandy topsoil by adding clay using delving, spading or a combination of delving and spading resulted in changes to the soil texture, CEC and OC content. Spading or spading + delving produced the most significant changes, with clay content in the 0-10 and 10-20 cm layers double that of the control. Delving alone changed the soil in a pattern related to the delver tines, with samples taken in the delve line different to those taken between delve lines. CEC increases corresponded to the increases in clay content in the soil.  OC content decreased in the 0-10 cm layer of soil with soil modification, but increased in the 10-20 and 20-30 cm layers, most likely as a result of the mixing process. OC content is likely to increase further in the modified soils over time as the soil reaches a new C equilibrium. Overall, soil modification using delving and/or spading has the potential to alter shallow sandy soils overlying clays to a significant degree. Erosion potential is reduced as a result of the texture change from sand to sandy loam or loam in the topsoil. Water repellence is overcome by the addition of the clay, and water holding capacity increased. The increase in CEC in the infertile sandy topsoil allows greater nutrient holding capacity in the upper soil layers where it is needed by crops and pastures. The changes in clay content, CEC and OC content may allow improved root growth to deeper levels in the soil, improving the capacity of crops and pastures to find nutrition and water, and leading to improved productivity.

## Acknowledgements

The authors would like to acknowledge Mr Nick Wood for allowing this trial to take place on his land, the Karoonda and Districts Agricultural Bureau for project support, the Government of South Australia Department of Environment and Natural Resources for funding support, Woolworths Landcare for funding support, John Field Statistical Consulting for statistical assistance, Mr Tim Dunstan of DUNSTANag for assistance with delving and spading and Mr Roger Groocock for technical advice.

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## THE PHYSICAL SOIL AND SOIL WATER

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# Modeling preferential flow through coarse textured substrates

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In semiarid environments coarse textured substrate is considered as an option for the construction of covers to encapsulate waste materials with a potential environmental risk. Dependent on local climatic characteristics, intensive rain events may lead to not only matrix flow, but also by-pass flow with deep drainage in short periods of time. Modelling of such fluxes when based on the water retention characteristic is challenged by the difficulty to quantify the amount of macro pores against textural pores. Such complex pore systems can be described by a bi-modal water retention curve and for this investigation a modified van Genuchten model was applied. Seepage data from a lysimeter experiment were used to derive a bi-modal water retention curve by inverse modeling. The hydraulic data were compared against an in-situ measurement of the hydraulic conductivity function, which was established for conditions close to saturation using the hood infiltrometer. The results showed a high level of agreement between the two strategies and revealed that preferential flow occurred only to water potentials greater than -0.5 to -0.8 kPa. The presentation shows the pathways for the determination of a bi-modal water retention curve and discusses the importance of the quantification of preferential flow in the context of constructed covers.

# Nutrient management planning on farms within a nitrogen cap and emissions trading regime

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To protect the pristine quality of Lake Taupo, Waikato Regional Council (WRC) have implemented a land and water management plan that caps the nitrogen(N) loads entering the lake, as well as an N trading scheme to bring about load reductions. Estimates of the cumulative inputs from pastoral agriculture, native and exotic forest, and urban areas are 41%, 30% and 2.5% of the total N input load, respectively. The variable loads that can be managed are the urban and pastoral agriculture contributions. Diffuse inputs from pastoral agriculture make up 94% of the manageable load. WRC used a "grandparenting" and benchmarking technique to determine the contribution of each farm's N leaching loss to the manageable load. Overseer Nutrient Budgeting software using a farm systems approach was used to predict each farm's N leaching loss. The benchmark years for grandparenting a farm's N loss were chosen as 2001-2004. The load allocated to a farm (N loss allowance) was predicted for the farm's best year of production in that period. Changes to farm management that bring reductions in N loss per farm generate N credits which can be traded. To generate a reduction in N load to the lake, The Lake Taupo Protection Trust has been funded by government with the aim of purchasing 20% of the manageable load by 2019. This paper presents nutrient management plans for a mixed sheep-beef-forestry farming system that generates N credits and compares the cost of generating those credits with the tradable N value and other income that may be recovered from carbon credits .

# Two types of deep drainage in a Vertosol under irrigated cotton

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## Abstract

Deep drainage under irrigated cropping is a waste of a scarce resource and can lead to rising watertables and salinity. Drainage under a furrow irrigated cotton-wheat rotation on a Grey Vertosol in northern New South Wales was investigated using a variable tension lysimeter at 2.1 m depth. The lysimeter was installed without disturbing the overlying soil and could accurately measure drainage because the collection trays had a vacuum applied that was continuously varied, so as to be equal to the matric potential at 2.1 m depth. Over two years, two distinct types of drainage were measured – matrix and bypass. Matrix drainage occurred during periods where rainfall exceeded ET so that a wetting front moved down the profile wetting successive layers until it passed out of the root zone. The drainage was at relatively low rates (0.5 mm/day) over extended periods (1 month). In contrast, bypass flow occurred immediately after furrow irrigation, with the rate peaking at up to 3.2 mm/day 25 hours after the irrigation front passed overhead. The water content and matric potential of the soil at 2.1 m remained largely unaffected by the irrigation, and the hydraulic gradient was upwards. Bypass drainage accounted for the bulk of drainage during the two year rotation. It not only lowers water use efficiency but also means a higher leaching fraction is required to prevent a build up of salt, because bypass drainage is less efficient at removing salt.

## Introduction

Furrow irrigation is commonly used by the cotton industry in Australia, since it is relatively low cost. For much of the history of the Australian cotton industry it was believed that deep drainage under cotton was insignificant because of the low permeability of the heavy clay soils where cotton is grown. Starting in the 1990s, concern about exactly how much drainage occurs increased, because of non-negligible estimates of drainage under dryland cropping systems in northern New South Wales (NSW) (Ringrose-Voase *et al.* 2003). Drainage was now seen as a waste of an increasingly scarce resource and having the potential to lead to rising watertables and salinity. This paper describes the use of a lysimeter to investigate drainage in detail at one location. The work should be seen as part of a multi-faceted approach to investigating deep drainage using other, less expensive methods (Gunawardena *et al.* 2011, Silburn *et al.* 2011).

## Methods

High frequency measurements of deep drainage were made using the lysimeter facility at the Australian Cotton Research Institute (ACRI) in northern NSW, west of Narrabri (30° 11.53' South, 149° 36.31' East). The lysimeter is located in an experimental plot under a cotton-wheat rotation. Cotton crops are furrow irrigated, but wheat crops only receive supplementary irrigation. Minimum tillage is used with stubble retention and permanent beds. Alternate furrows are used for traffic and irrigation. The plot is 200 m long from head to tail ditch.

The soil is a Haplic, Self-mulching, Grey Vertosol (Isbell, 1996), with 60% clay (<2 μm), 14% silt (2-20 μm) and 25% sand (20-2000 μm) above 1.2 m depth. Below 1.2 m, the clay content decreases to 50% by 2 m depth with corresponding increases in silt and sand to 20% and 30% respectively. Exchangeable sodium percentage increases down the profile from <1% of CEC at the surface to 6.5% of CEC at 2 m.

The variable tension drainage lysimeter at the facility was designed to minimize the effect of the instrument on the drainage flux. It does this by ensuring that the hydraulic gradient above the lysimeter collection trays is the same as in the surrounding soil. Brye *et al*. (1999) designed a similar system, which was improved by Pegler *et al.* (2003) by introducing automated regulation of the vacuum. The ACRI design included further modifications.

The ACRI lysimeter has six collection trays (each 0.91 × 0.29 m area, 0.13 m high) covering a total area of 1.58 $m^2$. The trays are stainless steel boxes with an upper surface of porous, sintered stainless steel, 1 mm thick with a nominal pore size of 0.2 μm. Once saturated with water it can hold water up to a potential of -28 kPa. The floor of each tray slopes to a drain in one corner, and each tray allows connection to a vacuum.

The trays were installed under the root zone of an experimental plot at 2.1 m depth, half way between the head and tail ditches. To prevent disturbance of the soil above the trays, they were inserted by excavating horizontally from a concrete access shaft (4 m deep, 2 m diameter) located just outside the plot. The absence of vertical walls above the trays means there is no interference with the natural shrink/swell behaviour of the soil. The ceiling immediately above the trays was prepared before inserting the trays by peeling away the soil using polyester resin to ensure a natural surface. Silica flour, graded to remove particles less the 15 μm, was packed into the gap above the top of the trays and the ceiling as a contact material.

The drain from each tray is connected to its own collection tank in the access shaft that was weighed automatically every 15 minutes by a load cell. Two vertical arrays of five tensiometers are installed though the wall of the access shaft at depths from 0.9 to 2.1 m to record soil matric potential every 15 minutes. A vacuum is applied to the trays via its internal riser tube. The vacuum is adjusted every 15 minutes by a data logger via solenoids connecting the trays either to a vacuum reservoir kept at -40 kPa or to atmosphere, so that the vacuum equals the average matric potential measured by the two tensiometers at 2.1 m depth. Four neutron probe access tubes were installed to 3 m depth around the lysimeter to allow measurement of soil water content at frequent intervals during the irrigation season.

## Results and Discussion

This paper will concentrate on results from one crop rotation from October 2008 to October 2010. Cotton was sown on 9 October 2008 and harvested on 12 June 2009. The first irrigation was not until 22 December 2008 because there was considerable rainfall (200 mm) falling between sowing and the first irrigation. The crop was irrigated a total of six times and received a total of about 512 mm of irrigation and 463 mm of rainfall. During the crop there were 53.9 mm of drainage. The electrical conductivity (EC) of the drainage water varied from 2.5 to 3.5 dS/m for most of the season, but increased after the fifth irrigation and reached a maximum of 8.1 dS/m after the sixth. The mean EC of the irrigation water was 0.4 dS/m.

Wheat was sown on 23 June 2009 and harvested on 18 November 2009. The crop received two irrigations totalling 240 mm and 109 mm of rainfall. There was only 1.4 mm of drainage during the wheat crop, with an EC of 13.3 dS/m. After wheat harvest, there was fallow for a year until the next cotton crop was sown on 4 November 2010. During the fallow period there was 771 mm of rainfall and 21.6 mm of drainage, with an EC of 6.6 dS/m.

Over the rotation, the drainage rate varied from zero to a maximum of 3.2 mm/day. The greatest rates occurred after irrigations early in the cotton season when the soil water deficit immediately before irrigation was least. The early irrigations also tended to produce the greatest absolute amount of drainage between irrigations. As the cotton season progressed, both the peak rates and the amounts of drainage decreased.

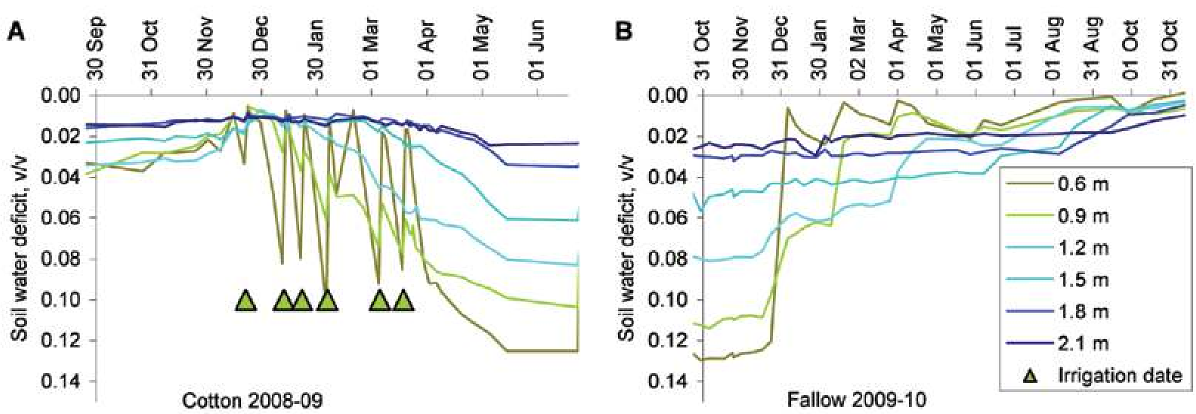


**Figure 1: Soil water deficit (relative to drained upper limit) of subsoil layers from 0.6 – 2.1 m depth during A0 the 2008-09 cotton crop and B) the 2009/10 fallow measured by neutron moisture meter.**

The drainage hydrograph exhibits two types of drainage – matrix and bypass. The former is illustrated by a drainage event starting in September 2010 during the 2009/10 fallow. The previous wheat crop had dried the soil profile creating a soil water deficit of 204 mm to 2.1 m depth (December 2009). There was a steady input of 640 mm of rainfall from December 2009 to August 2010. This caused a wetting front to move downwards through the profile that became more diffuse as it moved downwards and reached 1.8 m in August 2010 (Figure 1B). The drainage rate during this period was very low, 0.03 mm/day. Over this period the hydraulic gradient was upwards, and the matric potentials (-55 and -40 kPa at 1.8 and 2.1 m, respectively) were such that hydraulic conductivity would have been very small. The wetting front reached 1.8 m on 5 September and the matric potential increased rapidly, with the gradient becoming downwards.

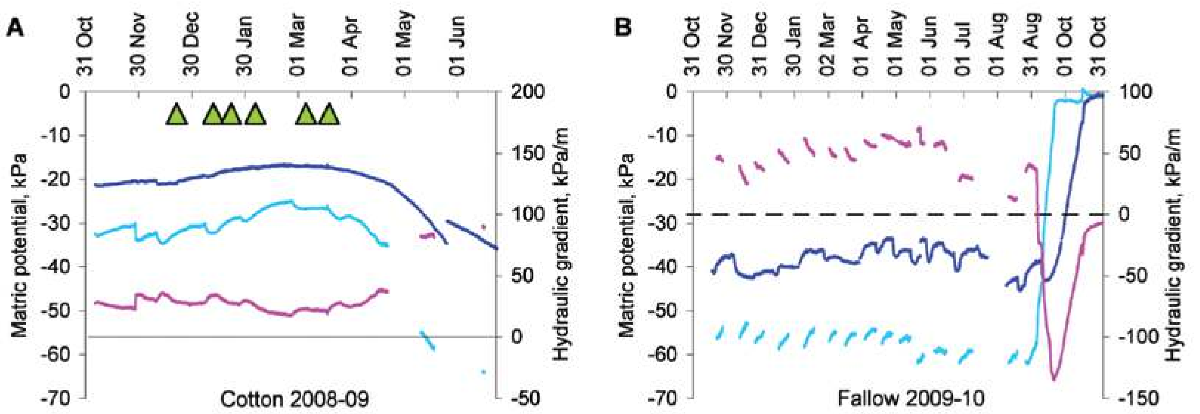


**Figure 2: Mean matric potential at 1.8 and 2.1 m depth during A) the 2008-09 cotton crop and B) the 2009-10 fallow. Also shown is the hydraulic gradient at 1.95 m depth (positive upwards).**

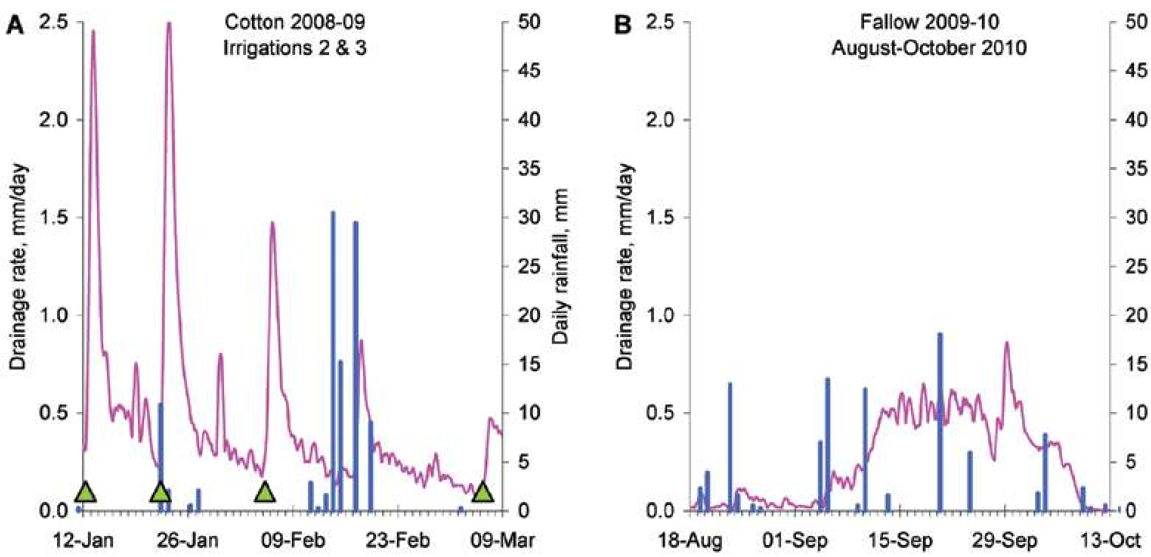


**Figure 3: Drainage rate at 2.1 m depth measured by the lysimeter (left vertical axes) and daily rainfall (right vertical axes) over 8 week periods during A) the 2008-09 cotton crop and B) the 2009-10 fallow.**

The drainage rate at 2.1 m started to increase on 5 September (Figure 3B), and reached a maximum of about 0.5 mm/day on about 15 September roughly corresponding to the maximum downward hydraulic gradient. The rate declined after 5 October corresponding to the time when the potential at 2.1 m was rising most rapidly. By 11 October the rate had returned to very low levels. Overall the wetting front took about a month to pass and accounted for 13.7 mm drainage. This type of drainage event can be considered matrix drainage because it results from a wetting front moving through the soil matrix over a period during which infiltration of water exceeds evapotranspiration, until the soil water holding capacity is exceeded. Further evidence that the drainage moved through the soil matrix is its relatively high EC.

Drainage during the cotton season contrasts with the matrix drainage described above. The subsoil was relatively wet when cotton was sown (Figure 1A), with a soil water deficit to 2.1 m of 135 mm. The 200 mm of rain between sowing and the first irrigation on 22 December only reduced the deficit to 115 mm because much was used by the crop. Irrigation had very little effect on the deficits of layers below 1.2 m. Similarly the matric potential at 1.8 and 2.1 m were largely unaffected by irrigation, with the hydraulic gradient at tray depth being upward for the whole season (Figure 2A). Nonetheless, 54 mm of drainage occurred. Generally the drainage rate started increasing about 6 hours after the irrigation front passed overhead and peaked after about 25 hours (Figure 3A). After peaking the rate declined more slowly than it rose, but still exponentially. The EC of the drainage was 2.5 – 3.5 dS/m – much less than during the fallow. All the above observations suggest that the drainage bypassed the matrix and flowed down macropores. In the upper 0.5 m these macropores are likely to be the surface cracks characteristic of Vertosols, but these cracks do not develop much beyond 1 m depth. One possibility is that bypass flow travels along the slickensides also characteristic of the deep subsoil of Vertosols.

The peak drainage rate after each irrigation declined from 3.2 mm/day after irrigation 1 to 1.5 mm/day after irrigation 4. As the cotton season progressed, larger subsoil deficits developed between irrigations. It is possible that whilst furrow irrigation wets the upper 0.5 m effectively due to the presence of cracks, the bypass pathways through the subsoil are less extensive, so that the amount of water that bypasses through the subsoil is mitigated by the deficit of the subsoil.

## Conclusions

High frequency measurements of drainage allowed different types of drainage to be detected. Whilst matrix drainage due to the passage of a wetting front is the most common type of drainage in most soils, bypass drainage is likely to be important only in situations where a large volume of free water is applied to a surface with extensive macropores, as in furrow irrigation of cracking clay soils. Apart from directly impacting on water use efficiency of irrigated systems, bypass flow is also less effective at removing salt from the profile. Hence the leaching fraction required to prevent soil salinity is greater than situations where only matrix drainage occurs.


## Acknowledgements

This work was funded by the Cotton Research and Development Corporation and the Cotton Catchment Communities Cooperative Research Centre. The authors thank Dr Nilantha Hulugalle and NSW Department of Primary Industries for their cooperation in locating the lysimeter in their experimental field.


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# Estimating the nature, timing, and depth of preferential flow in a texture-contrast soil

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## Abstract

Occurrence of preferential flow in soil is known to increase the risk of off-site agrochemical mobilisation to groundwater and waterways. This study investigated the ability of a commercially available soil moisture probe to determine the timing, depth and magnitude of preferential flow events in a texture-contrast soil. Occurrence of preferential flow was not related to rainfall magnitude nor rainfall intensity, rather preferential flow occurred when antecedent soil moisture was below 70-76 % of total soil moisture. High frequency soil moisture monitoring over a 21 month period was able to determine the timing, depth and wetting front velocity of preferential flow events. However the spatial variability of the preferential flow paths meant that two to three soil moisture probes were required to determine changes in soil moisture or the preferential flow flux.

## Key Words

Macropore flow, finger flow, funnel flow, duplex soil, EnviroSCAN

## Introduction

Preferential flow refers to processes in which infiltrating water by-passes the soil matrix, resulting in more rapid and deeper movement of water and solutes than would otherwise be expected (Hendrickx and Flury 2001). Numerous studies have demonstrated that preferential flow is both common and widespread (Flury *et al.* 1994) resulting in poor irrigation performance (Cullum 2009), reduced agricultural production (Garg *et al.* 2009), and off-site mobilisation of agrochemicals to waterways and groundwater (Jarvis 2007; Simunek *et al.* 2003). Land managers and regulators require the ability to simply and cost effectively determine the occurrence, depth and magnitude of preferential flow events. This knowledge will improve agricultural production through improved irrigation performance, and reduce risk of environmental harm through improved ability to identify inherently 'leaky' soil – landuse combinations that are prone to preferential flow.

Determining the occurrence or magnitude of preferential flows is difficult due to their non-equilibrium processes and high degree of spatial and temporal variability (Allaire *et al.* 2009). A number of laboratory procedures exist for determining the occurrence and magnitude of preferential flow events including solute break through curves, measurement of gas or fluid flow, and X-ray and magnetic resonance imaging (MRI). However these approaches require soil disturbance or excavation, imposition of artificial boundary conditions, or analysis of small sample volumes. A number of *in situ* approaches have been used to infer the presence of preferential flow including weighing lysimeters, flux meters, and buried soil moisture sensors (Allaire *et al.* 2009). However these devices are typically very expensive, impose artificial boundary conditions, and/or require considerable soil disturbance during installation. This manuscript outlines a series of simple, mostly *in situ* approaches for determining the timing, depth and magnitude of preferential flow events using high frequency soil moisture monitoring.

## Methods

Studies were conducted at the University of Tasmania farm, Australia, in a seasonally water repellent, texture-contrast (sandy loam overlying medium clay) Brown Kurosol. Mean annual rainfall at the site is 478 mm, and mean annual evaporation is 1324 mm. Dye staining experiments demonstrated that up to five different forms of preferential flow were responsible for infiltration into dry soils, whilst infiltration into soils at moisture contents near field capacity resulted in the development of unstable wetting fronts and lateral flow in the A1 horizon (Hardie *et al.* 2011).

Timing, depth and wetting front velocity of preferential flow events were determined by high frequency soil moisture monitoring using a capacitance EnviroSCAN Solo probe with six sensors mounted at 10 cm, 20 cm, 30 cm, 50 cm, 70 cm, 90 cm and 130 cm depths. Recordings were taken every 1–10 minutes between 26/6/08 and 10/7/09, and hourly between 19/9/07 and 30/5/08. The EnviroSCAN probe was mounted inside a 56 mm diameter 1.8 meter long PVC plastic access tube, which was installed by vertical ramming and removing soil from within the tube by auger. Subsequent dye staining experiments demonstrated that tube installation had minimal, if any, affect on infiltration flow paths.

Response to rainfall was said to have occurred when the soil moisture sensors had exceeded a 0.002 $m^3m^{-3}$ instrument noise threshold. Wetting front velocity was estimated by dividing the distance between sensors by the time between the sensors first exceeding the 0.002 $m^3m^{-3}$ threshold. Saturated hydraulic conductivity was determined from 100 mm diameter cores using a similar constant head approach (McKenzie *et al.* 2002). Occurrence of preferential flow was determined from presence of non-sequential soil moisture response with depth (PF-ns) and/or wetting front velocities exceeding 500 mm $hr^{-1}$ (PF-rate). Equilibrium flow (EQ) or uniform flow was said to have occurred when the soil moisture sensors responded sequentially with depth and wetting front velocities were less than 100 mm $hr^{-1}$(twice the saturated hydraulic conductivity).

The ability of soil moisture sensors to detect the passage of preferential flow paths was inferred from binary images of infiltration pathways. Infiltration pathways were determined by applying 4 g/L Brilliant Blue dye tracer (C.I. Food Blue 42090) to 25 mm artificial rainfall, to a 1.0 by 1.0 meter area. Dye stained areas were excavated in five vertical profiles, approximately 10 cm apart, 30 hours after irrigation ceased. Binary images of the irrigation infiltration flow paths were produced using Photoshop CS3 and Image J software. The likelihood that the soil moisture sensors would intersect preferential flow paths was determined by comparing the sensor detection area (effectively 10 cm radius) to the binary images of the infiltration pathways in very dry and wet soil conditions. The number of EnviroSCAN soil moisture probes required to accurately determine changes in soil moisture was determined from dye tracer stained infiltration pathways, in which the representative elementary volume of the infiltration pathways was determined from the relationship between the area of soil sampled by different numbers of EnviroSCAN sensors, and variance (CV) in the proportion of soil which participated in flow.

**Results and Discussion**

High frequency soil moisture monitoring (1-10 minutes) was able to determine the occurrence, depth and timing of preferential flow events. Soil moisture response to rainfall between the 23rd June and 28th July 2008 demonstrated that as antecedent soil moisture increased with each rainfall event infiltration switched from being entirely preferential to uniform (Figure 1).

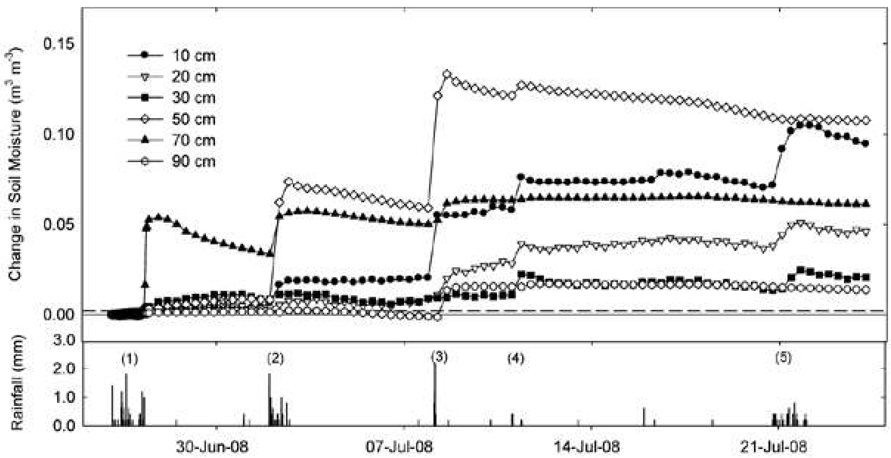


**Figure 1. Change in soil moisture and rainfall between 25th June and 25th July 2008. Numbers (1) to (5) refer to rainfall events. Note dashed line indicates 0.002 $m^3$ $m^{-3}$ soil moisture response threshold.**

In the first three rainfall events non-sequential (PF-ns) response from the soil moisture sensors was attributed to preferential flow routing water to 50 cm and / or 70 cm depth, before sensors higher in the soil profile (30 cm depth) had responded. Dye staining experiments indicated the rapid increase in soil moisture at 70 cm depth during the first rainfall event resulted from accumulation of thin (2-4 mm wide) rivulet flow in terminal shrinkage cracks and voids at the base of the soil profile (Hardie *et al.* 2011). By the fifth rainfall event, increased antecedent soil moisture had largely overcome the effects of water repellence in the topsoil and caused subsoil clays to swell, reducing the presence of shrinkage cracks. Consequently during the fifth rainfall event, infiltration occurred by uniform flow process in which the wetting front moved sequentially with depth at a velocity of only 20 mm $hr^{-1}$ (10 cm to 30 cm depth).

High frequency soil moisture monitoring of 48 rainfall events demonstrated that rainfall magnitude and rainfall intensity did not influence the occurrence of preferential flow, as no significant difference existed between the amount of rainfall, average rainfall intensity or maximum rainfall intensity between the equilibrium infiltration response (EQ) and the two preferential infiltration types (PF-ns & PF-rate) (Figure 2 a&b). Rather, preferential flow mostly occurred when antecedent soil moisture content was below 226 mm cumulative soil moisture (0-70 cm) or 70 % of total soil moisture, and did not occur above 246 mm (0-70 cm) or 76 % of total soil moisture (Figure 2c). Below the antecedent soil moisture threshold preferential flow resulted from water repellence in the topsoil and rivulet flow through shrinkage cracks in the subsoil. Above the soil moisture threshold equilibrium or uniform flow was promoted by the apparent lack of water repellence in the A1 horizon and closure of shrinkage cracks following clay swelling in the subsoil.

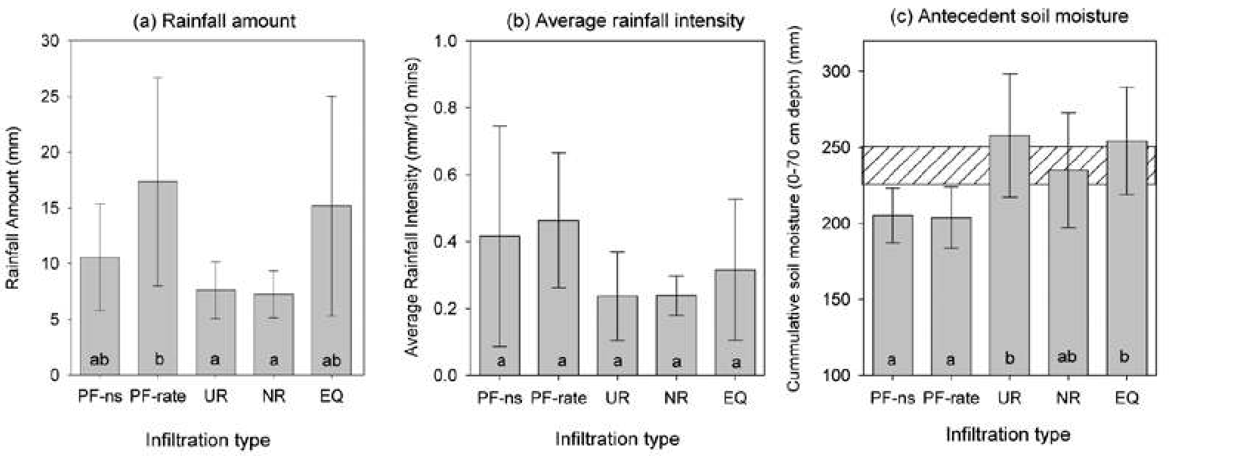


**Figure 2 Effect of (a) rainfall amount, (b) mean rainfall intensity on infiltration response. (c) Effect of antecedent soil moisture on infiltration type. Hatched area indicates threshold below which preferential flow occurs and above which uniform flow occurs. PF-ns preferential flow non sequential, PF-rate preferential flow excessive infiltration rate, UR unknown response, NR no response, EQ equilibrium flow. Error bars represent ±1 SD, significant (P<0.05) differences are indicated by different subscripts.**

Analysis of the binary images of the dye tracer stained infiltration pathways indicated that in dry soil where preferential flow was most apparent, the EnviroSCAN sensors had between 87% and 100% probability of detecting an infiltration flow path depending on placement depth. However only 9.5% to 40% of the sensor area of detection was wetted by infiltration (Figure 3a). Consequently the magnitude of preferential flow was not able to be determined from a single probe as small differences in sensor placement would have had considerable influence on the measured change in soil moisture.

Analysis of the relationship between variance in the proportion of soil which participated in flow and the sensor sampling volume indicated the REV of the preferential flow paths occurred at a sensor sample area between 4000 $cm^2$ and 6000 $cm^2$ (Figure 3b). Consequently in dry soil conditions in which preferential flow processes were most apparent, 2-3 EnviroSCAN probes were required to accurately measure changes in soil moisture.

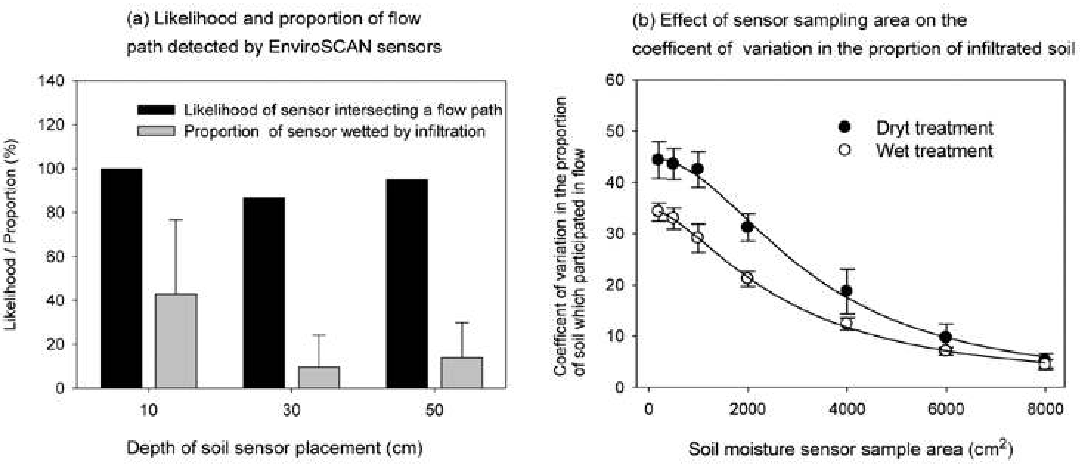


**Figure 3 Effectiveness of soil moisture sensors for detecting infiltration. (a) Likelihood, and proportion of the sensor sampling area being intersected by an infiltration flow path. (b) Effect of sensor sampling area on variation in the proportion of soil which participated in flow.**

## Conclusion

This study demonstrated that high frequency soil moisture monitoring via a capacitance probe mounted inside a vertically rammed access tube was able to determine the occurrence, timing, and wetting front velocity of preferential flow events following rainfall. However 2-3 soil moisture probes were required to determine the flux of preferential flow or change in soil moisture following preferential flow at the pedon scale. High frequency soil moisture monitoring demonstrated that occurrence of preferential flow was not related to either rainfall intensity or rainfall magnitude. Rather, preferential flow mostly occurred when antecedent soil moisture content was below 49 % of the plant available water content (PAWC), and did not occur above 60 % PAWC. Results indicate that high temporal frequency soil moisture monitoring may be used to identify soil-landuse combinations in which the occurrence of preferential flow may reduce irrigation efficiency or pose increased risk of agrochemical mobilisation to shallow groundwater or waterways.

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# The effect of row spacing and pre-sowing soil water status on soil water extraction by chickpea in Western Australia

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In the Mediterranean-type environment of south-west WA chickpea is sown into a dry soil profile and rainfall before or soon after sowing is required for germination and seedling establishment. Flowering and podding stages are particularly sensitive to drought stress and consequently rainfall through the growing season is required to ensure adequate soil water reserves to avoid plant water stress. We studied the effect of row spacing (23 cm, 50 cm and 75 cm) on the partitioning of soil water extraction by the chickpea crop at different growth stages. To determine the effect of initial soil water status on row spacing response main plot treatments with (75 mm, simulating summer rainfall) and without pre-season irrigation were imposed in a split plot design to a calcareous loamy earth at Merredin, WA. At sowing, the 1 m soil profile of the irrigated treatment contained only 14 mm more water than the unirrigated treatment, indicating that the pre-season irrigation was ineffective and a large proportion of the applied water was removed as evaporation before sowing, as runoff and drainage were negligible. Nevertheless, the additional soil water significantly improved crop yields through improved early biomass production, even though it was exhausted by pod formation and pod filling. The effect of row spacing on grain yield was not altered with pre-season irrigation, indicating the in season rainfall was more of a condition which led to chickpeas in narrow or wide rows performing better. This was confirmed in two seasons with high and low May to October rainfall. In 2008 (232 mm rainfall) grain yields in wide rows were least, whilst in 2007 (163 mm rainfall) the narrow row spacing yielded less than the wider row spacings.

# Thursday 6 December

## NEW TECHNIQUES IN SOIL SPATIAL ANALYSIS

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# Flexible Latin-hypercube sampling in remote areas

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**Abstract**

The Burdekin catchment is one of the largest sources of sediment to the Great Barrier Reef. We wish to collect new soils data to complement existing site data, in order to make specific land management recommendations for least-cost abatement of erosion sources. This paper highlights how to optimally select additional sampling sites that respect the locations of prior sites, as well as consider logistical concerns and our desire to sample the entire landscape continuum. We achieve this via a computationally intensive simulated annealing procedure to simultaneously optimise several different criteria. Our approach is very flexible and enables the creation of maps of local alternative sites when target sites are inaccessible.

**Key Words**

Kullback-Leibler divergence, simulated annealing, soil erosion, site selection.

**Introduction**

The health of the Great Barrier Reef (GBR), off the coast of northern Queensland, Australia, is a subject of immense ecological concern. Sediment-laden run-off from agricultural land is considered to be a key factor that influences the quality of water arriving to the GBR (Wooldridge 2009; Brodie *et al.* in press). Catchment-scale modelling of the lands that drain into the GBR has indicated that the Burdekin catchment (ANZLIC id**:** ANZCW0703005427, Figure 1) is the largest source of this sediment, exporting about 25% of the total average annual load (~4 Tg/year) (Kroon *et al.* in press).

The catchment is dominated by grazing of natural vegetation across the majority of the catchment. Past mapping and sampling programs (Figure 1a) have provided a rich but patchy legacy dataset of site and polygon information over the catchment. The ultimate goal of this project is the reinterpretation of existing soils information and collection of complementary new soil information to assist with soil-specific land management recommendations for least-cost abatement of erosion sources.

The purpose of this paper is to highlight how we select our sampling sites to achieve this goal. These sites are selected to achieve the desired mapping scale across the whole region (spatial spread), to minimise logistical difficulties in travelling across country to sites (ease of access) and to document the entire landscape continuum (McKenzie *et al.* 2008). This paper outlines our strategy for simultaneously optimising several criteria. This strategy is computationally intensive, and as it is quite general as well as involving datasets that cannot be simultaneously held entirely within available computer memory, the strategy will require unique coding on the part of the user for each new scenario. The code used for this particular scenario will be made available upon request.

**Methods**

*Covariate information*

The remotely sensed data-sources we gathered for the study region include: (i) a 30-m Shuttle Radar Topography Mission digital elevation model (SRTM-DEM—Farr *et al.* 2007, Figure 1b); and, (ii) airborne gamma radiometrics (Geological Survey Queensland). The hydrologically forced SRTM-DEM was obtained from CSIRO/Geosciences Australia. In total, 5 covariates were used for the derivation of the latin hypercube sampling (LHS) locations (Minasny and McBratney 2006). Terrain-related covariates included the STRM-DEM itself, topographic wetness index, curvature and terrain ruggedness. The final covariate used was the weathering intensity index, based on gamma radiometrics.

*Deciding where to sample*

When looked at separately, our optimisation criteria will lead to very different layouts of points across the region. Optimising for spatial spread will lead to a triangular layout of points spread evenly across the parts of the region with less historical information. Optimising for ease of access will see all sites next to roads.

Optimising with respect to the covariate space will give perfect LHS structure. None of these criteria are directly competing with any other; when one focuses only on one it can lead to degradation in terms of others, but it does not have to be so. While there is no universal optimal with respect to all criteria, there will be many potential sets of sites that serve as a good compromise between these different goals.

*Information requirements*

We assume the following information is available: The desired number of sites has already been chosen (n=600). All required covariate maps (*.img images, compatible with the Geospatial Data Abstraction Library— www.gdal.org/) were created using a common projection/datum. In our case this includes maps of:

1. distance from each potential site (30m by 30m pixel based on SRTM-DEM) to closest prior existing site (for spatial balance)
2. distances from each potential site to closest road (for ease of access) and closest stock route (all sites should be at most 3km from road / track)
3. maps of relevant covariates (for LHS).

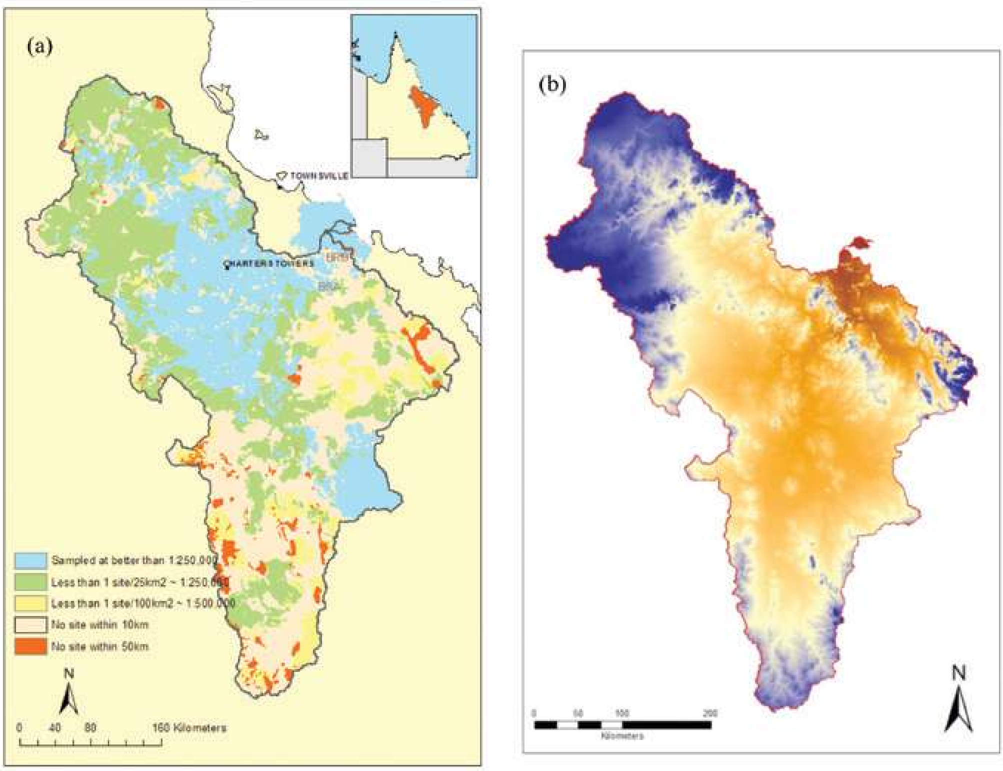


**Figure 1. The Burdekin Catchment Area: (a) adequacy of existing soil information; and, (b) hydrologically forced SRTM-DEM.**

*Optimisation criteria*

We specify mathematical functions associated with each optimisation criterion for which lower values are optimal. For spatial balance we use the sum of the reciprocal of the distances from each site to their closest neighbour (which includes prior existing sites). Zero distances are possible when two sites are co-located, in which case the reciprocal is replaced with a suitably large value. For ease of access we use the sum of the distances from each site to the closest road or stock route. For optimising in terms of LHS we use the Kullback-Leibler (K-L) divergence measure separately for each covariate (Kullback and Leibler 1951). This measure is a summation of terms of the form 2 O log(O/E) where E is the expected number of covariate values in a bin, and O is the observed number of points in that bin. Covariate information at prior sampling points is included here so that the additional site data is complementary to existing data and not duplicating prior sampling work.

This approach to LHS is slightly different to the approach taken by Minasny and McBratney (2006). Those authors use |O-E| as their optimisation criteria, where their expected number of observations per bin is E=1. Readers familiar with chi-squared tests might prefer to use (O-E)^2/E. The K-L divergence measure has an interpretation as a likelihood ratio statistic, and when O is approximately equal to E it can be shown that 2 O log(O/E) is approximately equal to (O-E)^2/E. The criterion used by Minasny and McBratney (2006) does not generalise well as the expected values get larger. The penalty associated with observing 89 out of 90 is the same as the penalty associated with observing 1 out of 2 whereas the other metrics would weight these scenarios differently.

The elephant in the room in terms of traditional conditioned LHS has to do with our large value of *n* and the computational difficulty of evaluating associated quantiles for very large spatial datasets. If all data cannot be held in memory concurrently they can be estimated using the quantiles of a subsampled spatial dataset or incremental quantile estimation (Chambers *et al.* 2006). Finding the minimum and maximum values is computationally feasible for large datasets. Using bins of equal width, one can also compute histograms of the covariates. Such histograms define the expected values used in the K-L computations and are readily available as meta-data from GIS software.

Minasny and McBratney (2006) also aim to match correlations between covariates directly. Our approach to LHS ignores this but we can work in principal component space as a way of ensuring that the correlation structure of the covariates is also found within the covariate values of the sampled sites.

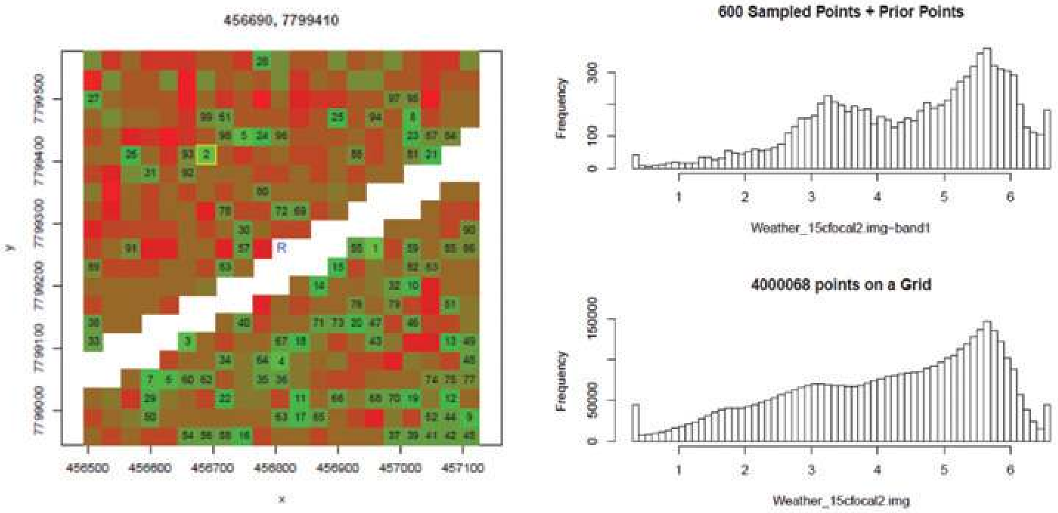


**Figure 2: Map of local alternatives (a) and histograms of weathering index values from our sample and a large subsample of the space (b). In (a): the target site is highlighted in yellow. The R indicates the point on the road closest to our target and the colour scheme rates appropriateness of each site to serve as an alternative to our target (green is good, red is bad, with several shades in between). The numbers in each pixel indicate the order of priority and for this example there is one site labelled 1 that provides a good alternative as it is closer to the road.**

*Optimisation and computation strategy*

Assume we have optimisation criteria $f_i(x)$ where *i* goes from 1 to K and x is a selection of *n=600* sites. We first use simple random sampling (SRS) of *n=600* sites to explore what values each optimisation criteria takes under SRS. Next we use simulated annealing (SA) to optimise each of the K criteria separately. Finally we combine the criteria based on their average (or median as appropriate) values under SRS and minimum observed values under SA. The combined value criterion is a sum of terms of the form $(f_i(x) – min_i) / (mu_i – min_i)$. Under SRS each component will on average equal 1, giving a combined value of K. If it were possible to fully optimise each component the combined value would be 0. For this example we have K = 7 criteria. The first deals with spatial spread, the second with ease of access and the final five are K-L divergence measures for our five covariates.

## Results

### *Implementation in the field*

Our computational approach is flexible enough to generate maps of alternative sites in the region of each target site. The major practical disadvantage of the conditional LHS approach is that one may not be able to visit a desired site for any number of reasons. Maps of local alternatives are produced to highlight sites that are comparable in terms of our optimisation criteria, or even make additional improvements on the selected sites. This gives the user on the ground the ability to select a valid alternative if required at short notice and eases the logistical burden of sample collection. Figure 2a shows one such example. Maps such as the example shown in Figure 2a are also produced in GIS friendly data formats.

### *Matching covariate distributions*

Figure 2b shows the histogram of weathering index values for our selected sites and previously sampled sites above a histogram for the same variable based on a large gridded subsample of sites. The K-L divergence measure has matched the shapes of these histograms very well. Plots for the other covariates show similar success.

### *How much optimisation do we achieve*

In our example we have K=7 terms and we can reduce the combined criteria from 7 (for SRS) to 2 in half a million SA steps, which is roughly equivalent to getting to within 70% of each optima when they are considered separately. Figure 2b shows how well K-L can match the distribution of sampled sites with the distribution in values for the population for weathering index. Our optimisation strategy targets ease of access specifically. All sites are required to be less than 3km from roads. The average distance travelled to our *n* sites when they are selected by SRS will be over 720km. Our final choice of sites will require just over 300km off-road travel. We could further reduce this by placing more emphasis on the ease of access in the combined optimisation criteria.

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# Paddock-scale soil carbon inventory and monitoring: A meta-analysis for sample size requirements

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## Abstract

The aim of this research was to collate published soil carbon variograms and use these to estimate the variance of the mean to estimate sample size requirements for paddock-scale carbon inventory and monitoring. In terms of inventory we calculated the number of samples required to estimate a mean where the 95% confidence interval was 10%, 20%, 30% and 50% either side of the mean. The number of samples required to detect a change from the baseline mean of 10%, 20%, 30% and 50% was also estimated. For inventory, estimating the mean became practical in terms of sample size (typically 5-10) when the precision required was a confidence interval of 20% either side of the mean. For estimating a change, a practical sample size (typically 4-8.25) could only detect a change of 30% from the baseline mean. In conclusion precise estimates for inventory are possible but smaller changes in carbon are expensive to detect. More efficient sample designs and cheaper measurement methods for carbon will reduce the sample size requirements, possibly to affordable levels.

## Key Words

Soil carbon; monitoring; sampling; sample size

## Introduction

There is an increasing interest in the inventory and monitoring of soil carbon due to the potential for soil carbon sequestration to offset greenhouse gas emissions. However there is an increasing realization that the costs associated with inventory and monitoring could be prohibitive. Due to this the importance of appropriate sample and monitoring designs cannot be under-estimated.

When designing a sampling or monitoring scheme the user is faced with two options: a design-based approach or a model-based approach (de Gruijter *et al.* 2006). The requirements for a design-based scheme are that all samples taken have an associated inclusion probability and are randomly selected based on this requirement. In the model-based approach there is no requirement for randomization. The key distinction between the two is that the estimation of uncertainty for any prediction (i.e. mean) has different statistical foundations. In the design-based approach the uncertainty is based on the chosen design and associated inclusion probabilities whereas in the model-based design it is provided by the model developed to describe the spatial variation. In broad terms a design-based approach is more useful to estimate the mean for a region, e.g. a paddock, whereas a model-based approach is more useful for predicting variation within a region, i.e. creating a map of soil carbon in a paddock. As a result design-based methods are considered by many to be more appropriate for carbon inventory and monitoring where, given the costs of measurement, we can only hope to estimate means for paddocks, farms etc.

Assuming a design-based scheme is to be adopted, then a key question is how many samples do I need to estimate and/or monitor carbon and how useful will my estimates be? Numerous studies have addressed this approach but the results have generally been specific to a particular site or area and may not be useful for other locations. The typical approach is to look at the number of samples required to estimate a specified change (Lark, 2009) and in some cases combine this with modelling to work out how long before the change can be detected (Hungate *et al.,* 1996). Based on this growing literature we feel some sort of meta-analysis would be useful to give some general guidelines on sample size requirements. However one limitation is that in many studies it is not clear if a true design-based sampling scheme has been adopted which could mean that the estimate of variance and mean is biased. An alternative is to examine the numerous studies of within-paddock spatial variation that have been published and in particular ones that have modelled the variation with a variogram. While these studies have adopted model-based approaches the work of Domburg *et al.* (1995) has shown how an unbiased estimate of the variance of the mean can be calculated from the variogram.

The aims of this paper are to: (i) compile a database of published soil carbon variograms with the restriction that the size of the study area approximates a paddock; and, (ii) extract unbiased estimates of the variance of the spatial mean to work out the number of samples required to estimate carbon and the change in carbon with different levels of precision at the scale of the paddock.

## Materials and Methods

*Variogram database*

McBratney and Pringle (1999) presented a database of published variograms for a range of soil properties, one of which was soil organic carbon (%) for which 9 variograms were found. We have augmented this with more recent studies and present 18 variograms in total. There are many more published variograms than this but not all meet the criteria of being peer-reviewed, covering areas 1–400 ha (the range of possible paddock sizes) and using raw rather than de-trended or transformed data. Table 1 summarises the key features of the database and Figure 1 plots the variograms.

**Table 1: Summary of data from studies that have published an organic carbon variogram.**

| Statistic | Depth (cm) | No. samples | Area (ha) | Mean C (%) | Variance C (%2) |
|---|---|---|---|---|---|
| Min | 10.0 | 28 | 1.0 | 0.18 | 0.003 |
| Q1 | 13.75 | 67.25 | 1.92 | 1.11 | 0.199 |
| Median | 19.0 | 99.0 | 7.75 | 2.08 | 0.504 |
| Mean | 18.9 | 170.3 | 44.72 | 2.46 | 0.762 |
| Q3 | 22.5 | 161.5 | 22.55 | 3.05 | 0.778 |
| Max | 30.0 | 576 | 400.0 | 7.4 | 5.1 |

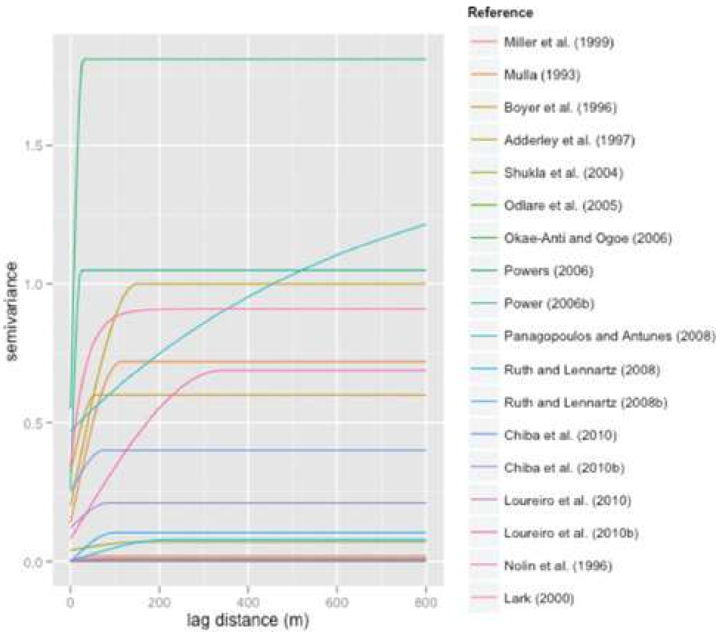


**Figure 1. Published soil organic carbon variograms.**

*Statistical analysis*

Domburg *et al.* (1994) showed that the variance of the sample mean of an area could be estimated from the semivariogram using the formula

$$\sigma_s^2(u_s) = \frac{\bar{\gamma}(B)}{n}, \tag{1}$$

where $\sigma_s^2(u_s)$ is the variance of the sample mean, $\bar{\gamma}(B)$ is the mean semivariance of a block of land, $B$, and $n$ is the number of observations.

In this work we assume we wish to measure and monitor carbon in a 5 ha block of land. We also assume simple random sampling will be adopted as Eqn. 1 is based on this assumption. For inventory we wish to know how many samples are needed to estimate the mean carbon with a 95% confidence interval that is (i)

10% (ii) 20% (iii) 30% and (iv) 50% either side of the mean. For inventory we wish to know how many samples are required to detect a change in carbon of (i) 10% (ii) 20% (iii) 30% and (iv) 50% from the baseline mean. The steps for analyzing each variogram are

1. Estimate $\bar{\gamma}(B)$ by numerical integration which involves randomly selecting pairs of locations in the 5 ha block and calculating the semivariance, and repeating this 10,000 times to estimate the mean semi-variance;

2. Estimate the $\sigma_s^2(u_s)$ for different numbers of samples, from $n$ = 2–200;

3. Identify the smallest value of $n$ that estimates the 95% confidence interval within the specified % of the mean ($\mu$), where the 95% CI is calculated by

$$CI(95\%) = \pm t_{0.025} \times \sqrt{\sigma_s^2(u_s)}, \tag{2}$$

where $t_{0.025}$ is the $t$-statistic for the 0.975 quantile with $n$-1 degrees of freedom, and $\sqrt{\sigma_s^2(u_s)}$ is the standard error of the mean;

4. Identify the smallest value of $n$ that estimates the 95% confidence interval for the change in carbon that is outside the specified % change in carbon relative to the baseline value, as in this case a significant difference has been detected. In this case the 95% CI is calculated by

$$CI(95\%) = \mu_d \pm t_{0.025} \times \sqrt{\sigma_1^2(u_1) + \sigma_2^2(u_2) - 2\sigma_{1,2}^2(u_1, u_2)}, \tag{3}$$

where $\mu_d$ is the difference in mean between two time periods or $|\mu_2 - \mu_1|$ based on $n_1$ samples in the first time period and $n_2$ samples in the second time period, $t_{0.025}$ is the $t$-statistic for the 0.975 quantile with $n_1 + n_2$-2 degrees of freedom, and $\sigma_1^2(u_1)$, $\sigma_2^2(u_2)$ and $\sigma_{1,2}^2(u_1, u_2)$, is the variance of the mean for time periods 1, 2 and the covariance between the 2 periods, respectively.

For estimating the number of samples required to detect a change we make the following assumptions:

a) that $\sigma_1^2(u_1) = \sigma_2^2(u_2)$;

b) that $n_1 = n_2$;

c) simple random sampling is adopted for each time period so the sites are not revisited and the covariance term in Eqn. 3 is equal to 0. While this results in less precise estimates of change and an inflation of the number of samples needed (Lark, 2009), it is considered by some more practical in an operational soil carbon accounting system as it prohibits land owners from attempting to cheat the system by for example adding organic inputs to sites they know will be revisited.

**Results and Discussion**

The results of the analysis are summarised in Table 2 for inventory and Table 3 for monitoring of soil carbon.

**Table 2. Number of samples required for carbon inventory with different levels of precision.**

| Statistic | CI +/-50% mean | CI +/-30% mean | CI +/-20% mean | CI +/-10% mean |
|---|---|---|---|---|
| Min | 2.0 | 2.0 | 2.0 | 3.0 |
| $Q_1$ | 3.0 | 4.0 | 5.25 | 12.75 |
| Median | 3.0 | 4.0 | 7.0 | 18.5 |
| Mean | 3.3 | 5.0 | 7.8 | 23.8 |
| $Q_3$ | 3.75 | 6.5 | 10.0 | 33.75 |
| Max | 5.0 | 9.0 | 17.0 | 62.0 |

Table 2 and Table 3 have large maximum values and given the skewness of the results it is best to examine the range of samples for the 1st and 3rd quartiles as these represent the middle 50% of studies. In terms of inventory these are prohibitively large for estimating a CI that is 10% either side of the mean ($n$ = 12.75-33.75) but a value of 20% is a good compromise ($n$ = 5.25-10.0). The results for detecting a change follow a similar pattern in that the number of samples to detect a change of 10% is too large ($n$ = 22.25-64.25) but a value of 30% is financially realistic ($n$ = 4.0-8.25). However, given that much carbon sequestration will result in smaller changes through management change rather than land use change, detecting a change of 30% may not be useful.

**Table 3. Number of samples required for detecting a change of differing magnitudes relative to the baseline mean**.

| Statistic | 50% change | 30% change | 20% change | 10% change |
|---|---|---|---|---|
| Min | 2.0 | 2.0 | 2.0 | 2.0 |
| Q1 | 3.0 | 4.0 | 6.5 | 22.25 |
| Median | 3.0 | 5.0 | 9.5 | 33.5 |
| Mean | 3.3 | 6.2 | 12.2 | 43.9 |
| Q3 | 3.8 | 8.25 | 17.0 | 64.25 |
| Max | 6.0 | 15.0 | 31.0 | 121.0 |

As found in previous studies, detecting a change requires more samples than does inventory for equivalent levels of precision. This is to be expected, as the precision of the confidence interval is larger for a change when we do not consider the covariance between sampling periods (Eqn. 3). Furthermore, the effect size is smaller for monitoring as we are trying to estimate a difference between two means rather than the actual mean. The amount of samples required is linked to the variogram structures, namely the sill semivariance and the range. The sill semivariance represents the maximum variation in the study area so as this increases so should the sample size requirements. However this interacts with the range in that the semivariance increases until this point. Therefore any increase in the size of the study area beyond the range will not increase the semivariance and have a declining impact on the mean semivariance and therefore the variance of the sample mean, and therefore sample size requirements. In this work we assumed a 5-ha block (~223m by 223m square) and based on the variograms presented (Fig. 1), 16 of 18 have reached their sill semivariance at this scale so increases in the hypothetical block size will result in minor increases in the sample size requirements. Furthermore, it should be noted that we have assumed simple random sampling and that more sophisticated designs should be able to improve the precision of the estimates and reduce the required sample sizes presented in Tables 2-3. Due to this the results presented here are at the upper limit of the likely sample size requirements.

In conclusion, we found that moderate sample numbers (5.25-10.0) are needed to estimate the mean soil carbon content of a block with a 95% CI that is 20% either side of the mean and a similar number of samples (4.0-8.25) could detect a change in carbon that is 30% of the baseline mean. Future work could combine this work with published design-based studies to augment the variogram database and it could easily be extended to soil nutrients for site-specific crop management. Finally we believe by collating and analysing past studies we have developed useful rule-of-thumb estimates for determining sample size requirements for the inventory and monitoring of carbon.

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# A preliminary attempt at fuzzy disaggregation of a regional-scale soil map

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## Abstract

One way to address the demand for soil information at ever-finer spatial scales is to disaggregate an existing soil map. We can do this on the basis that: (i) soil variability is closely related to topography, and that topography can be easily obtained at a relatively fine spatial resolution; and, (ii) the soil map provides information on how many (non-spatial) sub-units (herein 'entities') are expected to be found within any individual map unit. Essentially, the task is to allocate spatial locations to the entities. In this study we have disaggregated a 1:250,000 soil map for a region of north Queensland, Australia, to a scale 1:3000. Through expert knowledge, the *k* entities of each map unit were first attributed with a probability of occurrence. We then assembled a suite of 19 covariates for the study region, derived from topographic and gamma radiometric information, where the latter served as a surrogate for intrinsic soil variability. Next, fuzzy *k*-means classification of each map unit allowed us to allocate the entities to locations. Fuzzy classification is an intuitively appealing method because each pixel in the disaggregated map unit has a membership to each entity. Future work is needed to verify whether the mapped entities occur where they are expected to be found in the landscape. Other issues that require further research are dealing with highly skewed but correlated covariates, and how to classify the pixels that lie at the boundary of two or more polygons.

## Introduction

Soil survey is tasked with documenting the landscape continuum (McKenzie *et al.* 2008). The (choropleth) map produced by the surveyor classifies the landscape into discrete 'map units' that are associated with some stated spatial scale (Reid 1988). Such soil maps have been tremendously successful at conveying pedological knowledge for over a century now; however, the incongruity of applying discrete classification to a continuum has not escaped criticism (Burrough *et al.* 1997). Additionally, it is fair to say that technology has not been kind to the perception of the soil map, as remote sensing now generates spatial information at an extent, scale, and cost that is prohibitive to traditional soil survey. It is therefore tempting to disregard the soil map as an anachronism, though this would be disrespectful of the accumulated knowledge of generations of soil surveyors (Bui 2003). The obvious question is, can we use the spatial information contained in remote sensing to disaggregate (or, more specifically, 'dissever'—Malone *et al.* 2012) a soil map? If so, this would go some way to satisfying an increasing demand for landscape information at scales finer than that which conventional soil maps can provide.

Bui and Moran (2001) showed a way forward when they dissevered the units of a 1:250,000 soil map into 'facets' (their equivalent of entities), using *k*-means classification using covariates that comprised slope position and the bands of a Landsat MSS image. The method has two requirements: (i) an assumption that soil variability is closely related to topography (Jenny 1941; McBratney *et al.* 2003), and topographic information is readily available; and, (ii) the soil map provides information on how many (non-spatial) sub-units (herein 'entities') are expected to be found within any individual map unit. Essentially, the task is to allocate spatial locations to the map units' entities. Without either of points (i) or (ii) the scope for disseveration is limited. But while the *k*-means classifier used by Bui and Moran (2001) dissevers, it ultimately ignores the soil continuum within the map unit: the entities can only form a discrete nesting of the map unit. An intuitively appealing advance is to use fuzzy *k*-means (McBratney and Moore 1985) for the disseveration. As far as we know, nobody has successfully implemented this idea on an operational scale.

## Research context

The health of the Great Barrier Reef (GBR), off the coast of northern Queensland, Australia, is a subject of immense ecological concern. Sediment-laden run-off from agricultural land is considered to be a key factor that influences the quality of water arriving to the GBR (Wooldridge 2009; Brodie *et al.* in press). Catchment-scale modelling of the lands that drain into the GBR has indicated that the Burdekin catchment is the largest source of this sediment, exporting about 25% of the total average annual load (~4 Tg/year) (Kroon *et al.* in press).

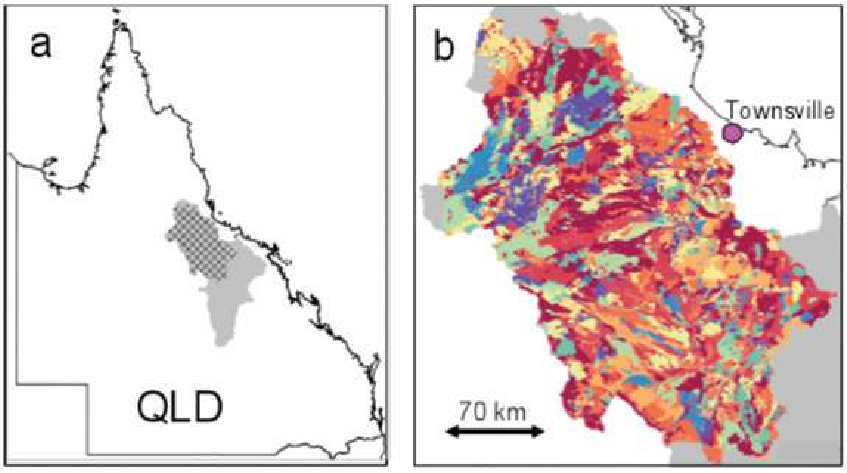


**Figure 1: (a) Cross-hatching indicates the extent the Dalrymple Shire soil map within the Burdekin catchment (in grey); (b) close-up of soil map, with map units attributed a random colour.**

Soil management has a crucial role in restricting the amount of sediment that can enter a waterway. Our aim here was to investigate, for a sub-region of the Burdekin catchment, whether its soil map (and associated knowledge about entity erodibility) could be disaggregated to a fine spatial scale, based on fuzzy *k*-means classification of readily obtained remotely sensed covariates.

## Material and Methods

*The Dalrymple Shire soil map*

The existing soil map of the Dalrymple Shire was published by Rogers *et al.* (1999), at a scale of 1:250,000 and an extent of 67,660 km$^2$ (Figure 1). This mapping was a refinement of a previous land degradation study (De Corte *et al.* 1994), but the sample of profile descriptions was still so sparse that we could not hope to derive a map of fine-scale soil erodibility.

*Covariates*

The remotely sensed data-sources we gathered for the study region were: (i) a 30-m Shuttle Radar Topography Mission digital elevation model (SRTM-DEM—Farr *et al.* 2007); and (ii) airborne gamma radiometrics (Geological Survey Queensland). The SRTM-DEM was obtained from CSIRO/Geosciences Australia, and had been hydrologically forced.

In total, 19 covariates were available for fuzzy disseveration of the soil map. Terrain-related covariates included the STRM-DEM itself, and the following derivatives: slope, mid-slope position, slope height, normalised height, curvature , plan curvature, topographic wetness index , distance to channel network, valley depth, and the MRVBF and MRRTF (respectively, the multi-resolution valley bottom flatness index, and the multi-resolution ridge-top flatness index—Gallant and Dowling 2003). The gamma radiometric-related covariates used for fuzzy disaggregation were the individual band values for potassium (K, %), uranium (U, ppm), thorium (Th, ppm), and the band ratios Th/K, U/K, U/Th and $U^2$/Th (IAEA 2003).

*Fuzzy disseveration*

We re-wrote the open-source code for the 'FuzMe' fuzzy *k*-means algorithm (Minasny 2004), so that it would work for the R statistical software (R Development Core Team 2011). Note that we have implemented the standard algorithm, not the variant that deals with extragrades. To avoid issues of computer memory, we processed each map unit independently. We stipulated that the following cases were excluded from classification: (i) pixels associated with a null value in at least one covariate; (ii) pixels that fell on the edge of a map unit; (iii) covariates that had no variation within a map unit; and (iv) covariates that were collinear with at least one other covariate. To be classifiable, a map unit had to have more pixels than covariates. When this condition was satisfied, Mahalanobis distance was used to describe the covariance of the covariates; however, due to strong skewness of the covariates (which violated the assumption of multivariate normality in the distance matrix), we truncated all covariate values to four significant figures, and converted them to ranks. FuzMe was then invoked to identify *k* classes in the ranked data. The fuzzy exponent was held constant at 1.1 throughout. Minimisation was attempted 5 times per map unit, and the classification that returned the smallest value of the objective function was retained. The number of pixels where each class had the greatest membership was counted and ranked. The entity with the largest expected probability in the map unit was then allocated to the pixels of the most populous class, and so on in decreasing order until the entity with the smallest expected probability was allocated to the least populous class.

**Results**

The R-written FuzMe algorithm took 4 days to dissever the Dalrymple Shire soil map, using a 64-bit PC running Windows 7, with an Intel Xenon CPU (3.47 GHz) processor and 24 GB of RAM. We considered this processing time to be substantial, though not prohibitive for the task. If the study region were much larger we would obviously be forced to increase computational efficiency. Two possibilities for this would be (i) re-packaging the relatively slow Mahalanobis distance computations in a compiled language, such as FORTRAN or C, to be called by R as a dynamic-link library; and (ii) parallel processing of the map units. In short, computational practicality does not pose a problem so long as the map units are processed independently.

On viewing the dissevered map (Figure 2a–c) it was apparent that some aspects of the method need to be refined. First—and probably most important—while clear topographic patterns were evident, there is a need to validate whether the 'spatialised' entities are associated with their most likely place in the landscape.

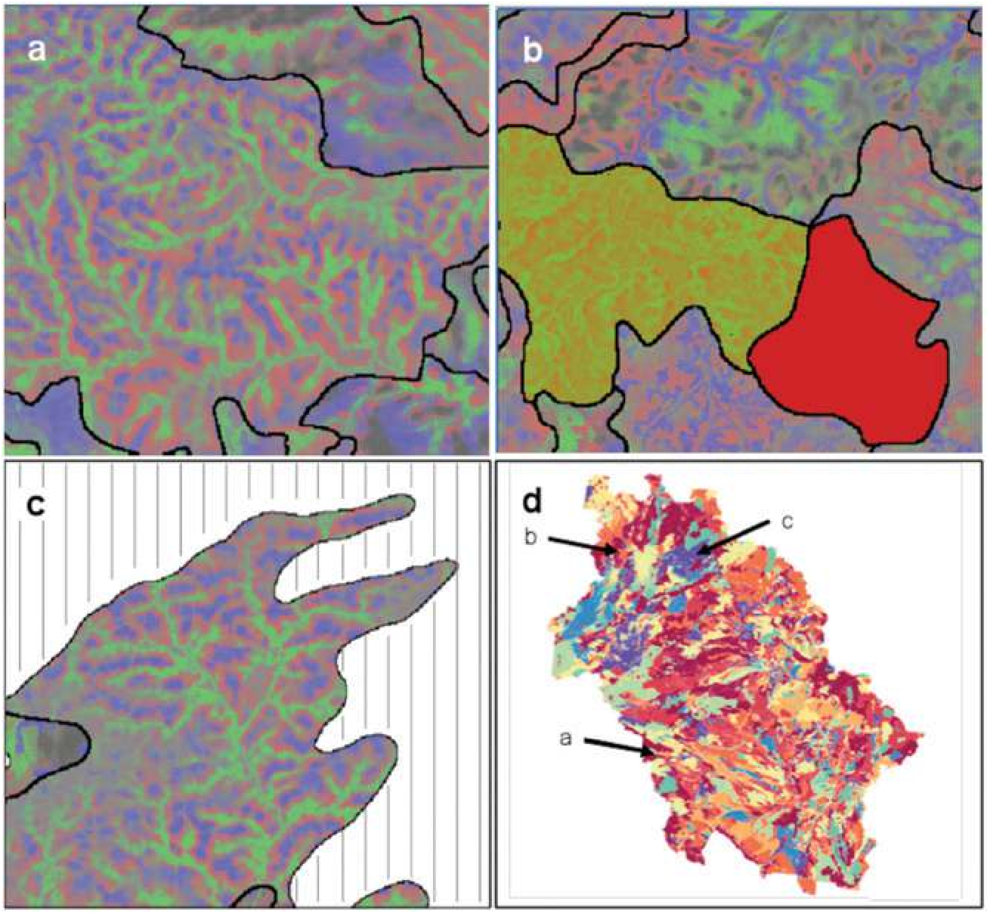


**Figure 2: Panels a–c are extracts from the dissevered soil map, and Panel d shows the locations of the extracts in the original choropleth soil map. Each extract is at a scale of approximately 1:90,000. For panels a–c: red hues indicate a large membership to the most probable entity, green hues indicate a large membership to the second-most probable entity (if present), and blue hues indicate a large membership to the third-most probable entity (if present); grey hues indicate large membership to a fourth- of fifth-most probable entity; black lines indicate polygon edges; vertical hatching indicates a faulty polygon that could not be processed.**

A second possible refinement is the elimination of edge effects in the classification, induced by processing each map unit independently: while all pixels should theoretically have non-zero membership to all entities in all map units, our method effectively stipulates that the membership of a pixel to the entities of a neighbouring map unit is zero. The pixels on the edge of two map units could be expected to belong strongly to more than one map unit, and were consequently excluded from the classification (the black lines in Figure

2a–c). It is possible that the edge memberships could be approximated through some kind of post-processing with a moving window. A third possible refinement is adequate display of the output: a conventional 3-band RGB colour scheme it is not satisfactory for map units with >3 entities. Localised examples of this can be seen in Figure 2a–c, with the colour fading to grey where the most probable class is the fourth- or fifth-most probable entity. Hengl *et al.* (2006) presented a method around this, based on the HSI (rather than RGB) colour model, although it would ultimately mean that every map unit has a different legend, which might be practically difficult to interpret. Future work should also focus on minimizing the effect of faulty polygon vertices, which are inevitable in an intricate digitisation of a soil map (Figure 2c). A final issue to consider is the treatment of strongly skewed, yet undoubtedly correlated, covariates. We circumvented the problem of skew by converting the covariates to ranks; in this way Mahalanobis distance metric becomes analogous to a Spearman (rank) correlation. Rank correlation itself could be used to pre-screen the dataset for redundant covariates prior to running FuzMe, although we have not yet pursued this option.

## Conclusion

We have made a preliminary attempt to dissever a regional-scale map, where the scale has been reduced from 1:250,000 to effectively 1:3000. Implementation of the method is computationally feasible for large study regions, but we have found that the resulting map raises complex issues related to presentation and interpretation. Some of these issues might be easily solved (colour schemes, redundant variables), although solutions for other issues remain less obvious (extensive validation requirements and edge effects between map units).

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# Scale-dependent (co-) variation of carbon and soil moisture

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## Abstract

Soil carbon and moisture in a landscape are highly variable in space, depth and time and many drivers influence those variations. Some studies have suggested that carbon and water related variables and their correlations as well as their drivers are scale-dependent. Hence, understanding the scale-dependency and correlation of carbon and water is required for proper management of our ecosystems. However designing an appropriate sampling scheme to capture those variations with reasonable accuracy to meet a low budget is the key challenge. Using a nested sampling design and hierarchical ANOVA seems a reasonable approach for the above challenge, however studies on such designs are limited. Therefore a study was aimed at understanding the scale-dependency and correlation of carbon and water using a nested sampling design, hypothesising that they are scale-dependent. The paper describes the baseline sampling stage with four spatial scales such as 5, 30, 100 and 400 m, univariate and bivariate analysis of moisture, carbon and respiration (in vitro) of surface soil data and their implications in a forested ecosystem. The dominant scale for most carbon and respiration related variables was the 5 m scale (60-100% of total variation) followed by 30 m scale (15-40% variation). At the 30 m scale, very strong correlation was observed between carbon and respiration (0.84). All observed variables showed a moderate correlation (~0.4) with carbon at 5 m scale except clay which showed strong correlation (0.67). The computed product moment correlation coefficients for variable combinations were very similar to the computed correlation coefficients at the 5 m scale.

**Key Words**

Nested design, scale-dependent correlation, scale-dependent variation

## Introduction

Understanding current soil carbon and water contents is crucial to research on carbon accounts; however it is not an easy task as they are spatially, temporally and vertically heterogeneous in a landscape. Therefore designing an appropriate sampling scheme to capture these variations with reasonable accuracy and cost efficiently as a preliminary reconnaissance is also difficult. Many authors have studied the spatial variability of soil moisture (Hu *et al.* 2011), and soil carbon (Chuai *et al.* 2012) and stated that both variables are influenced by a number of factors or drivers such as vegetation, topography, soil properties, parent materials, water transporting processes, depth to water table and/or meteorological conditions. However both the contribution of those drivers to the total variation of the variables and the correlations among variables tends to vary with spatial scales (Corstanje *et al.* 2008; Corstanje *et al.* 2007; Lark 2005). Many models are developed using data from one scale, therefore they may not necessarily perform similarly at other scales. Hence, it is vital to understand the scale-dependent nature, strength and type of relationship of carbon, water and their drivers as it has implications for how carbon and water in a landscape are managed. The importance of nested sampling designs is well documented in terms of their value as a way to understand variation across divergent scales but their use for understanding the co-variation of soil properties is less studied. Therefore, this study is aimed at understanding the scale-dependent correlation of carbon and water through a balanced nested sampling design hypothesising that the (co-) variation of carbon and water are scale-dependent. From this we can understand what drives variation in carbon and water at different scales and attempt to manage them.

## Materials and Methods

The study was conducted within a first-order forested catchment of the Wollondilly River at Arthursleigh (34°33’49”S and 150°04’58”E) near Marulan, in the Southern Highlands of New South Wales, Australia (Figure 1). The region has a temperate climate (based on the Köppen classification) with mean annual precipitation of 800 mm and mean annual temperature around 19°C (Bureau of Meteorology 2010). The study area is approximately 175 ha with an elevation range from 590-640 m above MSL sloping towards the northeast. There has been no detailed soil survey performed in the area, however, the soils have been identified as Tenosols/Rudosols in the present survey (unpublished data). Soils have developed over sedimentary rocks (Adaminaby Group), with a sandy texture consisting of 75-80% of sand and the soil depths (A-C horizons) range between 0.3 and 0.7 m. The density of the vegetation is relatively low and the tree stratum of the area is mainly formed by *Eucalyptus* species, which have not been affected by bushfires over the past 55 years. Scattered shrub vegetation is also present in the area, however, a quite distinct range of height is observed among the trees and shrubs, namely 10–15 m for *Eucalyptus* and 0.50–1.0 m for shrubs. Coarse woody debris and fine litter materials are also in abundance in the study area.

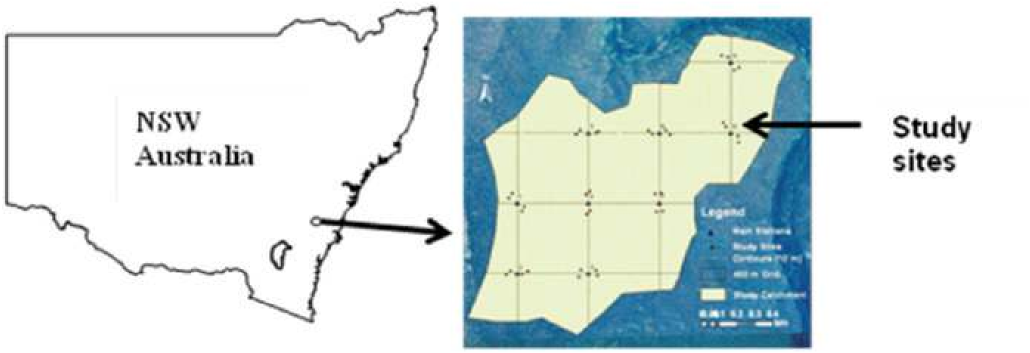


**Figure 1: Location of the study catchment (Arthursleigh) and sampling design.**

The experiment was conducted as a balanced nested design (Youden and Mehlich 1937) with nine main stations. Each station consisted of eight sampling locations covering four spatial scales, i.e. 5 m, 30 m, 100 m and 400 m (Figure 2). The main stations were selected using a 400 x 400 m grid (first-stage sampling) and the second-stage sampling points were selected 100 m away from first-stage points in a random direction. Third-stage sampling was selected 30 m away from second-stage points in a random orientation and fourth-stage points were selected 5 m away from third-stage sampling points. The sampling scheme consisted of 72 sampling points with full replication of each stage as shown by the topology in Figure 2. The distances selected in the study represent the tree scale (5 m), stand scale (30 m), hill-slope scale (100 m) and the soil-type scale (400 m) variations. We hypothesised that the strength of the relationship between carbon and water in a catchment and what drives the carbon and water variation both change with spatial scale.

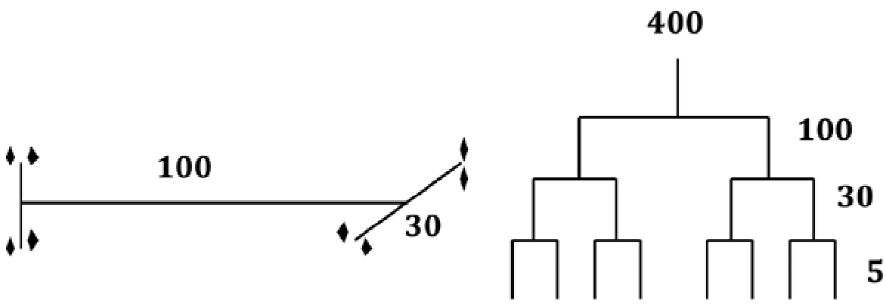


**Figure 2: Sampling points of a main station and topology of sampling design.**

Surface soil samples (0-10 cm) were collected from each of the 72 sites, using a hand auger and located with a GPS receiver. Smaller soil pits (15 x 10 x 10 cm) were used to collect soil samples due to the sandy nature of the surface soil. Soil sampling was done within a single day at around 3-6°C air temperature during winter (June 2010). Unsorted winter samples were immediately transferred to temperature controlled cooler chests, transported to the laboratory within the same day and stored at 4°C. Each soil sample was passed through a 2 mm mesh sieve and a portion of it was stored at 4°C without air drying to facilitate heterotrophic respiration analysis while the rest were air dried for carbon and clay analysis (particle size distribution). Separate soil samples were collected from 0-10 cm depth for surface gravimetric moisture determination.

Total soil carbon was determined using a Vario MAX CNS analyser (Elementar Analysen systeme GmbH). Air-dried soil samples were ground to pass through a 53µm sieve and subjected to total carbon analysis. Care had to be taken to grind whole sub-samples (approximately 7.5g ±0.25) to avoid over-sampling of the clay fraction, which has more carbon than the sand fraction. A flow-through gas exchange monitoring system was used for soil heterotrophic respiration measurements, comprised of an infrared gas analyser (IRGA) for $CO_2$ efflux (Jenkins and Adams 2010).

A multivariate linear mixed model was fitted to the data (Lark 2005) to estimate the variances and co-variances at each scale (5 m, 30 m, 100 m, 400 m), and from which a correlation can be estimated. Product moment correlation coefficients were also computed for comparison with the scale-dependent correlations.

**Results and Discussion**

Summary statistics of the measured variables of all 72 data points are presented in Table 1. Carbon and soil moisture were found to be positively skewed and subsequently transformed to normality. The observed coefficients of variation (% CV) of carbon-related values were within the range of 40-50% and the % CV of moisture content and percent clay was around 30%.

**Table 1. Summary statistics of the data**

| Variables | Min. | Max. | Mean | Median | Variance | Std Dev | CV% | Skewness | Kurtosis |
|---|---|---|---|---|---|---|---|---|---|
| Soil moisture (wb %) | 4.23 | 15.77 | 8.81 | 7.66 | 8.77 | 2.96 | 33.64 | 0.73 | -0.46 |
| Clay (%) | 0.10 | 3.46 | 1.94 | 1.94 | 0.32 | 0.57 | 29.34 | -0.55 | 2.01 |
| Total carbon (%) | 0.92 | 13.14 | 4.25 | 3.85 | 4.66 | 2.16 | 50.83 | 1.91 | 4.81 |
| Soil respiration at 30°C (µg/g of soil) | 2.87 | 17.65 | 8.1 | 7.23 | 10.39 | 3.22 | 39.81 | 0.87 | 0.36 |

The mean carbon and moisture content of the forest were about 4% (range 0.06-13.14%) and 9% (range 4.23-15.77%) respectively. Results of this study clearly indicate that the variations in carbon, soil moisture, clay content and heterotrophic respiration of the 0-10 cm soil layer are scale-dependent. The dominant spatial scales were 5 m followed by 400 m for both clay and soil moisture. The combined variation accounted for at both levels was 73- 75% (Table 2). In terms of carbon, 69% variation was observed at the 5 m scale followed by 25% at the 30 m scale (Table 2). The dominant scale for heterotrophic respiration was 5 m, which includes 69% of the variation, however similar variances were observed at 30 m and 100 m scales where all the other variables were less dominant. This is similar to past studies which have found variation in heterotrophic respiration at small spatial scales (i.e. 2.7 m in a 13 x 14 m area, Herbst *et al*. 2009). The finer scale variance of carbon also reflects the local differences in soil moisture due to pores, cracks and small-scale surface features such as furrows, tree debris and proximity to tree roots and shadows of the surrounding trees, which constrain microbial activity. The limited microbial activity can be attributed to the large amount of undecomposed leaf litter accumulated in the area. The 400 m scale represents the variation attributable to aspect, soil type and topographic changes. There are three soil types in the study area and the relief ranges from 0-26%. The soils of the study forest are derived from the same parent material (sedimentary rocks, Adaminaby group) and therefore are in agreement with the lower variation of variables at the 400 m scale compared to other finer scales.

**Table 2. Variance at each spatial scale**

| Scale (m) | Clay (%) | Resp 30°C | Log MC | Log C |
|---|---|---|---|---|
| 5 | 50.0 | 68.7 | 56.6 | 69.3 |
| 30 | 18.7 | 11.8 | 13.6 | 24.8 |
| 100 | 4.7 | 16.3 | 8.3 | 2.0 |
| 400 | 26.5 | 3.2 | 21.6 | 3.9 |

The correlations between carbon and other properties at different scales are presented in Figure 3. At the 30 m scale very strong correlation was observed between carbon and respiration (0.84). All observed variables showed a moderate correlation (~0.4) with carbon at the 5 m scale except clay which showed strong correlation (0.67). The switch in the sign of the correlations observed at the 100 m scale is interesting but perhaps not important as this variation contributes little to the overall variation. As suggested by Lark (2005) a correlation coefficient of 1.0, in this case at 400 m, needs to be interpreted carefully. Essentially it means the model has hit a boundary condition as the correlation is restricted to the range [-1, 1]. The product moment correlation computed between carbon and clay was 0.59 which is close to the scale-dependent correlation obtained at the 5 m and 30 m scales. A similar pattern was observed between carbon and soil moisture.

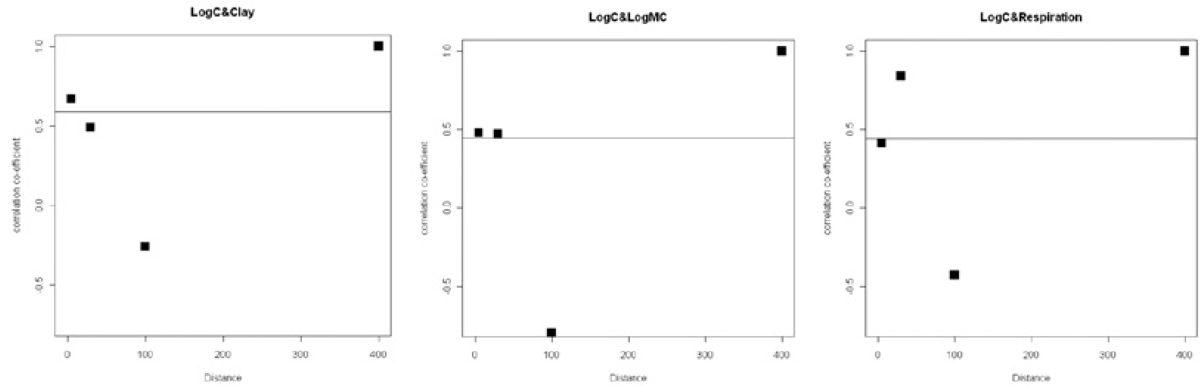


**Figure 3: Scale-dependent and product moment correlation coefficients (solid lines represent the product moment correlation coefficient for different variable combinations).**

## Conclusions and future work

The dominant scale of variation for soil carbon, soil moisture and heterotrophic respiration was the 5 m scale which comprised 50-69% variation and the least dominant scale was the 100 m scale. The next scale in order of dominance was the 400 m scale for soil moisture and clay but for carbon it was the 30 m scale. The relationship between carbon and other properties, as measured by correlation, did change with scales. This has implications for modelling as it implies that different model inputs are needed for different spatial scales.

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# Using GIS and radar imagery for assessment of land capability for arable agriculture in Rigo District, Central Province, Papua New Guinea

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## Abstract

Accurate and reliable spatial data on soil types is very limited in Papua New Guinea. The only regional digital dataset available is the Papua New Guinea Resource Information System (PNGRIS). This dataset is an highly extrapolated modelled GIS dataset which provides only a coarse and broad interpolation of probable soils which does not provide realistic on ground soil representation.

Using a 10 m Digital Elevation Model (DEM) derived from GEOSAR Radar data a four class Topographic Position Index was generated using Land Facet Tools Extension for ArcGIS (Jenness *et. al* 2011). This extension divided the topography into Ridges, Upper Slopes, Gentle Slopes and Valleys. Using this classification ”Lower Slopes” of less than and equal to 10$^{o}$ were selected as potential suitable sites for intensified arable agriculture. Slopes above 10$^{o}$ gradient are more susceptible to rill and sheet erosion due to the intense high precipitation events experienced during the wet season.

This broad topographic based classification was further constrained by the bedrock geology underling the previously identified lower slopes. The soils best suited to intensive agricultural production were identified as those areas underlain by intermediate or mafic rocks e.g. basalt, gabbro and other igneous rocks high in ferro-magnesium minerals and iron oxides and their derivatives which provide the potential for the development of more productive and sustainable agricultural soils.

This broad first pass classification identified 41,533 hectares of land for potential agricultural expansion within the Rigo District in the coastal lowlands and foothills centred about the town of Kwikila some two hours by road southeast of the capital Port Moresby. Of the 41,533 ha identified some 10,632 ha is identified within PNGRIS as being prone to waterlogging and inundation of varying duration and severity. Most observed current agriculture land also falls within this first pass classification.

## Introduction

Accurate and reliable spatial data on soil types is very limited in Papua New Guinea. The only regional digital dataset available is the Papua New Guinea Resource Information System (PNGRIS). This dataset is an extrapolated modelled GIS dataset which provides only a coarse and broad interpolation of probable soils which does not provide realistic on ground soil representation.

The coastal lowlands and adjacent elevated areas are known to have highly variable topography, including coastal plains, steep escarpments and dissected uplands/plateaux. Soils are also variable, with substantial areas of the coastal lowlands susceptible to inundation and steeper slopes overlain by shallow soils being susceptible to erosion (Bleeker 1983, Hanson *et al* 2001). For expansion of agriculture in Central Province, more detailed understanding of the location of suitable soils and their topographic limitations is needed. Historically, this would have been achieved through soil surveys, however, modern tools of GIS and radar imaging provide for rapid assessment of land capability for particular purposes. This assessment can be followed by ‘ground truthing’ and coupled with existing knowledge from, for example, field trials of crops and field data on soils, to assist in final assessment and decision making regarding agricultural development. This paper reports on the assessment of land capability in one potential area for agricultural development in Papua New Guinea, the Rigo district to the south east of Port Moresby.

## Methodology

Papua New Guinea Resource Information System (PNGRIS) data were initially analysed and considered to be an over extrapolation of the natural resource data sets available. Consequently, P-band GeoSAR radar elevation data, X-band GeoSAR Magnitude Radar Imagery, Regional Scale Geological Data and field observations of soils coupled with data from crop trial plots were used as primary data sources for the study.

X-band and P-band radar data is collected concurrently from each side of a survey aircraft at an elevation between 10,000 and 12,500 metres. The X-band wavelength penetrates clouds and reflects from tree canopy to deliver surface model data in forested areas and accurate terrain elevation in open areas. The P-band wavelength penetrates both clouds and tree canopy to deliver terrain elevation and surface feature extraction in forested areas. These characteristics make GeoSAR ideal for mapping large areas of mixed land cover particularly in Tropical areas such as Papua New Guinea (Williams and Jenkins 2009). The regional scale geological data provides the only credible bedrock information available for the selected study areas.

Tiled P-band radar surface points which penetrate all but the densest vegetation provide a high resolution model of the terrain. The points were provided by the Defence Imagery and Geospatial Organisation as ASCII point data with spacing of 2.5 metres, and were gridded to a mosaic of 10 m Digital Elevation Model (DEM) surfaces using the ArcGIS "3D Analyst" extension. This data provides a more accurate and higher resolution representation of the local topography than the publically available 30 and 90 metre Shuttle Radar Topography Mission (SRTM) data for the study area.

Using the 10 m DEM derived from the GeoSAR Radar data a four class Topographic Position Index (TPI) was generated using Land Facet Tools Extension for ArcGIS (Jenness *et al* 2011). This extension divided the topography into Ridges, Upper Slopes, Gentle Slopes and Valleys. Using this classification "Lower Slopes" of less than and equal to $10^{o}$ were selected as potential suitable sites for intensified agriculture.

This broad topographic classification was further constrained by the lithology or soil parent material underling the previously identified lower slopes. The area's deemed most suitable for intensified agricultural production were identified as those areas underlain by intermediate or mafic rocks or derived alluvium and colluvium which provide for deeper and base rich soil parent materials. Thus their derivative soils generally provide the potential for the more productive and sustainable agricultural lands. Limited numbers of soil profiles were described and soil types noted in road cuttings and gardens in the district.

**Results**

This broad first pass classification identified 41,533 ha of land for potential agricultural expansion within the Rigo district. The land was underlain by a wide range of parent material/bedrock, though most were of volcanic origin dominated by gabbro and basalt and a significant area of transported materials (9,899 ha, see Table 1 and Figure 1).

**Table 1: Geological bedrock and associated areas of land assessed as suitable for agricultural development in Rigo district, Papua New Guinea**

| Geological Bedrock | Area (Ha) |
|---|---|
| Basalt and andesite pyroclastics, lava and volcanic sandstone. | 3,955 |
| Basalt and andesite pyroclastics and minor lava. | 924 |
| Basalt and andesitic agglomerate, minor tuff; tuffaceous sandstone and volcanic conglomerate. | 725 |
| Basalt and minor andesite agglomerate and tuff, partly reworked | 312 |
| Basalt and pillow lava with gabbro and dolerite intrusives (dykes), minor calcilutite | 381 |
| Gabbro, diorite, dolerite, basalt and other acid differentiates. | 25,285 |
| Gravel, sand, silt, mud, clay: alluvium and beach deposits downslope and adjacent to mafic and intermediate bedrock | 9,899 |
| Massive green mafic schist derived from basalt, dolerite, gabbro, volcanic sediment; and minor calcareous and felsic schist or phyllite. | 52 |
| Total | 41,533 |

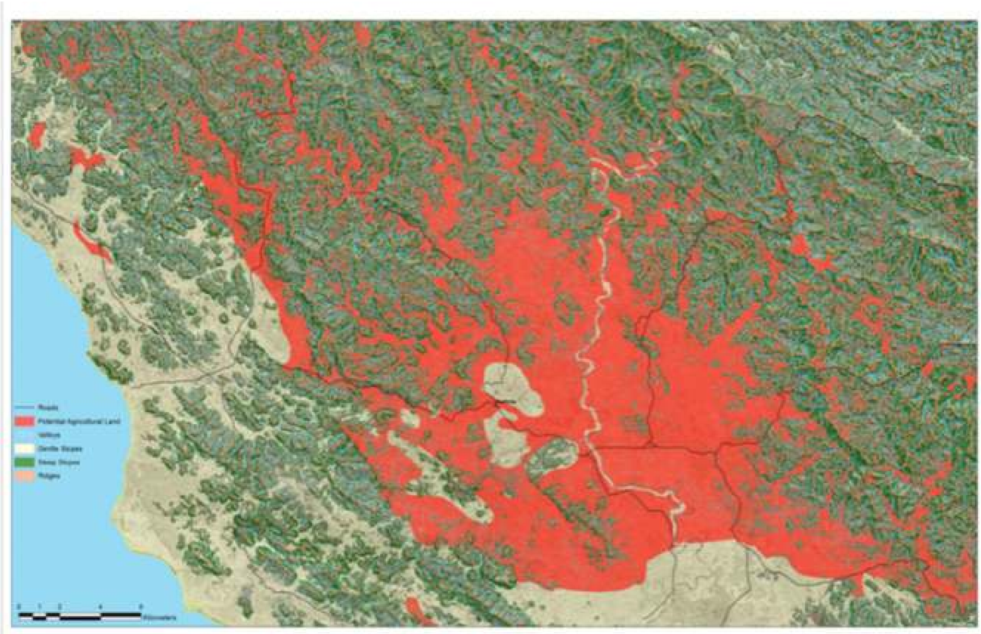

**Figure 1: Potential suitable Agricultural Land (areas in red) in the Rigo Province PNG**

The majority of this land is centred on the town of Kwikila some two hours by road south east from Port Moresby. Of the 41,533 ha identified some 10,632 ha is listed within PNGRIS as being prone to waterlogging and inundation of varying duration and severity (Table 2). Of the 10,632 ha susceptible to inundation 2,920 ha (Inundation types 1, 2 and 5 would probably be excluded from agricultural use).

**Table 2: Areas of land subject to inundation in Rigo district**

| Inundation Type | Area (Ha) |
|---|---|
| 1. Long term inundation | 2,843 |
| 2. Near permanent inundation | 6 |
| 3. Periodic brief flooding | 3,909 |
| 4. Seasonal inundation | 3,607 |
| 5. Tidal flooding | 71 |
| Total | 10,632 |

## Discussion

The GIS data and radar imagery has been combined to produce informative maps that can be used to prioritise areas for agricultural development. They have clearly identified potential areas, and by relaxing or tightening the constraints set when using the radar imagery, the area of potentially useable land would increase or decrease. For example, if the allowable slope was reduced to say, 7$^{o}$ for agricultural systems in which soil cover was limited between crops and during cropping while canopy cover was limited, therefore increasing the erosion risk, the area of suitable land would inevitably decrease. Conversely, if the assessment was being made for land uses involving perennial pastures, forestry and fruit trees the allowable slope limits could be increased, resulting in larger areas of potentially useful land being identified.

The present analysis has shown an extensive area of potentially useful land between elevated areas and many small, narrow areas of potentially useful land in nearby valleys and small areas in elevated areas. When combined with ground based observations along roads and in village gardens augmented with limited examination of augured soil profiles, the approach of using GIS and radar imagery is proving a very useful tool for assessment of land capability. Our team has also applied this approach in several other areas in Central Province, with similarly useful output, again with initial validation from ground based observations and soil data. Nevertheless, the approach must be complemented with other information, such as the data on inundation, to gain a more accurate assessment of land capability and guide development and agronomic decisions on crops to be grown and practices used on specific sites.

Clearly, there are significant areas of land available for agricultural development in Rigo district, and by extrapolation, other areas of Central Province and beyond. However, for effective development, appropriate agronomic practices will need to be developed; these being part of other work being conducted by the authors and other colleagues. Land tenure issues notwithstanding these data will assist the sustainable agricultural development process in PNG and increase employment and business prospects for local farmer cooperatives.

## Conclusions

We recommend using high resolution radar generated topographic coverage's in combination with soil parent material classification based on the mapped bedrock lithology as a base to generate more reliable land suitability maps and show national and local government development bodies, aid agencies and the village farmer cooperatives the areas available for land use intensification and sustainable national agricultural development. The maps would then be combined with other local data to provide a sound basis for development decisions and to guide agronomic practice.

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# Thursday 6 December

## SOIL CARBON AND CLIMATE CHANGE (LAND USE EFFECTS)

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# Land use effects on carbon saturation in Ferrosols of northern New South Wales

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### Abstract

Soil protective capacity and saturation deficit of organic carbon of 12 soil samples (Ferrosol), collected from three land uses (improved pasture, cropping and forest) was calculated using equation $y=6.67\ e^{0.0216\,x}$ where y is the protective capacity (C content of the silt + clay fraction expressed on a whole soil basis) and $x$ is the dispersed clay plus silt content (<53 um) (%). Saturation deficit is the difference between the protective capacity and the actual amount of C in the particle size fraction <53µm. Among the soil samples, improved pasture and cropping soils had a higher saturation deficit than forest soils. The larger saturation deficit most likely relates to the more intensive land use and loss of carbon through mineralisation processes associated with disturbance and reduced carbon inputs. However, two of the forest soils exceeded their theoretical carbon saturation limit by between 40 and 100% and work is ongoing to determine how that may occur.

### Key Words

carbon saturation, protective capacity, saturation deficit

### Introduction

Soil carbon saturation implies that soils have a finite capacity to stabilize and store organic carbon. The capacity of the soil to store C increases linearly with increasing dispersed silt+clay content of soils (Hassink 1997). Other factors, such as the chemical nature of the soil mineral fraction, presence of multivalent cations, and chemical nature of organic matter also influence organic carbon stabilization (Baldock and Skjemstad 2000). The level of C saturation also varies depending on land use (Six *et al*. 2002). Since multiple factors influence the protection of carbon in soil, the protective capacity and the degree of saturation of this capacity are complex to determine. The protective capacity determined by Hassink (1996) was applicable to soils of different regions of the world, except Australian soils. High temperature and low annual rainfall may account for Australian soils not reaching their maximum carbon storage capacity in some regions (Hassink 1996; Hassink 1997). However, the observed exponential rather than linear relationship between dispersed silt+clay content and C in Tasmanian Ferrosols may be due to their higher silt+clay content, lower temperature and high carbon inputs from pastures, resulting in a larger capacity to store soil carbon (Sparrow *et al*. 2006). We investigated the effect of land use on carbon saturation in northern NSW Ferrosols in warmer environments than the Tasmanian scenario.

### Methods

*Site information and soil sampling*

The soils analysed in the study were Ferrosols under three types of land use (improved pasture, cropping and forest). The soils were collected from the Dorrigo (elevation 746 m) region of the Northern Tablelands, NSW. Mean annual minimum temperature is 9.9°C, mean annual maximum temperature is 19.8°C, and the annual rainfall is 2074 mm (Bureau of Meteorology 2010). Soil samples (0-10 cm depth) were collected from each land use by selecting 3 separate blocks (50×50 m) along the slope of each paddock. In each block 10 random samples were collected and then composited into one.

*Soil analysis*

Particle size analysis was determined using the pipette method with sodium hexametaphosphate (5 g $L^{-1}$) as a dispersing agent and then 5 minutes sonification for complete dispersion of soils. The dispersed soil suspension was wet-sieved with distilled water through a 53 µm sieve and the solution dried at 40°C. The dried fraction was then homogenized by grinding with mortar and pestle and passed back through a 500 µm sieve to determine carbon content using LECO CNS analyser. Total organic carbon of the whole soil was also analysed. The C concentration of the silt+clay (<53 um) fraction was expressed on a whole soil basis (g $kg^{-1}$ soil). Soil pH was measured using 1:5 soil: water ratio. CEC was determined using 1M $NH_4Cl$ solution (Rayment and Higginson 1992).

*Protective capacity calculation*

The protective capacity of soils (maximum amount of C associated with silt and clay particles) was determined using the equation of Sparrow *et al*. (2006): $y$=6.67 $e^{0.0216x}$ ($r^2$ = 0.61, $p$ < 0.001) where y is the protective capacity and $x$ is the chemically dispersed silt + clay fraction (<53 um) (%).

*Saturation deficit calculation*

The difference between the theoretical value of carbon saturation and measured value of carbon in silt +clay fraction represents the soil carbon saturation deficit. It also represents the potential for soil to sequester C in stable form. This theoretical value of C saturation was also expressed as the C content of silt + clay fractions on a whole soil basis (g C $kg^{-1}$ soil).

## Results

The forest soils contained more total organic carbon and hence carbon associated with each size fraction, than improved pasture and cropping soils (Table 1). Two of the four forest soils had stored organic carbon up to their theoretical protective capacity and two had stored 30% and 100% more C than their theoretical capacity (Table 2). Cropping and improved pasture soils were theoretically undersaturated with carbon by between 20% and 40% on average (Table 2). The larger saturation deficit most likely relates to the more intensive land use and loss of carbon through reduced carbon inputs and mineralisation processes associated with disturbance.

## Conclusion

The effects of land use on the soil carbon saturation level are related to 3 main factors: amount of carbon input, level of disturbance and type of carbon (which is related to plant species). The first two factors determine whether cropping soils reach carbon saturation. However, carbon saturation in forest soils is affected by all 3 factors. Further research is necessary to explain the 'oversaturation' of the two forest soils with respect to silt+clay percentages. For example, temperature and/or rainfall, type of clay (2:1 or 1:1), and content of multivalent cations, amorphous or crystalline Fe and Al oxides might increase carbon saturation above theoretical limits. More research is necessary but this study demonstrates that under most circumstances, the theoretical saturation limits fall within those described by Sparrow *et al*. (2006) for Ferrosols.

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**Table 1. Selected characteristics of studied soils (0-10 cm)**

| Site | Land use | pHw (1:5) | CEC (cmol kg-1) | Particle-size distribution (%) | | | | C on particle (g kg-1 soil) | | | TOC (g kg-1 soil) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | <2 µm | <20 µm | <53 µm | 53-2000 µm | <2 µm | <53 µm | 53-2000 µm | |
| North Dorrigo | Improved Pasture | 6.1 | 31.1 | 60 | 85 | 94 | 6 | 31.3 | 47.4 | 12.7 | 60.1 |
| Dorrigo1 | Improved Pasture | 5.7 | 16.0 | 57 | 90 | 96 | 4 | 22.6 | 35.3 | 10.8 | 46.0 |
| Dorrigo 2 | Improved Pasture | 5.8 | 13.5 | 58 | 85 | 96 | 4 | 23.4 | 34.5 | 9.00 | 43.5 |
| Dorrigo 3 | Improved Pasture | 5.7 | 11.4 | 62 | 86 | 96 | 4 | 20.0 | 29.4 | 5.0 | 34.3 |
| North Dorrigo | Cropping | 5.6 | 26.6 | 54 | 84 | 97 | 3 | 23.7 | 42.2 | 10.8 | 53.0 |
| Dorrigo1 | Cropping | 5.8 | 14.8 | 39 | 56 | 86 | 14 | 15.6 | 28.2 | 8.6 | 36.9 |
| Dorrigo 2 | Cropping | 5.7 | 14.4 | 56 | 85 | 95 | 5 | 27.1 | 38.7 | 11.7 | 50.4 |
| Dorrigo 3 | Cropping | 5.4 | 11.5 | 66 | 87 | 96 | 4 | 23.8 | 33.5 | 7.0 | 40.5 |
| North Dorrigo | Forest | 5.2 | 10.7 | 64 | 91 | 94 | 6 | 31.3 | 50 | 23.9 | 73.8 |
| Dorrigo1 | Forest | 5.3 | 9.6 | 54 | 90 | 95 | 5 | 33.9 | 51.1 | 22.8 | 73.8 |
| Dorrigo 2 | Forest | 5.2 | 8.2 | 55 | 84 | 91 | 9 | 42.2 | 66.2 | 18.4 | 84.6 |
| Dorrigo 3 | Forest | 6.0 | 71.2 | 56 | 84 | 90 | 10 | 49.8 | 86.9 | 23.4 | 110.3 |

**Table 2. Calculated protective capacity and saturation deficit**

| Site | Land use | Protective capacity (g kg$^{-1}$ soil) | Saturation deficit (g kg$^{-1}$ soil ) |
|---|---|---|---|
| North Dorrigo | Improved pasture | 50.8 | -3.4 ±0.07 |
| Dorrigo1 | Improved pasture | 53.1 | -17.8 ±0.09 |
| Dorrigo 2 | Improved pasture | 52.7 | -18.2 ±0.23 |
| Dorrigo 3 | Improved pasture | 53.0 | -23.6 ±0.06 |
| North Dorrigo | Cropping | 54.0 | -11.8 ±0.05 |
| Dorrigo1 | Cropping | 42.8 | -14.6 ±0.05 |
| Dorrigo 2 | Cropping | 51.9 | -13.2 ±0.03 |
| Dorrigo 3 | Cropping | 52.9 | -19.4 ±0.17 |
| North Dorrigo | Forest | 50.7 | -0.7 ±0.07 |
| Dorrigo1 | Forest | 51.8 | -0.7 ±0.17 |
| Dorrigo 2 | Forest | 51.9 | 14.4 ±0.23 |
| Dorrigo 3 | Forest | 46.2 | 40.7 ±0.29 |

# The effect of agricultural land use on the soil carbon fractions of Red Ferrosols in North West Tasmania

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### Abstract

This research forms a part of a long term study, in which the SOC (soil organic carbon) content of Tasmanian Red Ferrosols was measured to determine the extent of management related change. Composite sampling was conducted at two depths (0-150 mm and 150-300 mm) over a total of 25 sites in northern Tasmania, both in 1997 and 2010. In addition to total organic carbon (TOC), the soil was divided into two size fractions: POC >50µm (particulate organic carbon) and HUM <50µm (humic organic carbon). The percentage of TOC decreased with increasing years of cultivation. The proportions of POC and HUM decreased at uniform rates compared to the TOC, suggesting that they are both prone to the same degree of depletion under cultivation. At the depth of 0-150 mm, the mean TOC in 2010 was significantly higher for sites classed as predominately pasture than for those sites classed as continuously cropped. At both depths, there were no significant changes in the percentages of TOC, or proportions of POC and HUM for either land use category between 1997 and 2010. The lack of change in the proportions of POC to HUM indicates that both fractions are affected evenly under both cropping and pasture. In other studies the SOC associated with the HUM fraction has typically been found to be far more resistant to depletion than that of the POC. The finding that both HUM and POC are uniformly affected by management suggests that Red Ferrosols may differ from other soil types in regard to carbon storage in the HUM fraction. The properties of these soils which may be responsible for the short term storage of SOC in the HUM fraction are the kaolinite clay type and the high content of iron and aluminium oxides. Further research is required to improve the understanding of the storage mechanisms of SOC in Red Ferrosols, particularly in the HUM fraction.

### Key Words
Ferrosol, soil organic carbon, particulate organic carbon, fractions, land use

### Introduction

The world's soils have been both a source and sink for atmospheric $CO_2$. Most agricultural soils have lost 30% to 75% of their antecedent soil organic carbon (SOC) pool or approximately 30 to 40 t C $ha^{-1}$ (Lal *et al.* 2007).

Models such as Rothamsted Organic Turnover Version 26.3, view the SOC dynamic as comprising several pools with varying half lives. In this study a modified version of the fractionation method employed by Skjemstad *et al.* (2004) was adopted to look at the changes in the labile particulate organic matter (POM) pool and the recalcitrant humate pools (HUM) in 25 Ferrosol soils in northern Tasmania. These soils were first sampled in 1997 (Sparrow *et al.* 1999).

### Methods

In 2010, soil samples were taken from 25 sites previously sampled in 1997. Individual sites were allocated to groups based on the dominant agricultural land use between 1997 and 2010. The two main land use categories were 'continuously cropped' and 'predominantly pasture'. At each site, soil samples were taken at 20 points within a grid at depths of 0-150 and 150-300 mm using a 100 mm Jarret auger. The 20 samples for each depth were combined, air dried at 40°C and sieved to 2 mm. Acid (1 M HCl) was added to a small subsample to determine the presence of carbonates. Oven dry moisture corrections were done at 105°C for 24 h. Bulk density was taken at three separate sites within the grid using 75 mm rings. A small subsample of each composite (5-10 g) was ground for 2 minutes using a Retch MM200 ball mill. Fine ground soil (20-30 mg) was then analysed using a Perkin Elmer CHN-S 2400 oxidative combustion analyser to determine TOC. The Walkley Black method was used for TOC determination on the original samples and for confirmation on the 2010 samples. Several steps were involved in the process of dividing each of the 100 samples into the of POC (>50 µm) and HUM (<50 µm) size fractions. Each sample was 'disaggregated' by adding 20 g of soil and 90 mL of 5 g/L sodium hexametaphosphate solution to a 250 mL plastic container. The container was then placed on its side and shaken horizontally at 180–200 rpm for 12–16 hours by use of a Gio Gyrotory ® Shaker. Fractionation at 50 μm was carried out by use of the Fritch Analysette 3 vibratory wet sieving device. Operating parameters were as follows: 20 second interval, 3 minutes minimum sieving time, 2.5 mm amplitude, and a water spray at a rate of approximately 150 mL $min^{-1}$. The fractions were then dried at 40°C, re-ground and analysed for TOC using oxidative combustion.

**Results**

There were significantly ($P < 0.001$) higher TOC levels in pasture soils than cropped soils at 0–150 mm in both 1997 and 2010 (Fig 1). The mean TOC for continuously cropped sites was 4.4 %, whereas the mean for predominately pasture sites was 6.6 %. The TOC was 33% higher under pastures in 1997 and 50% higher in 2010. At 150-300 mm there were no significant differences in the mean values of TOC between these categories in either year. Therefore, the type of land use has had a larger, and significant, influence on TOC in the surface 0-150 mm depth than below 150 mm in Red Ferrosols. At the depth of 0-150 mm there were no significant changes in the mean percentage of TOC between 1997 and 2010 within either the cropping or pasture land use categories (Fig. 1). However TOC in continuously cropped sites decreased proportionally by 10%, while TOC in predominantly pasture sites increased proportionally by 1.5 % over the 13 year period.

There were no significant changes in the proportions of POC or HUM between 1997 and 2010, in either land use category (Fig. 2). The proportion of POC decreased by 18.2 % in cropping soils, and decreased by 19.2 % in pasture soils. The proportion of HUM showed an increase of 5.1 % in cropping soils and an increase of 3.9 % in pasture soils (Fig. 2). Fig 3. shows the weak trends in changing ratios of POC to HUM at the two depths vs. the intensity of cropping, measured as total years of cropping in the last 38 years. The weak trends suggest an increase in the ratio of topsoil (0 -150 mm) HUM at the expense of POC as cropping frequency increases.

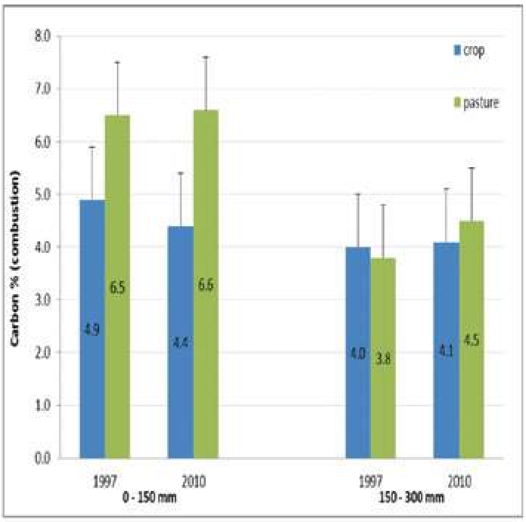


**Figure 1. Mean TOC difference between 1997 and 2010 for continuously cropped and predominately pasture sites. Error bars = standard deviation of mean.**

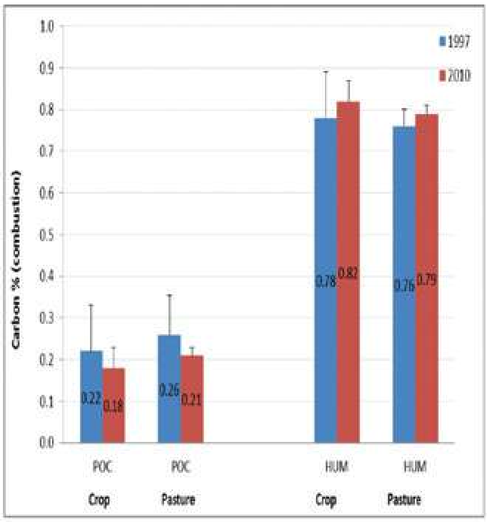


**Figure 2. Mean carbon (%) change in POC and HUM as proportion of the TOC between 1997 and 2010 for continuously cropped and predominately pasture sites, 0 - 150 mm depth.**

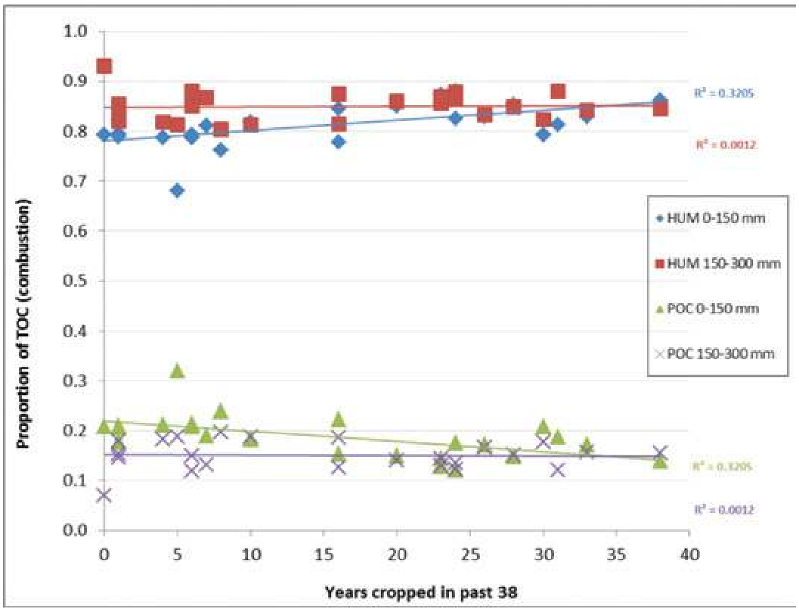


**Figure 3. Relationship between proportion of POC and HUM in TOC and cropping frequency.**

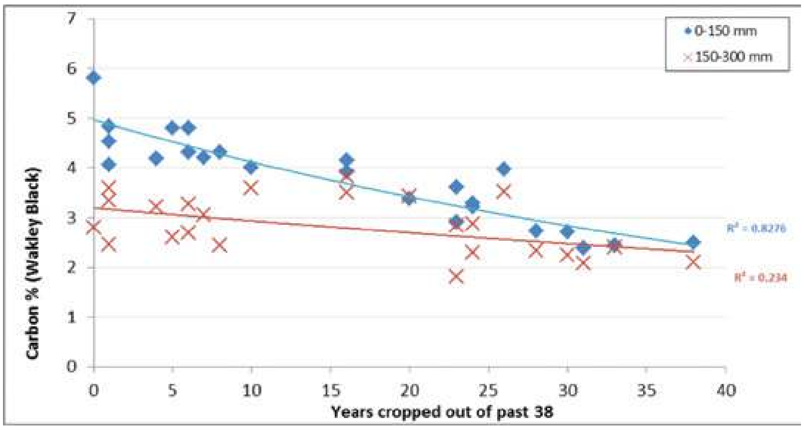


**Figure 4. TOC (%) in 2010 at both depths, plotted against cropping frequency.**

Fig. 4 shows that TOC in both the 0–150 and 150–300 mm depths decreased as the cropping frequency increased. However the trend in the surface was much stronger ($r^2$ = 0.83 compared to 0.23).

### Conclusions

There are four main conclusions that can be drawn from this study. Firstly, there were significant differences in TOC in the 0–150 mm depth between the two land uses measured in both 1997 and 2010. Compared to cropped soils, pasture land had 33% and 50% more carbon respectively in 1997 and 2010.

Secondly, despite a declining trend of TOC in cropping soils and higher concentrations of TOC in pasture soils, no significant change was measured in TOC concentrations between 1997 and 2010. This suggests that soil TOC concentrations are slow to change in established management systems. From the perspective of carbon sequestration a quick fix is unlikely.

Thirdly, there were no significant changes in the proportion of POC to HUM in either management system during the thirteen years of this study. This may be the result of the humate fraction not being well protected within the clay fraction of Red Ferrosols which is dominated by kaolin and iron oxides, or alternatively the POC fraction in the 1997 samples may have partially oxidised during storage.

Fourthly, this study would not have been possible without the archived 1997 samples. The availability of archived samples to future research in soil carbon will be invaluable.

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# Modelling tillage impact on soil organic carbon dynamics

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## Abstract

Tillage may have significant impact on soil organic carbon (SOC) dynamics in agricultural soil. Here, we used the APSIM farming systems model in combination with measurement data to test two hypotheses for tillage impact on SOC dynamics: 1) tillage increases decomposition rate of SOC and 2) tillage exposes protected SOC to decomposers. Adopting either of the two hypotheses, APSIM was able to simulate the SOC dynamics under four tillage intensities in a field experiment at Biloela. Otherwise, APSIM overestimated SOC under three intensive tillage treatments. However, without explicit understanding of mechanisms of SOC stabilization, it is difficult to judge the relative importance of the two hypotheses. Scenario analysis suggested that the two hypotheses had distinct effects on simulated long-term SOC dynamics, particularly under non-fertilization.

**Key Words:** Soil organic carbon, decomposition rate, carbon fraction, tillage, APSIM

## Introduction

Tillage may result in significant changes in surface soil conditions, such as soil bulk density, soil water and nutrient dynamics, and subsequently impact on dynamics of soil organic carbon (SOC). However, the tillage impact on SOC has been poorly captured in most SOC models, including APSIM, which leads to unreliable predictions of long-term SOC change under different tillage regimes.

There are mainly two hypotheses to explain the effect of tillage on SOC decomposition. **Hypothesis 1:** Tillage incorporates crop residue and surface SOC into the top soil layer. This not only provides nutrients and energy for soil microbes (Fontaine *et al*. 2007), but also mixes carbon into more favourable water and thermal conditions in the soil (Coppens *et al*. 2006). The end result is an increase of SOC decomposition rate. **Hypothesis 2:** Tillage fragments macro-aggregates and increases the surface area for soil microbes to attack and decompose the physically aggregate-protected SOC (Six *et al*. 1999; Six *et al*. 2000). This process would lead to the exposure of protected SOC to decomposers.

In this study, we aimed to: (1) test the performance of the current APSIM model in simulating observed SOC changes under four tillage intensities in a field experiment at Biloela, (2) examine whether APSIM can predict SOC changes by considering the impact of tillage intensity on either SOC decomposition rate (Hypothesis 1) or on exposure of protected SOC (Hypothesis 2) through inverse modelling, and (3) assess the difference in simulated long-term SOC dynamics under the two hypotheses through scenario analysis.

## Methods

### *Study site and experimental measurements*

The data collected by Radford and Thornton (2011) were used for this study. The experimental site was located at Biloela (24.37°S, 150.52°E) in Queensland of Australia. In April 1983, four tillage treatments, tradition tillage (TT), stubble mulch tillage (ST), reduced tillage (RT) and no till (NT), were conducted with four replications (four 72 m × 22 m plots). During the whole period of the experiment until 2003, treatments TT, ST, RT and NT received 83, 55, 44 and 0 tillage operations during the fallow period, respectively. Crop seeds were sown using a no-till planter at sowing time in all treatments. In 1989, the four replications for each tillage treatment were split into two fertilisation treatments, two for control with no nitrogen (N) applications and the other two for N applications. N was applied as urea at rates (kg N $ha^{-1}$) of 100 (in 1989), 80 (1990), 120 (1991), 75 (1994), 50 (1997), 50 (1998), 40 (1999), 50 (2000) and 40 (2002). After crop harvest in 2003, all tillage treatments were converted to uniform zero tillage (NT) across the entire trial area. Nitrogen was applied as urea at rates of 40 (in 2005), 50 (2006), 70 (2007) and 40 (2008) kg N $ha^{-1}$ as blanket applications to all plots. A wheat-sorghum rotation was adopted in the experiment. All crop residues were retained *in situ* after harvest during the whole experiment.

Crop grain yield was recorded for each cropping year. Detailed physical and chemical properties in the soil profile were measured in 1984 after about 1.5 years since the start of the experiment. Soil organic carbon content in the 0–10 and 10–20 cm soil layers was also determined in each plot by thoroughly mixing five soil cores on 13 April 1989, 14 May 1997, 15 September 2004 and 3 April 2008 (only fertilised NT and TT plots in the 0–10 cm layer in 2008). Detailed description of the experimental design and measurements were reported in Radford and Thornton (2011).

*APSIM parameterisation and simulations*

APSIM (Keating *et al.* 2003) v7.3 was used for simulating the SOC dynamics in each treatment. Daily climate data from 1889 to 2011 at Biloela were obtained from SILO Patched Point Dataset (http://www.longaddock.qld.gov.au/silo/). The measured soil data in 1984 were used to initialize the soil parameters in each soil layer, including soil organic carbon, bulk density and soil hydraulic properties. These data were measured after 1.5 years application of tillage treatments and would capture the possible impacts of tillage treatments on soil properties. Field management was set according to experimental records. Crop cultivar parameters were adjusted to match the simulated and observed flowering and maturity dates.

Total SOC in each soil layer for each tillage treatment was partitioned into four soil organic matter pools in APSIM, fresh organic matter (FOM), microbial biomass (BIOM), humus (HUM) and inert C based on inverse modelling according to Luo *et al*. (2011). In APSIM, the inert C pool is considered to be non–susceptible to decomposition. Hereafter, we refer to inert C as protected C and assume that tillage would possibly expose a fraction of this to decomposers. In cultivated soils, FOM (roots and incorporated plant materials) and BIOM only account for a small fraction of SOC, and HUM and protected C account for the majority. We set the initial FOM pool size in the whole soil profile to 0.6 t C ha$^{-1}$, which is 50% of the annual above-ground dry matter returned to the soil (1.2 t C ha$^{-1}$ year$^{-1}$ equivalent to ~3 t biomass ha$^{-1}$ year$^{-1}$). The BIOM pool was set to 2.6% of HUM according to a number of field studies in agricultural soils (Zimmermann *et al.* 2007).

*Test of model performance and the two hypotheses*

In this experiment, most of the tillage operations were applied to the soil depth of 10 cm. In the deeper soil layers (> 10 cm) which were not disturbed by tillage, we assumed that the fraction of each C pool was the same under the four tillage treatments. For SOC dynamics in the top 10 cm soil layer, we initialised the model in three different ways to do the following:

A. Test model performance in simulating SOC dynamics using the APSIM default decomposition rates.

B. Derive the maximum decomposition rate of HUM for each tillage treatment which leads to the best match of simulated and observed SOC (Hypothesis 1). The fraction of protected C was set the same as in A.

C. Derive the fraction of protected C for each tillage treatment which leads to the best match of simulated and observed SOC (Hypothesis 2). The decomposition rate was set the same as in A.

Option A gives the baseline performance of APSIM. We expected that in option B, the decomposition rate of HUM increases with increasing tillage intensity (NT<RT<ST<TT), while in option C, the fraction of protected C decreases with tillage intensity (NT<RT<ST<TT). Best fit of the model was achieved by

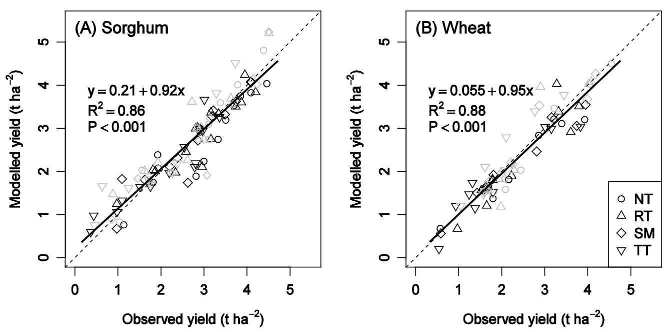


**Figure 1. Observed and modelled grain yields of sorghum (A) and wheat (B) under four tillage treatments. NT: no tillage, RT: reduced tillage, SM: stubble mulch tillage, TT: traditional tillage. For the same tillage treatment, black and grey symbols show the non-fertilization and fertilization treatments, respectively. Dashed line shows the 1:1 line.**

*Test model performance in simulating SOC dynamics using the APSIM default decomposition rates.*

Derive the maximum decomposition rate of HUM for each tillage treatment which leads to the best match of simulated and observed SOC (Hypothesis 1). The fraction of protected C was set the same as in A.

Derive the fraction of protected C for each tillage treatment which leads to the best match of simulated and observed SOC (Hypothesis 2). The decomposition rate was set the same as in AOption A gives the baseline performance of APSIM. We expected that in option B, the decomposition rate of HUM increases with increasing tillage intensity (NT<RT<ST<TT), while in option C, the fraction of protected C decreases with tillage intensity (NT<RT<ST<TT). Best fit of the model was achieved by minimising the root mean square error between simulated and observed SOC.

Finally, scenario analyses were conducted using a continuous wheat system from 1889 to 2011 to investigate the impact of changed decomposition rate (derived from option B) or fraction of protected C (derived from option C) on simulated long-term SOC dynamics under no tillage and traditional tillage. Two nitrogen application levels, 0 and 150 kg N ha$^{-1}$, were also used in each scenario to investigate the impact of N inputs.

## Results

In general, grain yields of wheat and sorghum were well simulated by the APSIM model. Using the derived decomposition rates of HUM (Option B) under four tillage intensities, APSIM explained 86% and 88% of the observed variation of sorghum and wheat yields (Fig. 1), respectively. Generally, there were no apparent differences in crop yields among different tillage intensities. Similar model performance was also achieved using the derived fraction of protected C (Option C) in response to tillage intensities (data not shown).

Fig. 2 showed the model performance in simulating SOC. The initial SOC contents were different among the four tillage treatments (Fig. 2), which may reflect the impact of tillage on soil bulk density and redistribution of SOC. Using the derived fraction of protected C under NT treatment and the default decomposition rate of HUM (Option A), APSIM overestimated SOC under the other three tillage treatments, particularly under SM and TT (Fig. 2). To achieve the best model performance, the maximum decomposition rate of HUM needs to be increased with tillage intensity, i.e., 1.5e–4, 1.6e–4, 2.2e–4 and 3.5e–5 per day under NT, RT, SM and TT, respectively (Fig. 3). If a fixed HUM decomposition rate was used, the fraction of protected C has to be decreased with tillage intensity, i.e., 0.5, 0.45, 0.3 and 0.04 under NT, RT, SM and TT, respectively (Fig. 3).

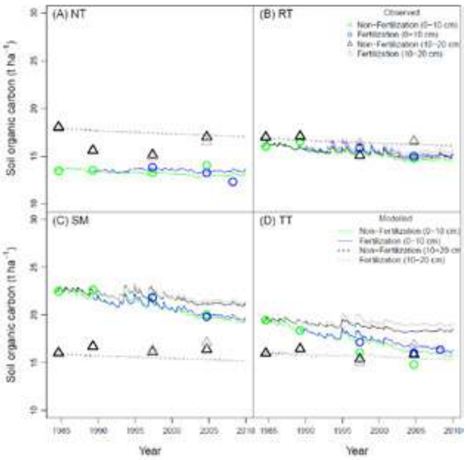


**Figure 2. Modelled (lines) and observed (points) soil organic carbon content in the 0–10 (solid lines) and 10–20 cm (dashed lines) soil layers under four tillage treatments from 1984 to 2009. NT: no tillage, RT: reduced tillage, SM: stubble mulch tillage, TT: traditional tillage. The solid black and grey lines show the simulated results based on default decomposition rate and the fraction of protected C derived under NT.**

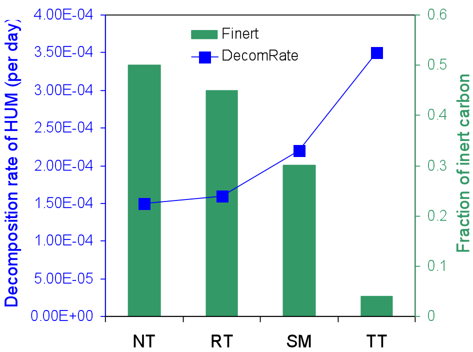


**Figure 3. Best-fitted decomposition rates of HUM pool based on hypothesis 1 and fraction of inert carbon based on hypothesis 2 under four tillage treatments. NT: no-tillage, RT: reduced tillage, SM: stubble mulch tillage, TT: traditional tillage.**

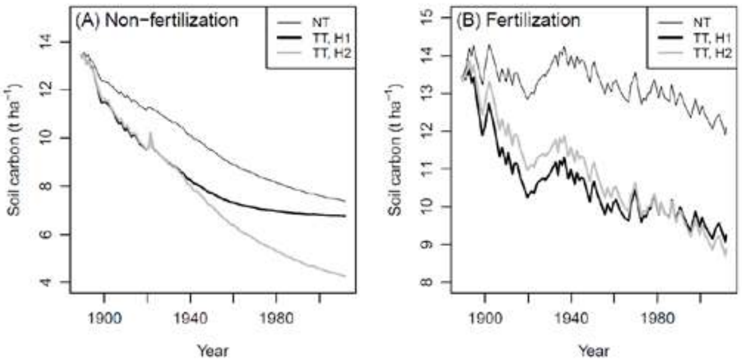


**Figure 4. Modelled long-term SOC dynamics in a continuous wheat system under non-fertilization (A) and fertilization (B). NT, no–tillage; TT, traditional tillage. Details see text for Hypothesis H1 and H2.**

Although the two approaches (relating either decomposition rate or fraction of protected C to tillage intensity) led to similar simulated SOC dynamics in the 25 years, they had distinct effects on simulated long-term (>100 years) SOC dynamics (Fig. 4). Under zero N input, the two approaches led to similar results on simulated impact of tillage on SOC in the first 50 years. Thereafter, the approach reducing the fraction of protected C with tillage intensity led to much bigger decreases in SOC (Fig. 4A). Under an N rate of 150 kg N $ha^{-1}$ $year^{-1}$, the simulated influences of tillage on SOC under two hypotheses were comparable (Fig. 4B).

## Discussions and Conclusion

Our results indicated that tillage impacts on SOC dynamics need to be considered in SOC models. After 1.5 years of tillage practice, SOC in the top soil showed great difference among tillage treatments, although the four treatments had similar initial soil properties and agricultural management (Fig. 2). The fraction of protected C derived under specific tillage intensity could not extrapolate to other tillage treatments. By relating either the HUM decomposition rate or the fraction of protected C to tillage intensity, the time course of SOC change could be well simulated. Otherwise, APSIM was not able to capture the SOC variations in different tillage treatments. However, the current available data do not permit an investigation of the relative importance of the two approaches, which have distinct effects on long-term SOC dynamics. The generality of the two approaches also needs to be evaluated when extrapolating to other sites and/or regions.

## Acknowledgments

Founding supports from the Grain Research and Development Corporation (GRDC) and the Department of Agriculture, Fisheries and Forestry (DAFF) are gratefully acknowledged.

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# Large nitrous oxide emissions after conversion from pasture to cropping in temperate south eastern Australia

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Agricultural systems in transition from pasture to cropping can be large emitters of nitrous oxide ($N_2O$) and the identification of mitigation methods is a priority. In this study, we measured $N_2O$ emissions after conversion of a high fertility pasture to winter wheat (*Triticum aestivum*) in temperate south-eastern Australia, following a transition year of subterranean clover (*Trifolium subterraneum*) dominated sward. Two potential mitigation techniques were tested: reduced tillage and the application of a nitrification inhibitor, dicyandiamide. Nitrous oxide flux was measured every 90 minutes for 12 months using a tuneable diode laser and automated chamber enclosures located on plots of four agronomic treatments; (1) Direct drilled wheat (2) Wheat sown after conventional cultivation (3) Direct drilled wheat with dicyandiamide (10 kg/ha) applied immediately prior to sowing and mid season (4) Wheat sown after conventional cultivation and dicyandiamide (10 $kg.ha^{-1}$) applied immediately prior to sowing and mid season. No nitrogen fertilizer was applied. The measured $N_2O$ flux was highly variable but covariance among the large number of measurements made each day enabled the fitting of cubic smoothing splines and the evaluation of treatment differences by linear mixed model analysis. The reduced tillage and dicyandiamide treatments demonstrated short periods of effectiveness but the mitigation was not significant when considered over the full year. Unexpectedly large nitrous oxide emissions were measured. Cumulative emissions over one year (1 February 2010 to 31 January 2011) averaged 35 kg $N_2O$-$N.ha^{-1}$, equivalent to 17 t $CO_2.ha^{-1}$ or 4.7 t $C.ha^{-1}$.

# Relationship between soil relative gas diffusivity and ruminant urine-derived $N_2O$ and $N_2$ emissions from a soil under varying bulk densities

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Soil relative gas diffusivity ($D_p/D_o$) has been shown to be a good predictor of $N_2O$ emissions. However, only a few studies have attempted to relate direct measurements of $D_p/D_o$ to $N_2O$ emissions and there are none from compacted and ruminant urine affected soils. The aim of this study was to determine the relationship between $D_p/D_o$ and ruminant urine-derived cumulative $N_2O$-N and $N_2$-N fluxes from a soil under varying bulk densities (1.1, 1.2, 1.3, 1.4 and 1.5 Mg $m^{-3}$) maintained at varying matric potentials (-0.2, -6.0 and -10 kPa) for a period of 35 days. To simulate urine application, urea was applied at a rate of 700 kg-N $ha^{-1}$ after a silt loam soil had been compacted uniaxially and packed at the varying bulk densities. Increasing soil bulk density reduced $D_p/D_o$ regardless of the soil's matric potential. The relationship between cumulative $N_2O$-N fluxes and $D_p/D_o$ differed with soil matric potential. As the soil matric potential increased from -10 to -6.0 kPa, cumulative $N_2O$-N fluxes declined with a decrease in $D_p/D_o$ while cumulative $N_2$-N fluxes increased. A decrease in $D_p/D_o$ to below a threshold value of 0.038 resulted in significantly higher cumulative $N_2$-N fluxes. Mean soil cumulative $N_2$-N: $N_2O$-N ratios ranged from 0.8 to 25 at 1.1 and 1.5 Mg $m^{-3}$, respectively. The study found that both $D_p/D_o$ and WFPS produced similar relationships with cumulative $N_2$-N: $N_2O$-N ratios.

# Thursday 6 December

## SOIL FERTILITY AND SOIL CONTAMINANTS

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# The forms of plant nutrient elements of Australian native species in ash

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### Abstract

This study characterized the ash of several Australian native plant species**:** silver wattle (*Acacia retinodes*), prickly Moses (*Acacia pulchella*), wandoo/white gum (*Eucalyptus wandoo*), red gum or marri (*Corymbia calophylla*), grass tree (*Xanthorrhoea preissii*), jarrah (*Eucalyptus marginata*) and harsh hakea (*Hakea prostrata*) and investigated the reactions of ash with soil. The ashes were alkaline with pH ranging from 12.3 to 13.8 and contained substantial amounts of plant nutrients. Amounts of calcium, magnesium, sodium and silicon varied depending on plant species and plant part. Many minor elements are also present including elements with no biological function. Available phosphorus (Colwell–P) in the ashes ranged from 30 to 199 mg/kg. All elements are mostly present in crystalline compounds which were identified using XRD and SEM. Minerals present in ash include calcite ($CaCO_3$), fairchildite ($K_2Ca(CO_3)_2$), nesquehonite ($MgCO_3.H_2O$), sylvite (KCl), lime (CaO), scolecite ($CaAl_2Si_3O_{10}.3(H_2O)$), quartz ($SiO_2$) (derived from dust), portlandite ($Ca(OH)_2$), periclase (MgO) and one apatite, probably resembling hydroxyl-apatite ($Ca_5(PO_4)_3(OH)$) and wilkeite ($Ca_5((P, S, Si)O_4)_3(OH,CO_3)$). Clearly the fertiliser effectiveness of ash will depend on the solubility of these minerals in soil solution. Ash has important liming and fertilizer values and its effectiveness in these roles is a consequence of the properties of these minerals and their reaction with soil.

### Key Words

Ash, mineralogy, water-soluble, synchrotron-XRD.

### Introduction

Wild and managed fires burn many thousands of hectares of forest annually and consume some or all of the vegetation and litter (Ulery *et al*., 1996). The intensity of forest burning is related to weather conditions, amount of fuel available and the condition of the fuel. Fires create ash consisting of organic and inorganic residues from the combustion process (Ùbeda *et al*., 2009). Ash may be defined as a complex mixture of charcoal, char and minerals (i.e. inorganic compounds) (Scott, 2010). The characteristics and the amount of forest plant ash produced under natural or controlled burning depend on several factors, including the species and the part of plant that was combusted (leaves, fruit, bark or wood), plant age and extent of combustion (Clapham and Zibilske, 1992; Ulery *et al*., 1993; Vance and Mitchell 2000; Demeyer *et al*., 2001).

Several studies have investigated the physical and chemical properties of ash from diverse plant species (Etiegni and Campbell, 1991; Khanna *et al*., 1994; Ùbeda *et al*., 2009; Gabet and Bookter, 2011). However, the forms of plant nutrients in ash including the forms of water-soluble elements and their interactions with soils has received little attention. The research described in this paper determined the amounts and mineral forms of elements in ash derived from several Australian native plant species and evaluated the reactions of plant ash with heated and unheated soil.

### Materials and Methods

#### Ash preparation

The Australian native plants used for the study are major constituents of forest vegetation at Bakers Hills, Darling Range, Western Australia: silver wattle (*Acacia retinodes*), prickly Moses (*Acacia pulchella*), wandoo/white gum (*Eucalyptus wandoo*), red gum or marri (*Corymbia calophylla*), grass tree (*Xanthorrhoea preissii*), jarrah (*Eucalyptus marginata*) and harsh hakea (*Hakea prostrata*). The leaves and wood samples were dried in an oven at 60$^{o}$C until the weight was constant (up to 7 days) and then cut to 1-2 cm to make the sample homogenous. A subsample of approximately 50 g was then placed in an aluminum oven tray and burned in open air for about 15 min with agitation to maximize burning, to simulate combustion in a forest fire in the field. Temperature was not recorded during the combustion process. Combustion was almost complete for each material with only small amounts of charcoal remaining in the ash. A key to the materials investigated and the abbreviations used to identify these materials are given in Table 1.

**Table 1. The nomenclature for plant ash and some properties of the ash. SSA= specific surface area , EC= electrical conductivity , Bic P= bicarbonate soluble P.**

| Key | Species and plant material | Ash % | Ash Color (Munsell) | SSA ($m^2/g$) | pH (1:5) | EC (1:5) (dS/m) | Bic P (mg/kg) |
|---|---|---|---|---|---|---|---|
| SW | Silver wattle wood | 1.72 | G1 8/N | 5.8 | 13.8 | 18.9 | 145 |
| SL | Silver wattle leaf | 3.39 | G1 5/N | 4.2 | 12.8 | 13.9 | 198 |
| PM | Prickly Moses leaf and twig | 2.81 | G1 4/N | 8.3 | 13.1 | 8.5 | 199 |
| WW | Wandoo wood | 3.07 | 2.5Y 7/2 | 9.2 | 13.6 | 15.1 | 176 |
| WL | Wandoo leaf | 3.02 | G1 5/N | 2.4 | 13.3 | 14.3 | 32 |
| RW | Red gum wood | 4.71 | G1 7/N | 2.4 | 13.8 | 23.2 | 183 |
| RL | Red gum leaf | 4.07 | G1 6/N | 2.5 | 13.4 | 18.4 | 30 |
| GT | Grass tree leaf | 3.40 | G1 3/N | 2.7 | 13.5 | 22.9 | 145 |
| JL | Jarrah leaf | 3.27 | 2.5Y 6/2 | 2.9 | 13.2 | 18.9 | 196 |
| HH | Harsh hakea leaf and twig | 4.01 | G1 4/N | 7.7 | 12.3 | 16.3 | 164 |

*Characterization of the ash*

Ash color was classified using a Munsell Color Chart (Munsell Color Co., 1998). Specific surface area (SSA) was measured using a Micrometrics Gemini 2375 instrument with VacPrep 061 using a five point BET method with $N_2$ as the absorbate. The pH of the ash was determined in a 1:5 deionized water extract. The samples were analyzed for bicarbonate extractable P (Bic P) (Colwell, 1963). Total carbon and nitrogen were determined on an Elementar CNS (Vario Macro) analyzer. Water-soluble elements in ash were determined using Association of Official Analytical Chemistry (AOAC) standard methods (AOAC, 1975). 0.5 g of ash was mixed with 100 ml of deionised water and placed on a mechanical shaker for 24 h. Extracts were then filtered through a 0.22 μm Millipore filter. Water-soluble elements were then measured using an inductively coupled plasma optical emission spectrometer (ICP-OES) (Perkin-Elmer, Norwalk, CT, USA). Ash content of plant material was determined by a separate dry combustion in a muffle furnace at 550$^o$C for 2 hours **(**Rayment and Higginson, 1992).

Total elements were determined in duplicate by ICP-OES after concentrated perchloric acid digestion of the ash (Rayment and Higginson, 1992). Conventional XRD of the ash was conducted on a Philips PW3020 diffractometer with a diffracted beam monochromator (CuKα, 50kV, 20mA). Powder samples were scanned from 4 to 70$^o$ 2θ, using a step size of 0.02$^o$ 2θ and a scan speed of 0.04$^o$ 2θ $sec^{-1}$. Synchrotron XRD (SXRD) analysis was performed at the Australian Synchrotron where powder samples were mounted into glass capillaries with a 1.0Å wavelength set for this analysis in order to provide a high peak/background and adequate resolution for identifying minor constituents.

The composition and morphology of the plant ash was examined by scanning electron microscopy (SEM) and energy dispersive X-ray spectrometry (EDS) using a JEOL 6400 instrument. Samples for SEM analysis were placed on metal stubs and carbon coated before analysis.

## Results and Discussion

*Characteristics of the ashes*

The ash content of the plant materials as the result of furnace burning ranged from 1.7 to 4.7%. Properties of plant ash produced by open-air burning are shown in Table 1. Ash was white (G1 8/N, G1 7/N, G1 6/N, G1 5/N, or G1 4/N) and grey (G1 3/N, 2.5Y 7/2 or 2.5Y 6/2). Grey ash contains more carbon (charred organic material) than white ash possibly reflecting differences in the intensity of combustion (Khanna *et al*., 1994; Neary *et al*., 1999; Lentile *et al*., 2006; Roy *et al*., 2010).

All the ashes were alkaline with pH ranging from 12.3 to 13.8. These high values are associated with the presence of carbonates, oxides and hydroxides of base cations in ash (Ulery *et al*., 1993). The electrical conductivity of the ash extracts (1:5) was high, indicating that the ash contains considerable amounts of soluble salts, with red gum wood ash containing the highest amount (23.2 dS/m) and prickly Moses leaf/twig ash the least (8.5 dS/m).

*Mineralogy and Morphology of ash (XRD and SEM)*

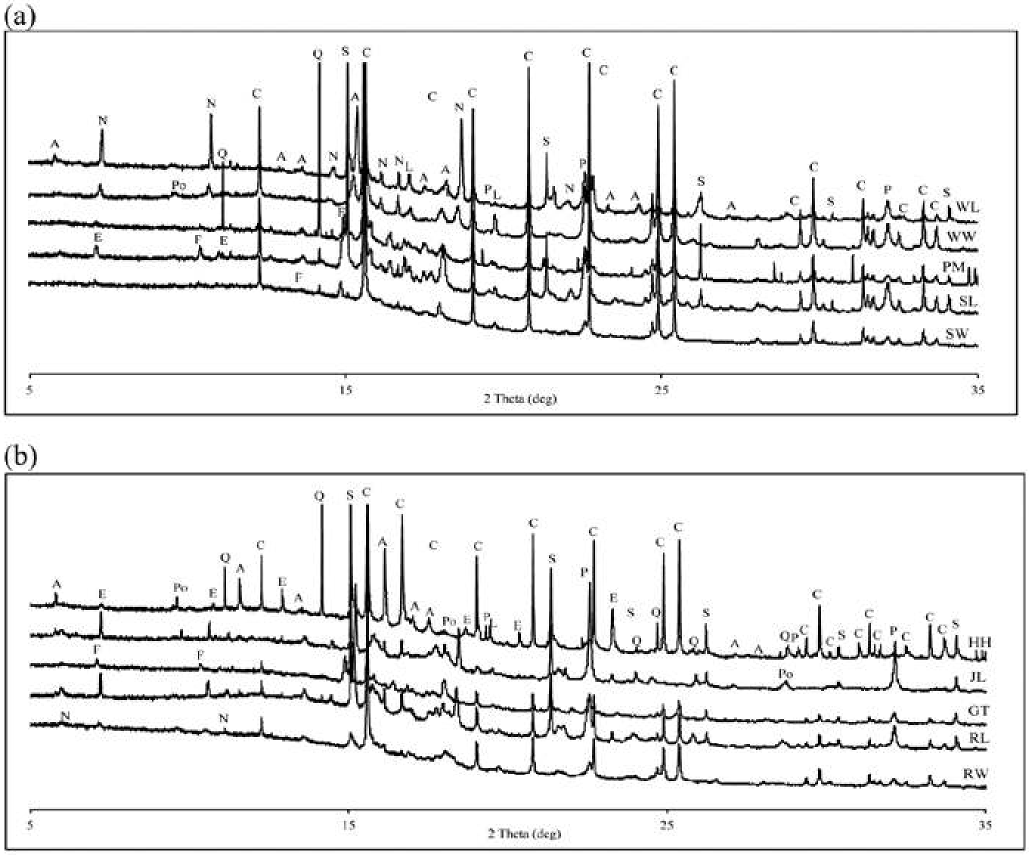


**Figure 1: (a) Synchrotron XRD patterns for RW, RL, GT, JL, and HH. (b) Synchrotron XRD patterns for RW, RL, GT, JL, and HH. (F=fairchildite (K2Ca(CO3)2), N=nesquehonite (MgCO3.H2O), C=calcite (CaCO3), S=sylvite (KCl), A=apatite (Ca5(PO4)3(OH)), L=lime (CaO), E=scolecite (CaAl2Si3O10.3(H2O)), Q=quartz (SiO2), Po=portlandite (Ca(OH)2) and P=periclase (MgO).**

The main compounds identified by SXRD (Figure 1.) in the plant ashes were various oxides, carbonates and hydroxides of Ca, Mg and K as has also been reported by Misra *et al*. (1993). The minerals present were fairchildite ($K_2Ca(CO_3)_2$), nesquehonite ($MgCO_3.H_2O$), calcite ($CaCO_3$), sylvite (KCl), lime (CaO), scolecite ($CaAl_2Si_3O_{10}.3(H_2O)$,), portlandite ($Ca(OH)_2$, periclase (MgO), apatite group minerals probably resembling hydroxyl-apatite ($Ca_5(PO_4)_3(OH)$) and wilkeite ($Ca_5((P, S, Si)O_4)_3(OH,CO_3)$) and quartz ($SiO_2$). Fairchildite, nesquehonite, scolecite, portlandite and periclase were not identified using conventional XRD. The proportion of water-soluble elements in ash was low, which may be attributed to the presence of these mostly poorly soluble minerals. Hydroxide minerals form when the ash is exposed to air as oxides of Ca, Mg and K react with atmospheric water (Misra *et al*. 1993). Reaction of these oxides with atmospheric carbon dioxide results in the formation of calcite and other carbonates.

Scanning electron microscopy of particles and associated EDS X-ray spectra for all the ashes (data not shown here) show that particles of diverse size, shape and composition occur. Ash particles vary in size with diameter ranges from <1 µm to 100 µm. Apart from distinct mostly prismatic, large calcite crystals, most grains seen in the micrographs are K- and Mg-rich aggregates of very small particles often containing Cl and they commonly contain little P, Fe, S and Si. The sensitivity of EDS for Na is extremely poor, so that the Na peak is not visible in EDS spectra. Liodakis *et al*. (2009) found that SEM spectra of particles of plant ash contained lines due to Ca, Mg, K, P and Si as observed in this work. The present results indicate the complex nature of ash, with a high diversity of composition, shape and size of particles, which are present in complex aggregates.

## Conclusions

The ash of these native plant species has diverse chemical, mineralogical and morphological properties and differs in solubility, depending on plant species and plant part. Compounds in the ash include fairchildite ($K_2Ca(CO_3)_2$), nesquehonite ($MgCO_3.H_2O$), calcite ($CaCO_3$), sylvite (KCl), lime (CaO), scolecite ($CaAl_2Si_3O_{10}.3(H_2O)$), quartz ($SiO_2$), portlandite ($Ca(OH)_2$), periclase (MgO). Much of the P in ashes is present as the mineral apatite which is more soluble when applied to acid soil but may be much less available to plants if the ash is deposited on naturally alkaline soil or on a soil that has been raised to a high pH value due to the liming action of ash.

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# Soil factors and foliar nutrient status associated with variation in upper mid-crown yellowing of radiata pine

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**Abstract**

Upper mid-crown yellowing (UMCY) is a visible magnesium (Mg) deficiency symptom widespread in semi-mature radiata pine stands throughout New Zealand, particularly on pumice soils in the central North Island. The soil factors and *foliar nutritional status* associated with variation in UMCY symptoms in radiata pine are not well understood. This study provides an update on UMCY research based on a nationwide clonal trial series covering a range of soils and climates. From 5 trials assessed in this study, we found UMCY symptoms caused needle loss and reduced tree growth by approximately 10% compared to trees without visual symptoms. UMCY symptom scores varied significantly across sites and were related to site altitude and especially soil Mg ($r = -0.69$), Olsen P ($r = -0.65$), K ($r = -0.53$) and K/Mg ratio ($r = 0.62$). Large variation in UMCY scores also existed among the clones tested, which was associated with foliar K/Mg ratio and Ca, Mg and other nutrient concentrations. Site-to-site correlations were strong for UMCY scores of the tested clones, indicating clonal stability in expression of the symptom across the sites. Some sensitive clones (e.g. S02C1, P07C10) have potential uses as indicator clones for Mg deficiency. This study has two important implications for management of operational clonal plantations in New Zealand: (1) UMCY tolerant clones should be deployed to sites with low soil Mg and high soil K/Mg, and (2) integrated management of site and clone resources is required for minimising the UMCY problem in radiata pine stands.

**Key Words**

UMCY, radiata pine clones, soil factors, *foliar nutritional status*

## Introduction

Magnesium (Mg) deficiency in radiata pine results in needle-tip yellowing and, if severe, crown dieback, growth loss (Beets *et al.* 1994) and a reduction in wood basic density (Beets, unpublished). In semi-mature stands it is commonly referred to as upper mid-crown yellowing (UMCY) (Beets *et al.* 1993; 2004). Large differences in the severity of UMCY symptoms occurred over small spatial scales at Puruki forest ($38^0 26'$S, $176^0 13'$E) in the first rotation, partly because of genetic differences in nutritional traits between trees. Suspected variations in site factors have eluded detection (Beets *et al.* 2004). Several trials established in a second-rotation radiata pine stand at Puruki forest have confirmed that soil and genetic factors are associated with variation in Mg deficiency symptoms observed on the pumice soils (Beets *et al.* 2004).

Similar information is lacking on site and genetic variation in UMCY symptoms across a range of sites with different soils and climates. To better understand the *occurrence* of UMCY *symptoms* and to develop cost-effective strategies for managing soils and genotypes to overcome this problem on a national basis, a nationwide clonal trial series covering a range of soils and climates was established to test the generality of UMCY results obtained from Puruki forest. This clonal trial series will provide good opportunities to assess growth, health, and wood quality characteristics in relation to soils and genotypes. The purpose of this study was to quantify the effect of site and clone on occurrence of UMCY symptoms and determine the soil and foliar nutritional factors associated with site and clonal variation in UMCY symptoms.

## Materials and Methods

### *Trial design and genetic material*

A nationwide clonal trial series was established in 2002-05 at 14 sites covering a range of soils and climates (Figure 1). For each of 14 trials, there were 4 plots. Three ramets of each of 40 radiata pine clones were randomly planted in each plot. Of the 40 clones, 20 clones were created through fascicle cuttings from the control-pollinated families selected for high volume growth rate and improved stem form (GF24-31) but unknown nutritional characteristics. The remaining 20 clones were created through fascicle cuttings from the seedlings of open-pollinated families (GF7) selected for different nutrition-related UMCY symptom scores and diameter growth.

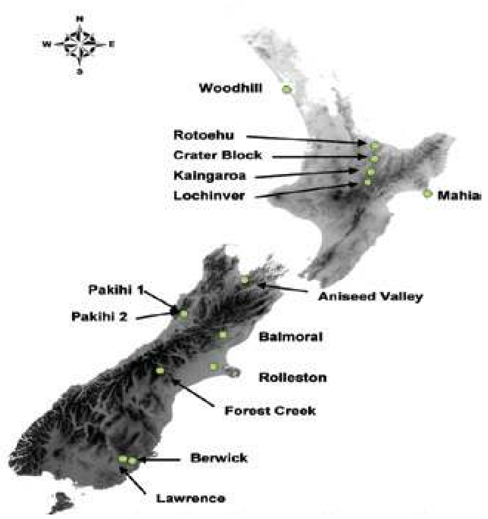


**Figure 1. Locations of the 14 trials.**

*UMCY symptom scoring and growth measurement*

Five of the 14 sites (i.e. Lochinver, Kaingaroa, Crater Block, Berwick and Lawrence), more *susceptible* to Mg deficiency, were selected for assessment of UMCY symptoms in spring of 2010 and 2011 (at age 5-8 years). UMCY symptom was scored for individual trees in all four plots at 5 sites using a scale of 1-5 (as below) developed for young trees (Beets *et al.* 2004):

1) Year 1 & 2 needles entirely green and healthy.

2) 2$^{nd}$ year needles are yellow tipped and current needles are green and healthy.

3) 1 year old needles are yellow tipped over part of the shoot and current are green.

4) 1 year old needles are yellow tipped over entire length of the shoot and current are green.

5) 1 year old needles are yellow tipped over entire length of the shoot and current have yellow needle tips.

Tree height and diameter at breast height (DBH) were also measured for individual trees at each site in winter of 2010 and 2011.

*Soil and foliar chemistry*

In the first year after the trial was established at each site, 35 soil cores per plot were collected from all 4 plots at the depth of 0–0.1 m. The soil cores were bulked by plot and site. Soil samples were air-dried and ground to pass a 2 mm sieve and then analysed for soil pH, total C, N and P, Olsen-P, *exchangeable Ca, Mg, K and Na, and CEC. Foliage samples were only collected from the trial at* Kaingaroa at the age of 5 years. The current-year needles were sampled from the youngest second-order branches of each of the 3 ramets of 40 clones within each plot. The foliage samples were bulked by clone (one bulked sample per clone per plot), with 160 samples in total. The foliage samples were oven-dried at 65ºC and ground for chemical analysis of C, N, P, K, Ca, Mg, B, Cu, Zn, Fe and Mn using ICP-OES after $HNO_3/H_2O_2$ digestion.

*Statistical analysis*

We transformed the categorical UMCY scores according to Beets *et al* (2004) before analysis. SAS mixed model of two-way ANOVA was used to test site and clonal effects and their interactions on UMCY scores and growth data. *Pearson correlation analysis was conducted to determine the relationships between UMCY scores and site, soil or foliar variables, and between sites in UMCY scores.*

## Results

*Site and clonal differences in UMCY symptoms*

The severity of UMCY symptoms varied significantly among the sites, with the highest mean score at Lochinver and the lowest mean score at Crater block (Table 1).

**Table 1: Site, age, mean DBH, tree height and UMCY score. Standard deviation is provided in parentheses. Sites with the same letter in the final column do not differ in mean UMCY score**

| Site | Age[1] | DBH (cm) | Height (m) | UMCY score | |
|---|---|---|---|---|---|
| Crater Block | 6 (5) | 120 | 6.5 | 1.46 (1.07) | D |
| Berwick | 8 (8) | 143 | 7.3 | 1.74 (1.03) | C |
| Lawrence | 8 (8) | 167 | 7.9 | 2.12 (1.22) | B |
| Kaingaroa | 5 (6) | 27.9 | 2.3 | 2.35 (1.43) | B |
| Lochinver | 7 (8) | 28.5 | 2.3 | 2.82 (1.49) | A |

[1] Age – when UMCY symptoms were scored, and when tree growth was measured (age in years in parentheses).

Large variations in UMCY symptom scores were also found among the clones tested in this study (Figure 2). Six clones had high scores but most were clustered with moderate to low scores.

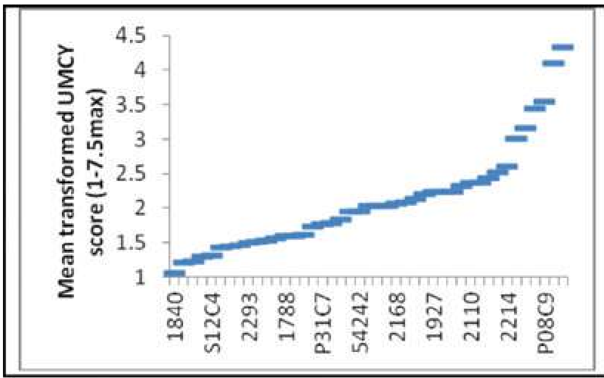


**Figure 2. Distribution of UMCY symptoms for a range of clones (mean values across 5 sites).**

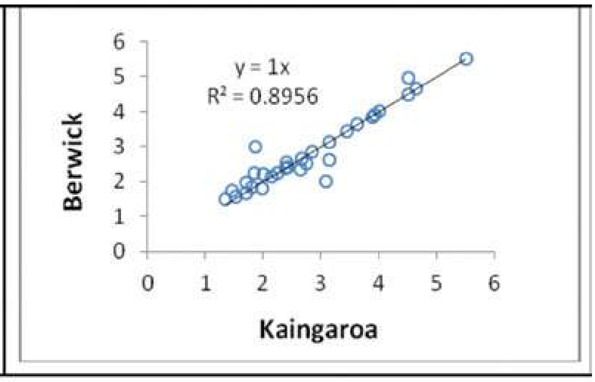


**Figure 3. Correlation between Berwick and Kaingaroa for UMCY scores of 32 paired clones.**

Clones S02C1 and P07C10 consistently showed severe UMCY symptoms across sites. This implies they had the potential of being used as indicator clones for Mg deficiency. Other clones susceptible to UMCY symptoms were P08C9, 2084 and S07C8. Clone 1840 had the extremely low UMCY symptom score, indicating tolerant to Mg deficiency. Other clones with low UMCY symptoms were 2469 2583, 1678, S12C4, 2526, 1608, P02C7, 2293, P26C5.

*Site-site correlations for UMCY symptoms*

Overall, there were good correlations among the sites for UMCY symptom scores (data not shown). A very strong correlation was found between Berwick, a South Island site, and Kaingaroa, a North Island site for clonal UMCY scores (Figure 3). This provides support for the robustness of the scoring method and also suggests a strong clonal effect on expression of UMCY symptoms.

**Table 2. Correlations and *p*-values (testing the significance of the correlation) for the UMCY scores of 3-year-old trees at Puruki paired with 5- to 8-year-old trees at each of 5 trials in this study**

| Site | Puruki |
|---|---|
| Crater Block | $r = 0.57$ ($P = 0.01$) |
| Lawrence | $r = 0.51$ ($P = 0.003$) |
| Berwick | $r = 0.47$ ($P = 0.007$) |
| Lochinver | $r = 0.43$ ($P = 0.015$) |
| Kaingaroa | $r = 0.29$ ($P = 0.11$) |

The 20 clones in this study were originally created from materials tested at the Puruki trials. We found good correlations between Puruki and each of 5 sites except Kaingaroa for UMCY scores of these clones (Table 2). This provides further evidence of strong clonal stability in expression of UMCY symptoms across sites.

*UMCY symptoms in relation to soil and foliar chemistry and other site factors*

Soil samples on a plot basis were collected one year after establishment for soil chemistry analysis.
The nutrient concentrations and ratios indicate that none of the sites had an optimal nutrient balance for

all critical elements (data not shown). We found across sites there were good correlations between UMCY symptom scores and *several soil chemical properties (Table 3)*. It has been reported that Mg deficiency in radiata pine is likely where soil exchangeable K:Mg ratios are in excess of 1.5, foliar Mg concentrations are less than 0.12%, and foliar K:Mg concentration ratios are greater than 10 (Olykan *et al*. 2001; Beets *et al.* 2004). However, the similar yellowing of radiata pine needles in an Australia soil is associated with K deficiency (Smethurst *et al*. 2007).

**Table 3. Relationships between the plot-based UMCY symptom scores and soil chemical properties**

| Soil chemical property | Correlation coefficient |
|---|---|
| Exchangeable Mg ($cmol_c\ kg^{-1}$) | $r$ = - 0.69 (n=20), $p < 0.001$ |
| Olsen P ($mg\ kg^{-1}$) | $r$ = - 0.65 (n=20), $p < 0.01$ |
| Exchangeable K ($cmol_c\ kg^{-1}$) | $r$ = - 0.53 (n=20), $p < 0.01$ |
| Exchangeable K/Mg ratio | $r$ = 0.62 (n=20), $p < 0.01$ |
| N/P ratio | $r$ = 0.48 (n=20), $p < 0.05$ |
| Exchangeable Ca/Mg ratio | $r$ = 0.44 (n=20), $p < 0.05$ |

Foliar nutrient data was only available from the Kaingaroa trial, where UMCY scores were assessed subsequently during the same year (i.e. 2009). The Kaingaroa data, using a needle–weight-adjusted comparison, showed smaller needles were associated with higher UMCY scores. Similarly, we found that across sites that trees with UMCY symptoms had reduced height and diameter by approximately 10% when compared to trees without symptoms.

UMCY symptoms were found to be negatively correlated with foliar concentrations of Ca, Zn, Mg and K in decreasing order. A combination of foliar K/Mg ratio and Na explained 80% of the variation in UMCY scores at this site. Data from more sites are required to test the applicability of this finding to other sites.

Although a limited number of sites were assessed in this study, we found that 79% of the site variation in UMCY scores could be explained by the altitude of sites ($P$= 0.06). A model incorporating trial age, stocking rate and altitude explained 86% of the site variation in UMCY scores ($P$ = 0.03).

## Conclusions

There were significant site differences in UMCY symptoms, which were associated with site altitude and soil chemistry, especially *exchangeable* Mg, Olsen P, *exchangeable* K and K/Mg ratio. A wide range of clonal variations were observed for UMCY symptoms, which were more related to clonal foliar K/Mg ratios. Our findings on UMCY symptoms were consistent with previous work reported from Puruki trials. This, together with a relatively small site by clone interaction, indicated clonal stability in expression of UMCY symptoms across sites. Some sensitive clones (e.g. S02C1, P07C10) could be used as indicator clones for Mg deficiency. Severe UMCY symptoms caused needle loss and reduced tree growth. This study implies appropriate clones need to be deployed to specific sites by forest owners and managers to manage the UMCY problem in radiata pine stands.


## Acknowledgements

We would like to thank the forest companies for their on-going in-kind support, NZ Ministry of Science and Innovation for funding, many Scion staff for collecting the data for this study, and Paul Charteris at Scion for valuable comments on this manuscript.


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# Soil amelioration by *Acacia* hybrid: An assessment of soil condition for re-establishing native species in the tropics

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### Abstract

Tropical acacias are used for reforestation and recovery of degraded lands. This study aimed to evaluate the potential of *Acacia* hybrid (*A. mangium* x *A. auriculiformis*) to improve the physical and chemical properties of degraded soils. The experiment was carried out in second- or later-rotation *Acacia* hybrid plantations in Central Vietnam. A total of 109 soil samples was collected from the 0 – 20 cm topsoil of 30 plantations representative of five ages (0.5, 1.5, 2.5, 3.5 and 4.5-5.5 year-old) in six locations, and in nearby fallow land at each location. *Acacia* plantations significantly enhanced ($P<0.05$) total organic carbon (TOC), total nitrogen (TN), exchangeable calcium (Ex-Ca), magnesium (Ex-Mg), sodium (Ex-Na), electronic conductivity (EC) and bulk density (BD) when compared to fallow land. However, *Acacia* plantations increased soil acidity. Within the 5.5-year-old rotations examined, most soil properties were not significantly changed with increasing plantation age ($P>0.05$). However, the trends showed many nutrient properties declined during the first 2 or 3 years after establishment. After 4 years, TOC recovered to initial levels, though base cations remained lower. Soil properties were strongly related to initial soil and site factors such as clay content, gravel volume, slope angle and elevation.

**Keywords:**
*Acacia* hybrid, soil amelioration, site management, nurse crop

### Introduction

Tropical *Acacia* species have been planted widely in many countries in reforestation programs (Turnbull *et al.* 1997). Their tolerance of difficult site conditions and ability to produce a marketable product in a less than 10-year rotation (Turnbull *et al.* 1997), make them ideally suited to this purpose. This fast growth and being leguminous species are assumed to be associated with recovery of soil nutrient capital and an acceleration of nutrient cycling (Brockwell *et al.* 2005). In Vietnam, a natural hybrid of *A. mangium* × *A. auriculiformis* has been the dominant species for commercial planting since the late 1990s (Kha 2001). This study determines how soil nutrients and some other soil properties change following establishment of *Acacia* hybrid plantations on degraded lands in Vietnam.

The research was undertaken in lowland areas of Thua-Thien-Hue province in Central Vietnam, ranging from 18 to 80 m altitude and 3° to 20° slope angle (Fig 1). Silvicultural practices were relatively homogeneous across the sampled plantations.

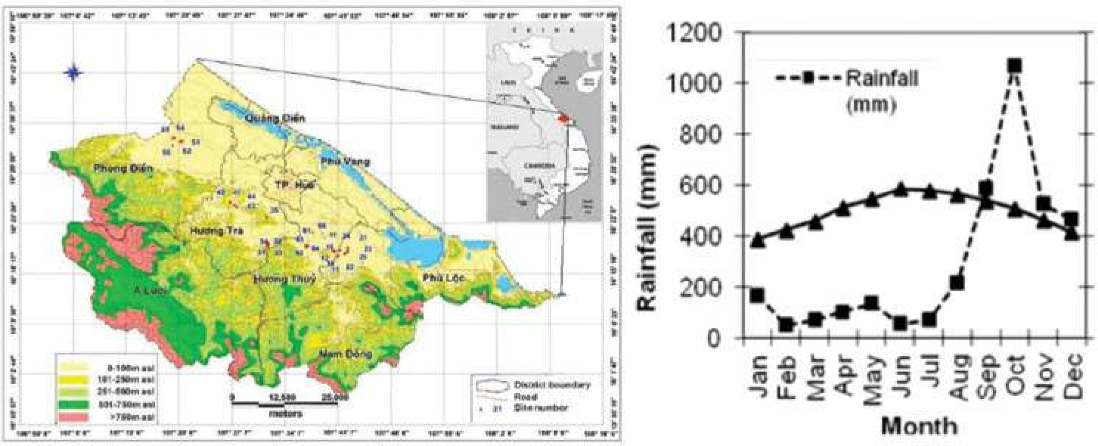


**Figure1: Map of the sampled plantations, mean monthly rainfall and air temperature from 2005-2009 at Hue City (Thua Thien Hue Statictical Office 2010).**

## Material and methods

A total of 109 soil samples was collected from 30 plantations representative of five age classes (0.5, 1.5, 2.5, 3.5 and 4.5-5.5 year-old) in six locations (Fig. 1). Soil was also collected in adjacent fallow land at each location as an unafforested control. Three or four replicated samples from each site were composites of five soil cores randomly collected from the topsoil layer of 0 - 20 cm.

Soil sample preparation and analysis followed the Australian Laboratory Handbook of Soil and Water Chemical Methods (Rayment and Higginson 1992). Bulk density (BD) was determined after samples had been dried at 105$^{o}$C for over 24 h and weighed after cooling in a desiccator. Soil $pH_{H2O}$ and electronic conductivity (EC) were measured in a 1:5 mixture of soil and deionised water. Soil $pH_{CaCl2}$ was measured in a 1:5 mixture of soil and 0.01M calcium chloride. Total organic carbon (TOC) and total nitrogen (TN) were determined using a CHNS/O Element Analyser (PerkinElmer). Extractable phosphorus (Ext-P) was determined using the Olsen manual colour method. Soil exchangeable cations (Ex-K, Ex-Ca, Ex-Mg and Ex-Na) were extracted with 0.01M silver-thiourea $(AgTU)^{+}$. Particle size analysis was determined by a hydrometer. All data are reported as unit per oven dried weight.

The differences between soil properties of the plantations and fallow land were examined using the t-test procedure in SAS 9.2. The dependencies of soil properties on age, clay, gravel volume, slope and elevation were examined using the PROC MIXED procedure in SAS 9.2. The mixed model included a random effect corresponding to site and orthogonal polynomials for age. The analyses were weighted to allow for non-homogenous variances by site. Stepwise selection based on *P* value ($P$<0.05) was used to select variables other than age to identify the best fitting model.

## Results and discussion

TOC and TN were significantly higher in plantations than fallow land (Table 1), presumably due to the rapid production and decomposition of litter and N-fixation by acacias. Similarly, Ex-Ca, Ex-Mg and Ex-Na were significantly higher in the plantations; this may be attributed to the potential for substantial loss of cations from these sandy soils in the fallow lands during heavy rainfall. Conversely, $pH_{CaCl2}$ and $pH_{H2O}$ were significantly lower in plantations than fallow land. It has been reported previously that N-fixation leads to soil acidity due to nitrate leaching (Binkley and Giardina 1997). EC was much higher in plantations than fallow land, perhaps related to higher levels of exchangeable cations in the plantations. BD showed that soil compaction was lower in plantations. Vegetation cover plays an important role in mitigating soil compaction as it facilitates enhancement of soil organic matter (SOM), soil organism activity and root penetration. Ext-

**Table 1: Means and standard deviations of soil properties in 0 – 20 cm topsoil of second (or later) rotation *Acacia* hybrid plantations and nearby fallow lands in Thua Thien Hue, Vietnam.**

| Properties | Fallow lands | Age | | | | |
|---|---|---|---|---|---|---|
| | | 0.5 | 1.5 | 2.5 | 3.5 | 4.5-5.5 |
| TOC (%) | 0.84±0.17 | 1.22*±0.37 | 1.16*±0.38 | 1.15*±0.49 | 1.10*±0.32 | 1.33*±0.48 |
| TN (%) | 0.07±0.02 | 0.09*±0.02 | 0.08±0.02 | 0.08±0.03 | 0.08±0.02 | 0.09*±0.03 |
| C/N ratio | 12.3±1.9 | 13.1±1.7 | 13.7±1.8 | 14.5*±2.5 | 14.0±2.1 | 14.8*±1.3 |
| Ext-P (mg $kg^{-1}$) | 2.48±0.99 | 2.55±1.25 | 1.81±0.91 | 2.03±1.25 | 2.12±1.03 | 2.03±1.25 |
| Ex-K (cmol $kg^{-1}$) | 0.025 ±0.014 | 0.022±0.012 | 0.019±0.009 | 0.018±0.008 | 0.017±0.008 | 0.018±0.006 |
| Ex-Ca (cmol $kg^{-1}$) | 0.14±0.05 | 0.73*±0.41 | 0.64*±0.29 | 0.60*±0.42 | 0.54*±0.28 | 0.58*±0.30 |
| Ex-Mg (cmol $kg^{-1}$) | 0.10±0.03 | 0.21*±0.09 | 0.18*±0.06 | 0.19*±0.06 | 0.19*±0.06 | 0.19*±0.07 |
| Ex-Na (cmol $kg^{-1}$) | 0.02±0.02 | 0.13*±0.03 | 0.13*±0.06 | 0.15*±0.06 | 0.13*±0.07 | 0.13*±0.06 |
| pHCaCl2 | 3.98±0.10 | 3.84*±0.16 | 3.86±0.12 | 3.87±0.12 | 3.85*±0.11 | 3.78*±0.07 |
| pHH2O | 4.52±0.13 | 4.35*±0.20 | 4.40±0.15 | 4.43±0.19 | 4.40±0.16 | 4.30*±0.14 |
| EC | 32.7±5.5 | 69.4*±23.6 | 60.5*±17.4 | 58.5*±15.1 | 60.5*±18.1 | 69.4*±18.2 |
| BD (g $cm^{-3}$) | 1.55±0.09 | 1.37*±0.10 | 1.42*±0.10 | 1.37*±0.06 | 1.38*±0.13 | 1.36*±0.09 |
| Clay (%) | 19.1±4.9 | 17.8±3.9 | 19.7±3.9 | 17.0±5.8 | 18.8±6.1 | 19.2±5.3 |
| Silt (%) | 9.9±5.7 | 8.6±5.3 | 8.5±3.5 | 8.3±6.3 | 10.0±7.4 | 10.5±6.2 |
| Sand (%) | 71.0±9.7 | 74.1±8.1 | 71.9±5.9 | 74.9±11.1 | 71.5±12.3 | 70.2±9.1 |

Within each row, means of age classes with asterisk (*) are significantly different ($P$<0.05) from means of fallow lands. TOC – total organic carbon, TN – total nitrogen, Ext-P – extractable phosphorous, Ex-K – exchangeable potassium, Ex-Ca – exchangeable calcium, Ex-Mg – exchangeable magnesium, Ex-Na – exchangeable sodium, EC – electrical conductivity, BD – bulk density

P and Ex-K in plantations were not significantly different with those in fallow lands. Acacias require P for symbiotic N-fixation as high P concentration is required in nodules (Sun *et al.* 1992); K is very easily leached on sandy soil (Kasongo *et al.* 2009).

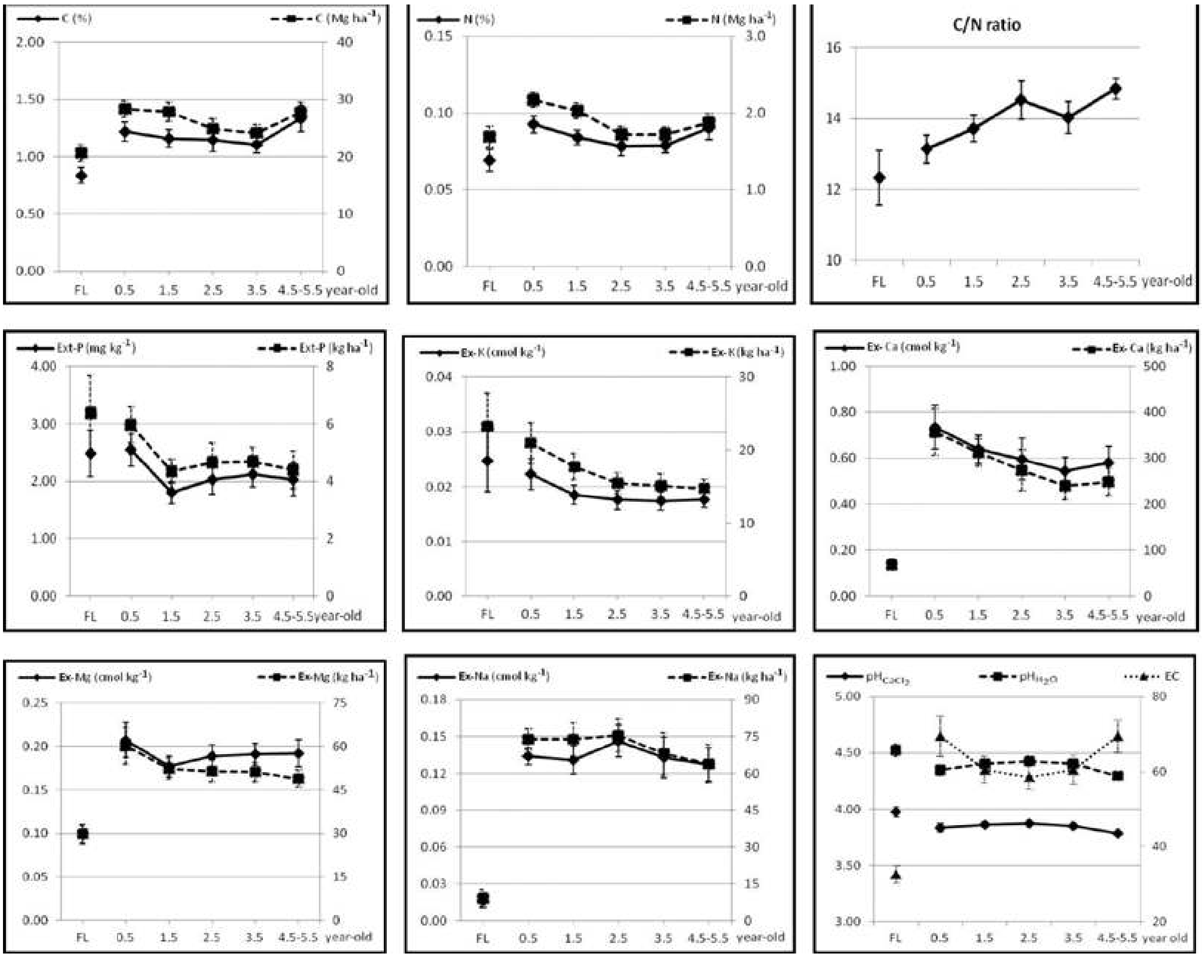


**Figure 2: Means and standard errors of soil properties in 0 – 20 cm topsoil of second (or later) rotation Acacia hybrid plantations and nearby fallow lands (FL) in Thua Thien Hue, Vietnam. Nutrient stock per hectare was the product of nutrient value in soil mass with soil mass volume after subtracting the gravel volume.**

Within a five year rotation, only carbon:nitrogen ratio (C/N) and EC changed significantly with time in the plantations (Table 2). Other variables did not change significantly, but levels of TOC and TN followed a trend which showed depletion in the first 2 - 3 years and then recovery by age 4 - 5 years (Fig. 2). The base cations, except for Ex-Na also declined but recovered slightly by age 4-5 years. C/N increased, indicating higher litter deposition but lower decomposition rates with time.

The conventional practice in land preparation for tree planting in Vietnam is to burn all residues and debris from the previous rotation. This inevitably leads to a loss of nutrient stock. The depletion of TOC after planting is in accordance with other research which showed that TOC levels decreased until age five years before recovering to pre-planting levels (Paul *et al.* 2002). Disturbance caused by land preparation and susceptibility of bare land to erosion are the major reasons for this decline.

Although clay content of soils in this study was low, it still significantly affected TOC, TN, C/N, Ext-P, Ex-K, Ex-Ca, Ex-Mg and EC (Table 2). Variations in gravel (>2 mm) volume across the study sites correlated with significant changes in TOC, TN, C/N, Ex-Ca, $pH_{CaCl2}$ and EC. In this high rainfall area, slope is presumed to affect soil nutrient contents because of soil erosion, but this might be controlled by other factors e.g. ground vegetation, so only C/N, Ex-K, Ex-Mg and EC were affected by slope. Site elevation of 18 - 80 m was not expected to have much effect on soil properties, but results indicated C/N, Ex-K, Ex-Ca, Ex-MG, $pH_{CaCl2}$, $pH_{H2O}$ and EC were related to altitude. Altitude may not be the core factor for these changes, but it may relate to primary factors e.g. soil type or parent materials which may be varying with elevation.

**Table 2: Final models of soil properties of Acacia hybrid plantations in Thua Thien Hue, Vietnam.**

| Independent variable | Effect | Dependent variable TOC | TN | C/N | Ext-P | Ex-K | Ex-Ca | Ex-Mg | Ex-Na | $pHCaCl_2$ | $pHH_2O$ | EC |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Age | Age | 0.1194 | -0.0085 | 3.7212 | -1.8103 | -0.0112 | -0.0531 | -0.0706 | -0.0042 | 0.0108 | 0.0511 | 23.2963 |
| | P | ns | ns | 0.0024 | ns | ns | ns | ns | ns | ns | ns | <.0001 |
| | Age2 | -0.1076 | -0.0017 | -1.4362 | 0.7922 | 0.0038 | -0.0274 | 0.0312 | -0.0002 | 0.0095 | 0.0051 | 7.1806 |
| | P | ns | ns | ns | ns | ns | ns | ns | ns | ns | ns | <.0001 |
| | Age3 | 0.0192 | 0.0007 | 0.1700 | -0.0990 | -0.0004 | 0.0073 | -0.0037 | 0.0002 | -0.0029 | -0.0038 | -0.5481 |
| | P | ns | ns | ns | ns | ns | ns | ns | ns | ns | ns | 0.0077 |
| Clay (%) | Estimate | 0.0029 | 0.0005 | -0.0061 | -0.0359 | 0.00002 | 0.0079 | 0.0005 | | | | 0.0991 |
| | P | 0.0338 | 0.0157 | 0.0014 | 0.0361 | 0.0092 | 0.0097 | 0.0418 | | | | <.0001 |
| Gravel volume (%) | Estimate | 0.0063 | 0.0004 | 0.0045 | | | -0.0012 | | | -0.0008 | | -0.0114 |
| | P | 0.0004 | 0.0032 | 0.0005 | | | 0.0224 | | | 0.0019 | | <.0001 |
| Slope (o) | Estimate | | | -0.0381 | | 0.00006 | | 0.0021 | | | | 0.7385 |
| | P | | | 0.0004 | | 0.0230 | | 0.0079 | | | | <.0001 |
| Altitude (m) | Estimate | | | -0.0467 | | 0.0003 | 0.0053 | 0.0015 | | 0.0025 | 0.0028 | 0.5698 |
| | P | | | <.0001 | | <.0001 | 0.0188 | 0.0004 | | 0.0070 | 0.0323 | <.0001 |

*ns* – not significant ($P$>0.05)

## Conclusion

*Acacia* hybrid plantations have a positive effect on TOC, TN, exchangeable cations, EC and BD, while not on Ext-P and Ex-K. In short rotations not exceeding 5.5 years, soil acidity increased. However the contents of most soil nutrients remained unchanged, though many declined during the first 2 to 3 years after planting. TOC then recovered; base cations still remained lower. Some soil properties were strongly related to initial soil and site factors such as clay, gravel volume, slope and elevation. *Acacia* hybrid significantly improves degraded soils, but to maintain nutrient stocks, appropriate site management will also be necessary.

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# Worms and waste – uncovering a potential partnership

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As a product of human excreta, biosolids can be a vehicle for numerous contaminants (heavy metals, pharmaceutical products and human pathogens). While these characteristics present challenges, the waste material also offers opportunities for re-use as they contain high concentrations of valuable nutrients. Uncertainty is a key barrier to land application and both social and cultural acceptability are fundamental to achieving sustainable re-use. Engagement on this issue is very difficult to structure in practice, grounding scientific research in 'live' case studies gives a practical context to ensure that the science research is relevant and will support a real decision-making process for communities. We have been working closely with a small Mäori settlement near Taupö, New Zealand. The community have expressed a wish to explore sustainable options for management of their own waste on-site. They are interested in potential solutions that are: low cost and technically robust, simple and easily maintained with the potential for cost recovery. In this study we investigated the viability of vermicomposting the mixtures of wastes produced by the settlement and associated commercial ventures that include: horticultural factory waste, septic tank waste, and wastes from a dairy processing factory. Results to date show that vermicomposting can stabilise the nutrient content of septic tank waste and reduce some microbial contamination. Although complete inactivation of *Escherichia coli* was achieved, some microbes (e.g. campylobacter) are able to survive the vermicomposting process.

# A 6-year perspective on differences in soil properties, production and energy use for organic and conventional dairy farming systems in New Zealand

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The Agricultural Research Group on Sustainability (ARGOS) has compared soil quality and economic and energy efficiency between conventional and organic management of New Zealand dairy farms. A null hypothesis was formed to test the premise that there was no difference in soil fertility properties, energy use, or energy efficiency between the two management systems. Two panels consisting of 12 pairs of matched organic and conventional farms, termed clusters, were monitored for 6-years.

Of the 20 soil properties reported; P, S, pH, cation exchange capacity (CEC), exchangeable-Ca, organic-C and C/N ratio values were significantly different between management systems. Energy inputs and outputs were significantly higher for conventional farms but the energy-return-on-investment (EROI) was not significantly higher for conventional, in part due to the wide range of values within each panel. Although organic farms proved less energy intensive, their production was about 30% lower on average. The ~20% premium paid for organically-produced milk over conventional milk meant the cash farm surplus (MS income minus farm expenses) was not significantly different between panels.

**Keywords**

energy return on investment, energy usage, soil test, pastoral, earthworms

## Introduction

The New Zealand Agriculture Research Group on Sustainability (ARGOS) has been investigating soil quality since 2004 on a range of primary sectors including dairy. A major part of any comparison of sustainability and resilience between agricultural production systems is soil quality and whether an organic system produces fewer deleterious effects than a "conventional" system. Similarly, organic systems may be perceived as more energy and/or economically efficient than a conventional system. We set out to test a simple null hypothesis i.e. that there are no differences in soil quality, energy or economic efficiency between dairy management systems.

## Materials and Methods

*Research design*

The ARGOS programme concentrated on establishing groups (clusters) of commercial farms that were under the target management systems and in close proximity to ensure soils, topography and climate were similar within clusters (Table 1). Soil sampling was conducted from randomly chosen permanent soil monitoring sites (SMS) but from within the dominant landforms identified for each farm. For flat farms only one landform was sampled whilst for hill farms, slope, flat and crest landforms might be sampled. This approach was used to mitigate the effects of large spatial variability within the farms, and individual paddocks formed the minimum management unit (MU).

Initially, ten clusters were set up with each consisting of an organic farm matched with a conventional counterpart but this was increased to twelve clusters in 2006 (i.e. a total of 24 farms). The 12 farms under each management system constituted a panel. All 12 clusters were located in the North Island in the Waikato (4), Taranaki (3), Manawatu (3), Coromandel (1- Waihi) and Auckland (1-Pukekohe) regions.

*Management system*

The organic farms in this study all used accredited organic production protocols and have achieved organic accreditation status although at the time of the initial sampling in 2005 half of the dairy farms were still mid-way through the accreditation process (2-3 years). In the final sampling in 2011, all farms had achieved accreditation for at least four years. Conventional represents comparable farms not specifically using organic protocols.

*Production and energy data*

Comprehensive production and energy usage values were compiled for each farm from annual reviews and from ARGOS reports and other sources that previously provided a detailed resource inventory for each panel (Barber and Lucock 2006; Barber *et al.* 2008; McDowell and Condron 2000; Wells 2001). Energy embodied in outputs (i.e. products) were specifically milksolids (MS) whilst energy inputs per hectare (GJ/ha) included fertiliser, electricity and supplementary feed (Table 2). These inputs embodied the three main imported energy sources used on farm (Wells 2001) whilst cull cows was the only energy output excluded in this analysis. From these, energy-return-on-investment values (outputs/inputs= EROI) were calculated for each management system.

*Soil measurements*

Within each management unit (paddock or block), 3 SMS were randomly selected, the samples bulked and measurements were made for soil bulk density (SBD) (Grossman and Reinsch 1994) and earthworm populations and weights (Fraser *et al.* 1996). Soil fertility (as outlined in Blakemore (1987)) and biological analyses were conducted on a bulked sample of soil from the 3 SMS within each MU. Soil samples were collected from the standard sampling depths for pasture (0-7.5 cm). For brevity we report earthworm, SBD and soil fertility analyses.

*Statistical analysis*

Quantitative data was analysed using a mixed model fitted with restricted maximum likelihood (REML) using GenStat v.12 (GenStat Committee 2009). This method allowed analysis of the entire data-set simultaneously, even though the data was not balanced and there were several gaps where data for some variables was not collected over all the sampling rounds.

A measure of the variability associated with the predicted means is provided by the approximate least significant differences (LSD). LSD's are provided at the 5% level with t=2 (t=2 is justifiable since the data set is large; >250 observations for the soil analysis and >80 observations for the production data). The data was arranged in a hierarchical structure with each fixed effect being assessed at different strata (different levels of random effects) i.e. Random Effects = Cluster, Property, LF, Year; Fixed Effects = Management, LF. We report on overall system means only.

## Results and Discussion

*Soil fertility*

Olsen-P values for the organic panel were consistently less than those for conventional panel but remained in the optimal range (~30). Organic farms, although restricted to the use of slowly-soluble forms of P fertiliser such as reactive phosphate rock (RPR), can maintain optimal P availability as shown here. There is some tendency to slowly decreasing values for the organic panel over the 6-years but it is not currently of concern. Average P use for organic farms was about 25% lower than for conventional although there was a wide range between farms (IQ range 0-48). Conversely, Olsen-P values for the conventional panel were about 50% higher (~50), on average, than the top of the recommended optimal range (20-40) for New Zealand pastures (Roberts *et al.* 1999) and were essentially unchanged over the same period. Soils with such high Olsen-P values soils have a capacity for P loss that might lead to water quality issues from farmland runoff (McDowell *et al.* 2001; Watson and Foy 2001; Wilcock 1986 ; Wilcock *et al.* 2006).

Soil pH, CEC, cation and total base saturation (BS) values were high generally for both management systems but more so for organic (Table 3). With no soluble-N fertiliser applied, and less urine deposition from lower stocking rates, organic farms are likely to have less soil acidification (Table 4). Many of the cluster soils have a significant variable-charge component (Theng 1980) so increased pH often induces an increase in cation exchange capacity (CEC). Sulphate-S values were substantially lower for the organic panel compared with conventional but both reduced significantly over the 6-years although Organic panel values remained in the optimal range (Table 2).

Many soil organic matter (SOM) variables had higher values for the organic panel on average than for conventional but only organic-C was significant and the difference changed little over the 6-years (Table 3). Organic-S was the only exception to this trend, where conventional values were significantly higher and probably resulted from differences in sulphur application rates (~17 kg S/ha/y vs. ~40 kg S/h on average for organic and conventional panels, respectively). Over the monitoring period, however, both actually increased. Shipper et al. (2007) have documented a gradual loss of soil carbon from some New Zealand soils, especially dairy pastures, in the past 20 years but further longitudinal monitoring, and to greater depth, is required to establish whether such a trend might develop here.

Although SBD was lower, and earthworm numbers were greater, for the organic panel overall, these were not significantly different ($P<0.05$). Earthworm mean weights were in fact lower for organic (i.e. organic average worm weight was lower) but again this was not significant.

Potentially (anaerobic) mineralisable-N (AMN) values were, on a per weight soil-N basis, almost the same for both panels indicating no increased loss (or build-up) of organic-N under the organic system. Organic farms have a strong reliance on biological fixation and need to ensure that nutrients important to legume nutrition are kept at optimal levels. To date, this seems to have occurred.

*Production and energy data*

Stocking rates and MS production were 20% and 30% lower, respectively, for the organic panel (Table 4). Income, expenses and cash farm surplus (per ha basis) were largely similar although the range was highly variable within each panel. The premium paid for organic milk production of about 20% largely overcame the production differences to achieve approximate economic parity for both panels. Organic farms were considerably lower users of energy in inputs such as fertiliser. However, lower energy inputs were balanced by lower outputs so that the EROI values were essentially the same for both panels. In conclusion, organic dairy production was less intensive but not more efficient than conventional production.

**Conclusions**

The null hypothesis was supported for a number of variables with notable exceptions in variables tied to fertiliser use. Lower P availability on organic farms was potentially beneficial to the environment. Soil quality was, however, largely similar between panels. Organic management utilises less energy in terms of inputs, so is less intensive that conventional management, but its lower outputs meant that EROI is similar to conventional management. Whilst a premium for organic milk production is maintained, organic dairy farmers remain viable.

**Table 1: Region, mean property size, soil classification and horizon depth (A & B) for each dairy cluster.**

| Cluster | Region [1] | Av. area (ha) | Aspect | Av. soil depth A (cm) | B | NZ Soil [2] Order |
|---|---|---|---|---|---|---|
| 1 | WAI | 96 | hill | 23 | 45 | A |
| 2 | WAI | 50 | hill | 20 | 52 | A |
| 3 | WAI | 89 | flat | 18 | 47 | A |
| 4 | WAI | 167 | hill | 25 | 53 | A |
| 5 | COR | 100 | hill | 26 | 57 | A/Br |
| 6 | TAR | 102 | flat | 27 | 48 | A |
| 7 | TAR | 84 | flat | 19 | 46 | A |
| 8 | TAR | 109 | flat | 22 | 42 | A |
| 9 | MAN | 150 | flat | 24 | 47 | Gl/R |
| 10 | AUK | 133 | flat | 21 | 49 | A |
| 11 | MAN | 93 | hill | 20 | 38 | Gl |
| 12 | MAN | 445 | flat | 18 | 45 | Gl/R |
| Mean | | 135 | | 22 | 47 | |

[1] Region key: AUK= Auckland, WAI= Waikato, TAR= Taranaki, COR= Coromandel, MAN= Manawatu;

[2] Dominant soil types for SMS in cluster farms or orchards. Key to NZ soil order: A= Allophanic, Br= Brown, Gl= Gley.

**Table 2: REML mean and statistics for inorganic soil fertility and physical analyses for dairy organic and conventional panels.**

| System | Year | Olsen-P mg P/L soil | Resin-P mg P/L soil | SO4-S mg S/L soil | pH | CEC cmol+/L soil | Ca | Mg | K | Base saturation (%) Total | Ca | Mg | K | SBD[b] g/cm3 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Organic | 2005 | 33 | 101 | 21 | 6.1 | 19.9 | 12.3 | 2.0 | 1.0 | 76 | 59 | 9.9 | 5.3 | 0.84 |
| | 2007 | 30 | | | 6.2 | | | | | | | | | 0.78 |
| | 2011 | 28 | 90 | 12 | 6.4 | 22.3 | 13.9 | 1.9 | 0.8 | 76 | 63 | 8.7 | 3.7 | 0.78 |
| | Mean | 30 | 96 | 16 | 6.2 | 21.1 | 13.1 | 2.0 | 0.9 | 76 | 61 | 9.3 | 4.5 | 0.80 |
| Conv. | 2005 | 51 | 134 | 28 | 5.9 | 19.0 | 10.5 | 1.7 | 1.1 | 70 | 54 | 9.3 | 5.7 | 0.85 |
| | 2007 | 50 | | | 6.1 | | | | | | | | | 0.80 |
| | 2011 | 51 | 119 | 20 | 6.2 | 20.8 | 10.7 | 1.8 | 0.8 | 66 | 52 | 9.1 | 4.4 | 0.82 |
| | Mean | 51 | 126 | 24 | 6.1 | 19.9 | 10.6 | 1.8 | 0.9 | 68 | 53 | 9.2 | 5.1 | 0.82 |
| System | | *** | ** | ** | * | NS | ** | NS | NS | ** | ** | * | *** | NS |
| | Year | NS | NS | *** | *** | ** | NS | NS | *** | NS | NS | * | *** | *** |
| Sys*Yr | | NS | NS | NS | NS | NS | NS | NS | NS | NS | NS | NS | NS | NS |
| LSDsp [a] | | 9 | 28 | 7 | 0.1 | 2.5 | 2.0 | 0.4 | 0.2 | 7 | 7 | 1.6 | 1.0 | 0.05 |

[a] average LSD for system*year interactions; [b] soil bulk density (0-15 cm).

**Table 3: REML mean statistics for soil organic matter fertility and earthworm analyses for organic and conventional dairy panels.**

| System | Year | C %w/v | N | C/N ratio | Organic-S mg S/L soil | mg S/kg soil-C | AMN[a] kg N/ha | mg N/kg soil-Nc | Earthworms No. #/$m^2$ | Wgt $g/m^2$ |
|---|---|---|---|---|---|---|---|---|---|---|
| Organic | 2005 | 8.5 | 0.81 | 10.5 | 11.1 | 140 | 323 | 29.6 | 584 | 239 |
| | 2007 | 9.0 | 0.90 | 10.1 | | | 260 | 22.5 | | |
| | 2011 | 8.9 | 0.82 | 10.9 | 12.4 | 150 | 290 | 26.6 | 323 | 113 |
| | Mean | 8.8 | 0.84 | 10.5 | 11.7 | 145 | 291 | 26.2 | 440 | 169 |
| Conv. | 2005 | 7.9 | 0.76 | 10.3 | 11.3 | 154 | 311 | 30.0 | 505 | 291 |
| | 2007 | 8.3 | 0.84 | 9.9 | | | 242 | 22.2 | | |
| | 2011 | 8.1 | 0.76 | 10.5 | 14.5 | 186 | 273 | 27.1 | 296 | 123 |
| | Mean | 8.1 | 0.79 | 10.2 | 12.9 | 170 | 275 | 26.4 | 396 | 204 |
| System | | * | NS | * | NS | * | NS | NS | NS | NS |
| | Year | NS | ** | *** | ** | * | *** | *** | *** | *** |
| Yr*Sys | | NS | NS | NS | NS | NS | NS | NS | NS | NS |
| LSDsp [b] | | 0.7 | 0.09 | 0.3 | 2.0 | 23 | 30 | 2.5 | 160 | 60 |

[a]AMN= anaerobic mineralisable-N; [b]average LSD for year*system.

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## THE PHYSICAL SOIL AND SOIL WATER

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# A quantitative and functional soil classification for modelling the terrestrial water balance of north-eastern Australia

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Abstract

Tasked with modelling sediment, nutrient and pesticide transport from the land to the Great Barrier Reef lagoon (GBR), we developed a functional soil classification system based on soil hydrologic characteristics such as infiltration capacity and plant-available water holding capacity. Sixteen Functional Orders (FOs) describe the top level of the two-level hierarchical system, with the lower level (soil groups) differentiated by depth of the A and B horizons and soil permeability. Of the 448 soils groupings theoretically possible, 341 exist in the GBR catchments. The Australian Soil Resource Information System (ASRIS, www.asris.csiro.au) was interrogated and statistically analysed to provide data for the soil groups. Some data, such as runoff curve number, are not in ASRIS, and were allocated across groups at the FO level according to expert opinion. A total of 58,040 soil polygons in the GBR catchments were each assigned to one of the 341 soil groups.

**Keywords**

soil parameters, water balance, ASRIS, classification

## Introduction

The Great Barrier Reef (GBR) is an ecologically and economically important part of north-eastern Australia. It is vast (more than 25,000 km$^2$ of coral reef) and diverse, with more than 70 distinct biological zones (Holland 2011), and much of it is under threat from sediment, nutrient and pesticide runoff from agricultural land use in the adjacent catchments (Waterhouse *et al.* 2010). The Australian and Queensland governments have implemented 'Reef Plan' to improve the health of the GBR by setting end-of-river water quality targets and fund the necessary changes in land management (State of Queensland 2009). To report on progress towards the end-of-system water quality targets, a modelling program has been implemented. Modelling is well suited to predicting changes in the water balance and water quality benefits of changes to land management (Robinson *et al.* 2010).

Models are widely used for estimating the past and future water balance and transport of sediment, nutrients and pesticides. The HowLeaky model (Rattray *et al.* 2004, Robinson *et al.* 2010) is more comprehensive than most and contains detailed sub-models of all of the processes of interest: sediment, nutrient and pesticide losses from agricultural land (Shaw *et al.* 2011, Robinson *et al.* 2011).

Soil characteristics are important for modelling runoff, deep drainage, erosion and other environmentally important processes. Infiltration capacity, water-holding capacity, permeability and similar characteristics are the physical measures of soil's roles in environmental processes. However, despite long-standing knowledge of these relationships, soil descriptions and classifications are often weighted towards pedological factors such as soil colour, morphology and inter-horizon differences and transitions. Our aim was to develop model parameters for soil groups defined by quantitative, *functional* characteristics, especially those recorded in comprehensive soil databases that are based on measurements or reliable pedotransfer functions.

## Methods

### *Functional Orders*

A high-level classification of soil units was required to separate groups that are most different in terms of hydrologic behaviour. Runoff is of prime concern, as it is the pre-cursor to soil erosion and most nutrient and pesticide movement, as well being the loss process for water that would otherwise transpire, evaporate or drain beyond the root zone. Orders of the Australian Soil Classification (ASC) (Isbell 2002) considered similar in terms of runoff and major physical characteristics were grouped into units called Functional Orders (FO, Table 1). Key determinants of a soil's FO are properties such as macro-porosity and sodicity that affect rainfall infiltration, as well as plant-available water capacity (PAWC).

An example of clustering then splitting Orders of the ASC occurs in our Sodosol FOs (8, 9 and 10, Table 1). These include sodic examples of the ASC's Chromosols, Kurosols, Kandosols and Calcarosols, which are functionally similar. Together, this collection of "functionally Sodosol" soils includes loamy-surfaced soils where their high proportion of fine sand results in surface seals and low infiltration rates (P. Wilson, pers. comm.). Therefore, we separated our Sodosols with loamy surface textures from those with sandy surface texture, and attributed high runoff curve numbers to them. Furthermore, Sodosols with deep sandy surfaces have low PAWC, and these were separated from soils with shallow sandy surfaces.

**Table 1: Rules for assigning soil polygons to Functional Orders (FO), using data from ASRIS**

| FO | Functional Order name | ASC Orders | Required conditions within ASC Orders |
|---|---|---|---|
| 1 | Heavy Vertosols | Vertosol | Surface clay% >=50 |
| 2 | Dermosols (structured clay/ clay loam surface) | Dermosol | A horizon texture = clays and clay loams |
| 3 | Dermosols (sealing loamy surface) includes non-sodic: Chromosols, Kurosols, Kandosols and Calcarosols | Dermosol | A horizon texture = mainly silty loams |
| | | Chromosol, Kurosol, Kandosol, Calcarosol | B horizon Exchangeable Sodium Percentage (ESP)<15% and A horizon texture = loams and clays |
| 4 | Dermosols (sandy surface) includes non-sodic: Chromosols, Kurosols, Kandosols and Calcarosols | Dermosol | A horizon texture = sands and fine sands |
| | | Chromosol, Kurosol, Kandosol, Calcarosol | B Horizon ESP<15% and A horizon texture = sands and fine sands |
| 5 | Hydrosols (structured clay/ clay loam surface) including Organosols | Hydrosol, Organosol | A horizon texture = mainly clays and clay loams |
| 6 | Hydrosols (sealing loamy surface) | Hydrosol | A horizon texture = mainly silty loams, some clay loams |
| 7 | Hydrosols (sandy surface) | Hydrosol | A horizon texture = sands, loamy sands, clay sands |
| 8 | Sodosols (loamy surface) including sodic: Chromosols, Kurosols, Kandosols, Calcarosols | Sodosol | A horizon texture = mainly loams, loamy sands, fine sands, sandy clays |
| | | Chromosol, Kurosol, Kandosol, Calcarosol | B horizon ESP>15%, A horizon texture = mainly loams, sandy loams, silty loams, clay loams, some clays |
| 9 | Sodosols (moderately deep (>0.5 m) sandy surface) including sodic: Chromosols, Kurosols, Kandosols, Calcarosols | Sodosol | B horizon depth >0.5 m, A horizon texture = sandy |
| | | Chromosol, Kurosol, Kandosol, Calcarosol | B horizon depth>0.5 m, B horizon ESP>15%, A horizon texture = sandy |
| 10 | Sodosols (shallow (<=0.5 m) sandy surface) including sodic: Chromosols, Kurosols, Kandosols, Calcarosols | Sodosol | B horizon depth<0.5 m, A horizon texture = sandy |
| | | Chromosol, Kurosol, Kandosol, Calcarosol | B horizon depth<0.5 m, B horizon ESP>15%, A horizon texture = sandy |
| | | Calcarosol | B horizon ESP>15%, A horizon texture = loamy sand |
| 11 | Tenosols, Rudosols and Podosols (all sandy) | Tenosol, Rudosol | A horizon texture = sandy |
| 12 | Rudosols/Tenosols (loamy) | Rudosol, Tenosol | A horizon texture = mainly loams, loamy sands, clay fine sands, sandy clays, clay loams |
| 13 | Ferrosols | Ferrosol | |
| 14 | Unclassified | | |

| | | | |
|---|---|---|---|
| 15 | Medium Vertosols | Vertosol | Surface clay% >=40 and <50 |
| 16 | Light Vertosols | Vertosol | Surface clay% <40 |

For the 16 FOs that occupy the top level of the hierarchical system (Table 1), a few data, including runoff curve number, were allocated at FO level using expert opinion. When modelling the GBR catchments, some attributes that were sparsely represented in spatial terms were also assigned based on the mean for each FO; for example, available phosphorus (Colwell test, mg/kg).

**Table 2: Runoff curve numbers (antecedent moisture condition 2) for Functional Orders, categorised by profile permeability**

| Functional Order | Very slowly permeable | Slow, medium or fast |
|---|---|---|
| Ferrosols | 74 | 74 |
| Tenosols/Rudosols/Podosols (sandy surface) | 78 | 78 |
| Dermosols (structured clay/clay loam surface), Dermosols (sandy surface) including non-sodic Chromosols/Kurosols/Kandosols | 78 | 78 |
| Hydrosols (structured clay/clay loam surface), Hydrosols (sandy surface) | 78 | 78 |
| Light Vertosols (clay <= 45%), Medium Vertosols (45% < clay <= 55%), Heavy Vertosols (> 55% clay) | 84 | 78 |
| Dermosols (sealing loamy surface) including non-sodic Chromosols/Kurosols/Kandosols | 88 | 88 |
| Unclassified | 88 | 84 |
| Hydrosols (sealing loamy surfaced) | 88 | 88 |
| Rudosols/Tenosols (loamy surface) | 88 | 88 |
| Sodosols (loamy surface), Sodosols (mod deep (>0.5 m) sandy surface), Sodosols (shallow (<0.5 m) sandy surface) including sodic Chromosols/Kurosols | 94 | 88 |

*Parameter estimation for each soil polygon*

Sub-units of FOs, known as soil groups, were differentiated by depth of the A and B horizons, and soil profile permeability. Soil polygons were defined through the Australian Soil Resource Information System (ASRIS, Brough *et al.* 2006) and each assigned, according to their physical characteristics, to a soils group. Of the 448 soils groups possible, 341 were found to exist in the GBR catchment.

Soils attribute data in the ASRIS database was interrogated to provide the parameters needed to model each of these soils groups. These included soil hydraulic parameters such as drained upper limit, plant-extractable lower limit, and air dry moisture contents. These values are the mean of the individual parameters assigned to each polygon in each group. While this provided sensible and useful parameters, there was still a requirement for some generic rules. For example, some instances of low soil water content at saturation (SAT, %) were adjusted upwards to a minimum feasible value (Drained Upper Limit+5).

**Results**

Figure 1 shows the spatial distribution of soil Functional Orders (FO) in the Burnett-Mary catchment. Significant variation in the spatial resolution of the FOs is easily seen. Each of the soil polygons in the GBR catchments were assigned to one of 341 soil groups. There are many soil polygons within each region (for example 13,500 in the Burnett-Mary and 58,040 overall). Statistical averaging for the groups resulted in the loss of some detail - measurements in a particular polygon were sometimes replaced with an average. However, the averaging results in parameters with consistent quality and importantly, provides parameters in the vast majority of polygons for which there are no measurements.

**Discussion**

Soil functional attributes and spatially distributed data for soil groups were used to develop soil parameter files suited to water balance and erosion models such as HowLeaky? (Rattray *et al.* 2004). In comparison to methods that derive soil parameters by comparison, extrapolation and interpolation from well-known, ‘reference’ soil types, we confidently predict that this method provide more accurate and reliable parameters for the many soils that lack a well-known “relative”. It will, however, be some time before this can be tested objectively, due to the effort needed to collect the necessary field data for testing. However, some advantages of our method are already clear: it is less subjective, more transparent, more repeatable and can be improved through time, as new field data are entered into ASRIS and new relationships are developed between parameters.

Importantly, ASRIS is based on measurements on thousands of soils. That survey work is ongoing and the approach used here is able to be repeated and improved as additional soils data becomes available.

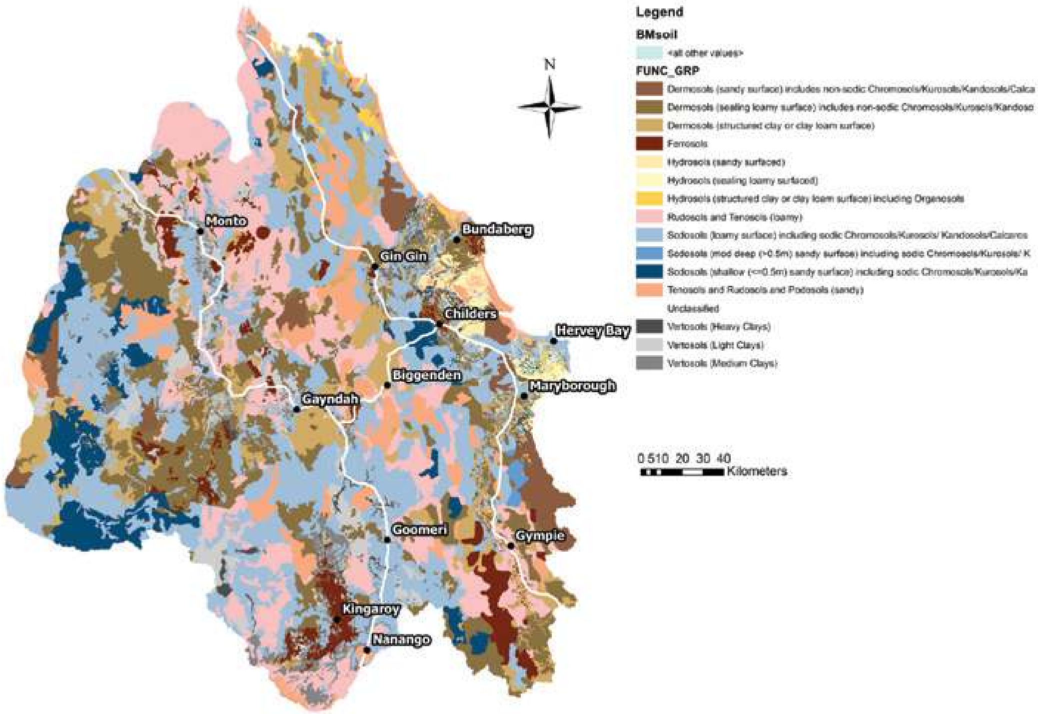


**Figure 1: Distribution of 16 Functional Orders in the Burnett-Mary region of the GBR catchments.**

**Acknowledgements**

We are grateful to Peter Wilson (Bundaberg) who was instrumental in defining the Functional Orders, and the anonymous referee whose help was much appreciated.

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# Assessing water quality from farms – how much detail is required for a model to be useful?

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## Abstract

Agriculture is being implicated in declining water quality and its impact on natural assets such as the Great Barrier Reef and the Murray-Darling river system. With a paucity of empirical data to support impacts of practice change, water-balance simulation models are used to extrapolate empirical data. These models are used routinely to synthesis data from experimental studies and then used to extrapolate to a range of land use and soil types well beyond experimental conditions. A common challenge for modellers is how best to describe a land use system in terms of model parameters.

This paper describes the impact of input detail on predictions of runoff, soil loss and water quality. We find there is scope for using models that describe land use systems and soil types that are less demanding then a literal interpretation of system conditions. Two applicationsof a model are used to demonstrate this proposition. Greater detail in a model or its inputs does not necessarily result in greater confidence in estimates. A model with fewer inputs can be easier to set-up, diagnose and apply. If there are few trade-offs in simplicity, there are likely to be efficiencies to data collection and model application.

## Introduction

Agriculture is increasingly being expected to demonstrate its environmental credentials to governments and the community. As an example, sugar cane, grain and horticulture producers and graziers within the Great Barrier Reef (GBR) catchments are being encouraged to adopt practices that aim to reduce sediment, nutrient and pesticide loads reaching the lagoon by 20-50% within a target date.

While farmers have made many changes in agronomy in order to improve water use efficiency and soil stability, we have little basis for demonstrating in quantitative terms that there has been positive progress in water quality outcomes. Indeed, with greater use of fertiliser and pesticides there is a community perception that contamination of the environment has increased. High rates of adoption of new technologies such as conservation tillage provide evidence that farmers are adaptive and open for improvement (D'Emden and Llewellyn 2006). Empirical evidence of improved water quality is scarce because field studies are expensive and require several years to collect representative samples of climate given the episodic nature of runoff and pollutant movement.

Water balance models are a standard tool for estimating hydrology and water quality attributes of land use systems, often relying on limited studies to "calibrate" the model. A key step in calibrating a model is determining how best to describe system conditions encountered in the real world.Where does the user source sensible parameter values to apply in a model?

This paper describes the impact of using a range of input detail on model performance. Two key inputs will be explored; the level of detail in soil water description and vegetation description.

## Methods

*Vegetation description for grazing study.*

In order to explore the impact of level of vegetation description on model estimates of runoff and soil loss for grazing systems, the Howleaky model (V5.37.06, McClymont *et al*. 2011) was applied to data from the Mt Mort pasture study conducted by the Queensland Department of Natural Resources and Mines (1991–2000) (DM Silburn pers. Comm., Rattray *et al*. 2006). Runoff and soil loss were measured on three land management practices (grazed, exclosed and bare soil treatments) at "Old Hidden Vale", Mt Mort, 27 km SSE of Gatton in southern Queensland.

Three levels of vegetation and soil description were applied to simulate runoff and soil loss at Mt Mort (Figure 1):

- A time series of measured green and residue covers and optimised soil descriptions for each treatment (a literal re-enactment of the experiment);
- The above vegetation description and a site average soil description; and
- A generic mean monthly green and residue cover distribution reflecting pasture growth and residue for each treatment and a single soil description as above.

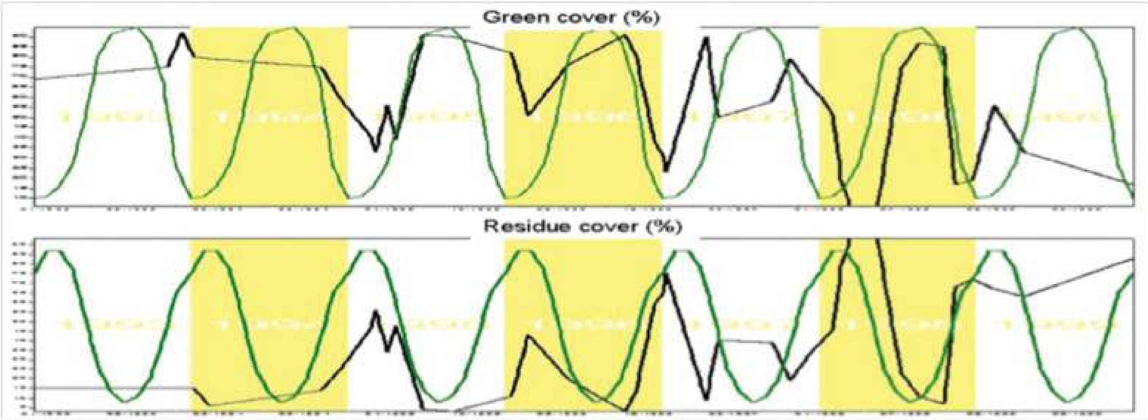


**Figure 1: Two levels of cover description for the exclosed treatment used in testing Howleaky: measured time series of observed cover (black line); and a generic seasonal distribution which captures typical seasonal variation (green line). Green and residue cover are both described.**

*Soil description for cropping study*

In order to explore the impact of level of soil description on model estimates of runoff and soil loss in cropping systems, three levels of detail of soil water description were used to simulate a winter cropping regime near Wallumbilla in south western Queensland (Freebairn *et al*. 2009). Two contrasting tillage - fallow stubble cover regimes were used to demonstrate hydrology and water quality responses to fallow management conditions.

Plant available water capacity (PAWC) and internal drainage are derived from wilting point, drained upper limit and total porosity of soil layers. Air dry and saturated content are used to describe evaporation store and deep drainage respectively.

Soil is described using three levels of detail shown in Figure 2:

- six soil layers with all four parameters for each layer, (n=24 where n=number of parameters);
- four soil layers,(n=12); and
- A pseudo three layer soil specified by: available soil water range of the surface layer; "evaporation depth"; and rooting depth (n=3).

The third soil description method ignores soil water below wilting point and represents a very simplistic view of soil water stores.

Approximately 5-12 other variables were required to describe evaporation, runoff and drainage processes. The three soil descriptions used here were constructed to have similar PAWC values. Hydrology, erosion and water quality estimates were used to compare the results for the various levels of input.

**Results**

*Vegetation descriptions for grazing*

Table 1 presents observed and predicted runoff and soil loss for three grazing treatments, each with three levels of detail in system description. The model was able to predict the impact of management on hydrology and erosion with reasonable reliability and estimates for the three levels of description are similar. Figure 3 presents a cumulative time series of runoff for the data summarised in Table 1.

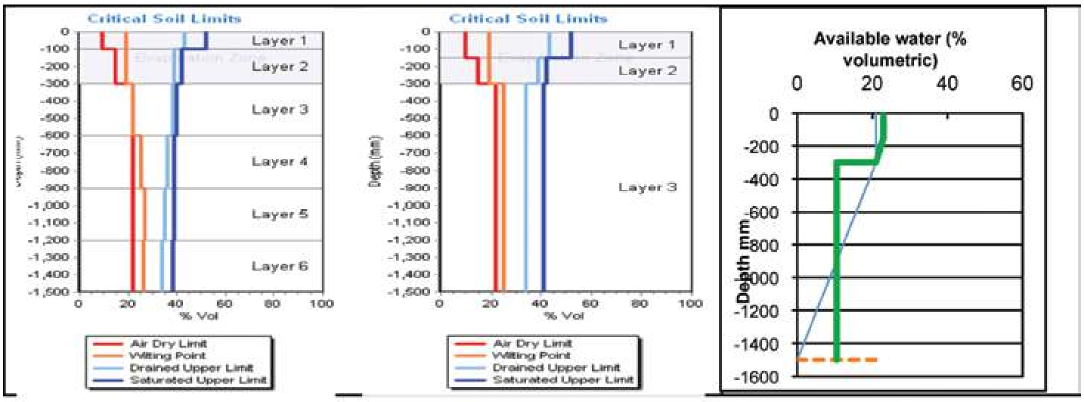


**Figure 2: Three levels of soil water description for a brown sodosol near Wallumbilla: a full description with six layers (ApSoil profile No. 064); the same soil with 3 layers; and the much simplified soil water description based on three input values.**

**Table 1: Summary of Observed and Predicted Average Annual Runoffand Soil Loss for the Mt MortGrazing Trial (1993-1999) with Three Levels of System Description.**

| Description of Management system and model parameterisation | Observed | | Predicted | |
|---|---|---|---|---|
| | Runoff (mm) | Soil loss (t/ha) | Runoff (mm) | Soil loss (t/ha) |
| Level 1 model description (time series of measured green and residue cover and individual soil type) | | | | |
| Bare | 136 | 46 | 159 | 50 |
| Grazed | 22 | 0.5 | 18 | 0.7 |
| Exclosed | 3 | 0.06 | 4 | 0.1 |
| Level 2 model description (generic time series of green and residue cover and sitesoil type) | | | | |
| Bare | 136 | 46 | 147 | 65 |
| Grazed | 22 | 0.5 | 15 | 0.4 |
| Exclosed | 3 | 0.06 | 12 | 0.2 |
| Level 3 model description (generic time series of green cover, residue cover and soil type) | | | | |
| Bare | 136 | 46 | 142 | 54 |
| Grazed | 22 | 0.5 | 20 | 0.7 |
| Exclosed | 3 | 0.06 | 17 | 0.2 |

**Wallumbilla–cropping**

The impact of soil description on runoff, soil loss and sediment concentration estimates for cropping under two levels of soil management is presented in Table 2. Soil with six or three layers resulted in similar hydrology, erosion and sediment estimates while the simplified soil water description estimated slightly higher runoff. In terms of estimating differences between management practices, all three soil descriptions provide similar estimates in terms of absolute values and relative rankings.

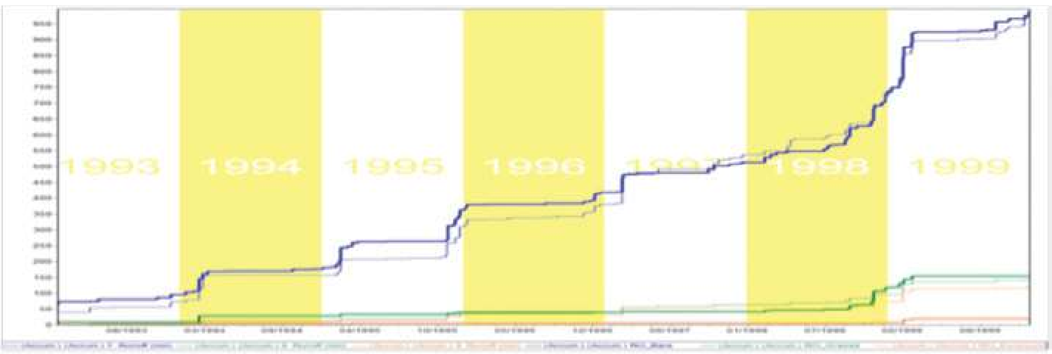


**Figure 3: Simulated (bold lines) and measured cumulative runoff (mm) for three grazing treatments: bare (blue); grazed (green); and exclosed (orange) at Mt Mort, QLD (1993 - 1999).**

**Table 2: Summary of Runoff, Erosion and Event Mean Sediment Concentration for Two Cropping Systems with Three Levels of Soil Description**

| Soil description ‘n’ refers to the number of parameter values in each description | Runoff (mm) | Hill slope soil loss (t/ha) | Sediment Event Mean Concentration (g/L) |
|---|---|---|---|
| High tillage, low residue retention | | | |
| Full soil description, 6 layers n= 32 | 48 | 5.2 | 1.5 |
| Partial soil description 3 layers n=19 | 47 | 5.4 | 1.1 |
| Simple soil description n= 10 | 69 | 8.2 | 1.2 |
| Low tillage, high residue retention | | | |
| Full soil description, 6 layers n= 32 | 65 | 0.27 | 0.06 |
| Partial soil description 3 layers n=19 | 64 | 0.34 | 0.05 |
| Simple soil description n= 10 | 85 | 0.64 | 0.08 |

## Conclusion

One conclusion from these two case studies is that while there may be differences in estimates of hydrology, soil loss and water quality from a model with a range of detail in system description, the differences between model estimates are small compared to difference in hydrology and water quality associated with soil management. It is unlikely that simplified model inputs will result in serious errors in estimates of hydrology, soil loss or water quality.

A generic set of model parameters is likely to be more useful and transferable than a parameter set that describes an experiment explicitly, especially since vegetation patterns from an experiment are unlikely to be repeated and soil types are spatially variable.

This paper does not propose that model inputs should be simplified *per se* but it does suggest that we can be more pragmatic in describing systems at least in terms of vegetation and soil water holding properties. Vegetation can be described by a generic green and residue cover pattern (mean monthly values) while soil water description requires a reasonable estimate of PAWC. One advantage of using water balance as a foundation for estimating water quality is that these models provide realistic estimates of water excess (runoff and deep drainage) and timing of excesses using qualitative assessments of soil properties, vegetation patterns and weather data.

We have found that if soil PAWC can be estimated from texture and rooting depth, and combined with descriptions of vegetation and of the soil’s propensity to crust at the surface, then estimates of hydrology and water quality responses to management can be achieved for a wide range of environments with reasonable confidence and efficiency.

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# Soil water storage and drainage under irrigated cotton sown on permanent beds in a Vertosol with subsoil sodicity

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**Abstract**

Comparative studies of drainage and leaching in irrigated cotton-based cropping systems in Australian Vertosols are sparse. Our objective was to quantify soil water storage, drainage and leaching in four cotton-based cropping systems sown on permanent beds in an irrigated Vertosol with subsoil sodicity. Drainage was inferred using the chloride mass balance method and soil water storage with a neutron moisture meter from September 2005 to May 2011. The experimental treatments were: cotton monoculture (CC), cotton-vetch (CV), cotton-wheat where wheat stubble was incorporated (CW), and cotton-wheat-vetch where wheat stubble was retained as *in-situ* mulch (CWV). Soil water storage was generally highest under CW and CWV and least with CV. An untilled short fallow (~3 months) when combined with retention of crop residues as surface mulch as in CWV was as effective in harvesting rainfall as a tilled long fallow (~11 months) with stubble incorporation in CW. Drainage under cotton was generally in the order of CW ≥ CWV > CC = CV >>> fallows. No definitive conclusions could be made with respect to the effects of cropping systems on salt and nutrient leaching. Leachate contained less nitrate-N, Mg and K, but $EC_w$ was ~6 times higher than in irrigation water. The greater salinity of the leachate may pose a risk to shallow groundwater resources.

**Key words**

Vertisol; Haplustert; Permanent bed; Stubble retention; Rotation; Hydrology

## Introduction

Comparative studies of drainage and leaching in irrigated cotton (*Gossypium hirsutum* L.)-based cropping systems in Australian Vertisols are few. Weaver *et al*. (2005) observed that drainage in a saline-sodic Vertisol were greater under crop sequences that included crops tolerant to sodic conditions (cotton, wheat (*Triticum aestivum* L.)) than when a sensitive crop (dolichos (*Lablab purpureus* L.)) was part of the sequence. Similarly, in a Vertisol where sodic conditions existed at a depth of approximately ≥1m, drainage under a cotton-wheat rotation was greater (Hulugalle *et al*., 2010) than those under continuous cotton. The greater drainage under the cotton-wheat rotation may have resulted in better leaching of sodium salts out of the root zone. The ability of crops to penetrate to depth, and thus create preferential flow pathways such as soil cracks, biopores and slickensides through which drainage and leaching are facilitated appears to be a significant driver of drainage in the previously-mentioned studies (Hulugalle *et al*., 2010; Ringrose-Voase and Nadelko, 2010). Most authors concur, however, that the primary pathway of drainage and leaching in clay soils is preferential flow through soil cracks, slickensides and biopores.

The objective of this study was to quantify soil water storage and drainage using the chloride mass balance method in four irrigated cotton-based cropping systems sown on permanent beds in a Vertisol with subsoil sodicity. We used the chloride mass balance method as it could be easily applied on a plot scale, was far less costly and caused little disturbance to individual plots relative to lysimeters, addressed issues such as spatial variability which cannot be done with lysimeters, and was more accurate than Darcian flux and water balance methods (Willis *et al*., 1997). We acknowledge that although it may underestimate drainage due to preferential flow under unsaturated conditions, in our particular situation the advantages far outweighed the disadvantages. The experiment commenced in August 2002, but results presented in this report pertain to the period September 2005 to May 2011.

## Materials and Methods

The experiment was located at the Australian Cotton Research Institute, near Narrabri (149°47'E, 30°13'S), NSW, which has a sub-tropical semi-arid climate. The soil at the experimental site is a self-mulching, endohypersodic, grey Vertosol; very fine (Isbell, 2002). Mean particle size distribution in the 0-1.2 m depth was: 64 g/100g clay, 11 g/100g silt and 25 g/100g sand. Exchangeable sodium percentage, ESP, in the 0.6-

1.2 m depth was 12 and electrochemical stability index, ESI, 0.02. ESP did not exceed 6 in the shallower depths.

The experimental treatments consisted of four cotton-based rotations sown on permanent beds: cotton monoculture (CC), cotton-vetch (CV), cotton-wheat where wheat stubble was incorporated into the beds after harvest with a disc-hiller (CW), and cotton-wheat-vetch where wheat stubble was retained as an *in-situ* mulch into which the following vetch crop was sown (CWV). Vetch (*Vicia* spp.), which is a prostrate, leguminous crop, in cotton-vetch and cotton-wheat-vetch was killed during or just prior to flowering through a combination of mowing and contact herbicides, and the residues retained as *in situ* mulch into which the following cotton was sown. The experiment was laid out as a randomized complete block with three replications and designed such that both cotton and rotation crop phases in the last two rotation treatments were sown every year. Individual plots were 165 m long and 20 rows wide. The rows (beds) were spaced at 1-m intervals with vehicular traffic being restricted to the furrows. Cotton and rotation crops were furrow irrigated when rainfall was insufficient to meet evaporative demand. Irrigation frequency was dependent on in-crop rainfall, availability of irrigation water and stored soil water. Average irrigation rate was 1 ML/ha (= 100 mm) of water.

Except during the 2006-07, 2008-09 and 2010-11 seasons, when volumetric soil water content at cotton sowing was measured on soil cores extracted from the 0-1.20 m depth, water content at 0.20 m, 0.40 m, 0.60 m, 0.90 m and 1.20 m was measured with a neutron moisture meter (CPN 503-DR Hydroprobe®) which had been calibrated *in situ* (Greacen, 1981). Surface soil water content was measured gravimetrically. Soil chloride concentration was evaluated in samples taken after cotton sowing and after cotton picking. Four 50-mm diameter soil cores were extracted from each plot with a tractor-mounted soil corer from the 0-0.10 m, 0.10-0.30 m, 0.30-0.60 m and 0.60-1.20 m depths. Air-dried soil was passed through a 2 mm-sieve and chloride concentration determined by $AgNO_3$ titration of a saturation extract (Beatty and Loveday, 1974). Water sampled from the head-ditch during each irrigation was analysed for chloride by titrating with $AgNO_3$ (Rayment and Higginson, 1992). Irrigation water was alkaline, salinity relatively low and sodium adsorption ratio (SAR) generally moderate. Average (± sd) $pH_w$ was 8.3±0.40, $EC_w$ 0.4±0.06 dS/m and SAR 3.2±1.46. Drainage was inferred with the chloride mass balance method. Previously published models (USSL, 1954; Slavich *et al*., 2002) were used to estimate deep drainage assuming steady state conditions and transient state conditions. Model validity was assessed after Weaver *et al*. (2005). Results were analysed using analysis of variance for a randomised block design, and means and standard errors of the means were calculated.

## Results and Discussion

### *Soil water storage and infiltration*

Among cropping systems, soil water storage was generally more under rotations that included wheat crops such as CW and CWV (Fig. 1) and less with those that did not (CV, CC). Differences among cropping systems were greatest when rainfall was the major source of water used by the crop such as the commencement of the cotton season. When a major proportion of water requirements were supplied by irrigation, however, similar but smaller differences in soil water storage were present.

Factors such as the total amount of rainfall during the intervening fallow since the previous crop, presence or absence of stubble cover, intensity of soil disturbance and rotation crop may influence the differences observed during the early part of the cotton season. Average rainfall during the fallow that preceded cotton from 2005 to 2010 was of the order 46 mm in CV, 229 in CC, 424 mm in CW and 101 mm in CWV. The retention of the vetch and wheat as surface mulch in CWV, however, negated the benefits of the additional fallow rain in cotton-wheat by reducing evaporation and improving infiltration (Scott *et al*., 2010). In addition, soil disturbance associated with incorporating wheat stubble in CW may have enhanced water evaporation and runoff and inhibited infiltration. In summary, a shorter fallow (average of ~3 months) when combined with retention of crop residues as surface mulch and avoidance of tillage was as effective in harvesting rainfall as a longer fallow (average of ~11 months) with stubble incorporation and tillage.

The relatively low soil water storage values in CV during the initial 4-6 weeks of the cotton season appears to be largely due to the very short fallow (average of ~3 weeks) period between killing vetch and sowing cotton, and thus the least fallow rain. Soil water storage values throughout the season, even after irrigation, in CV and CC were generally lower than those in CW and CWV, and indicate that wetting up did not occur to the same extent as in the cropping systems where wheat was sown. We suggest that this may be because of poorer soil structural condition, and consequently, lower infiltration and water-holding capacity in the CV and CC.

*Drainage*

Drainage in cropped plots was many times higher than that in fallowed plots and reflects the higher water inputs in the former (Fig. 2). Although there were large variations among seasons, CW and CWV had generally higher drainage than either CC or CV. Previous studies in other sites have reported that drainage pore numbers (Hulugalle *et al*., 2005) were higher in soils where wheat had preceded cotton. Drainage decreased sharply with depth, particularly between the 0.30-0.60 m and 0.60-1.20 m depths (Fig. 2). Average drainage in the 0.30-0.60 m depth of plots sown to cotton was of the order of 76 mm and in the 0.60-1.20 m depth, 29 mm. A large part of this decrease was probably due to the high ESP (> 6) and low ESI (< 0.05) values (Hulugalle *et al*., 2012). ESI values < 0.05 are indicative of dispersive soils that have low rates of hydraulic conductivity. Highest seasonal drainage occurred during 2006-07 and 2010-11 (Fig. 4), seasons with contrasting climatic conditions. A dry winter preceded a summer virtually devoid of rainfall during the 2006-07 cotton season, thus necessitating frequent irrigation, whereas the 2010-11 season was characterised by frequent and high rainfall events between October and January (Fig. 1). A high frequency of early season irrigation, high rainfall events or a combination of the two is claimed to enhance deep drainage in furrow-irrigated cotton in Vertosols (Gunawardena *et al*., 2011).

## Conclusions

Soil water storage was higher under rotations that included wheat crops (CW, CVW) and less with CC and CV. Differences were greatest when rainfall was the major source of water used by the crop. A short fallow (average of ~3 months) when combined with retention of crop residues as surface mulch and avoidance of tillage (CWV) was as effective in harvesting rainfall as a long fallow (average of ~11 months) with stubble incorporation and tillage (CW). Drainage under cotton was generally in the order of CW ≥ CWV > CC = CV >>> fallows. The most efficient cropping system in terms of water conservation was CWV rotation with *in situ* stubble mulching.

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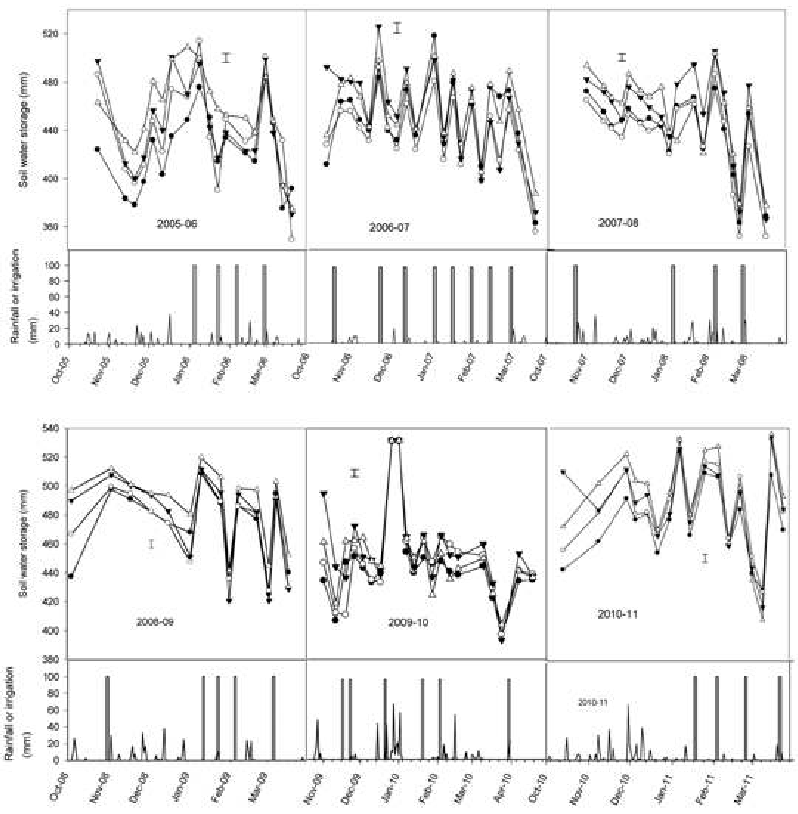

**Fig. 1. Variation of soil water storage in the 0-1.2 m depth with cropping system during the cotton seasons of 2005-06, 2006-07, 2007-08, 2008-09, 2009-10 and 2010-11. ●, Cotton-vetch; ○, Cotton-cotton; ▼, Cotton-wheat. △, Cotton-wheat-vetch. Vertical bar is standard error of the mean (cropping system x date).**

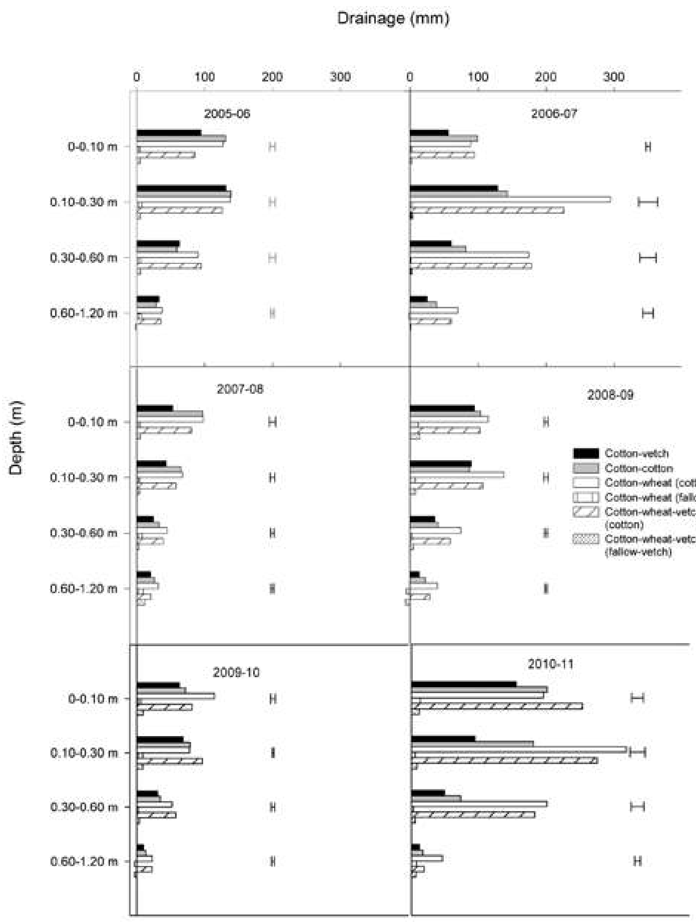

**Fig. 2. Variation of drainage with cropping system and depth during the cotton seasons of 2005-06, 2006-07, 2007-08, 2008-09, 2009-10 and 2010-11. Horizontal bar is standard error of the means among cropping system for a given depth.**

# Influence of soil water status and compaction on $N_2O$ and $N_2$ emissions from synthetic urine

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## Abstract

We conducted a laboratory experiment to quantify $N_2O$ and $N_2$ emissions during three saturation/drainage cycles (from 0 to -10 kPa tension) from repacked soil cores compacted at pressures of 0, 220kPa and 400kPa and treated with or without $^{15}N$-labeled synthetic urine applied at 600 kgN ha$^{-1}$ (enrichment of 50 atom%). Daily gas fluxes of $N_2O$, $N_2$ and $CO_2$ were quantified using mini-headspace chambers placed over the cores. Soils sampled prior to each cycle and on completion of the experiment were analysed for mineral-N, dissolved organic-C and pH.

$N_2O$ emissions were low until the third saturation/drainage cycle, with total emissions highest in the 220kPa compaction treatment. $N_2$ was being emitted by 7d, with total emissions highest in the 400kPa treatment. Maximum $N_2O$ and $N_2$ emissions occurred close to saturation and declined as soils drained to -10 kPa. The ratio of $N_2O$ to $N_2$ emitted during denitrification was linked to factors such as soil pH, soil water status, $NO_3$-N concentration and C supply. Our results suggest that the volume fraction of water (VWC) is a better metric than water filled pore space (WFPS) for predicting $N_2O$ emissions across soils with varying bulk density.

## Introduction

Urine deposition onto pasture is New Zealand's largest source of nitrous oxide ($N_2O$) emissions, representing *c*. 80% of the $N_2O$ emissions inventory including direct and indirect emissions (de Klein *et al.* 2003). Denitrification occurring in anaerobic soils is the major process leading to $N_2O$ production. Soil water status is a key determinant of these emissions as it influences air-filled porosity and oxygen diffusion into and through the soil.

The aim of this research was to develop better understanding of the role that soil physical characteristics and changing soil water status play in regulating $N_2O$ emissions. This knowledge will be used to develop practical tools for predicting when there is greatest risk of $N_2O$ emissions from urine patches.

## Material and methods

Top soil (0-10 cm) from a Horotiu silt loam (Typic Orthic Allophanic Soil) under rye grass-clover pasture was collected, coarsely sieved (> 10mm) and repacked into soil cores (ID = 8.5 cm, 4cm depth), soil properties are given in Table 1. These repacked cores were moistened to -10kPa tension and compacted at pressures of 0, 220 kPa and 400 kPa and treated with or without $^{15}N$-labeled synthetic urine applied at 600 kg N ha$^{-1}$ (enrichment of 50 atom%). Soil cores were then subjected to three successive, 12-day saturation/drainage cycles from 0 to -10 kPa tension (Fig. 1). The tension was controlled on tension tables. Daily gas fluxes of $N_2O$, $N_2$ and $CO_2$ were quantified using mini-headspace chambers placed over the cores. Soils sampled prior to commencing the 2$^{nd}$ and 3$^{rd}$ cycles and on completion of the experiment were analysed for inorganic-N, dissolved organic-C (DOC) and pH. Soil water status was also monitored daily and changes in VWC, WFPS, air filled porosity (AFP) generalized density corrected diffusivity (GD diffusivity) and matric potential (MP) were calculated.

**Table 1. Some initial chemical and physical properties of the soil used.**

| Sand (> 53 µm) | Silt (2-53 µm) | Clay (< 2 µm) | C | N | pH | Inorg-N |
|---|---|---|---|---|---|---|
| | % | | g kg$^{-1}$ | | | mg kg$^{-1}$ |
| 44 | 51 | 5 | 71.2 | 7.8 | 6.1 | 11.8 |

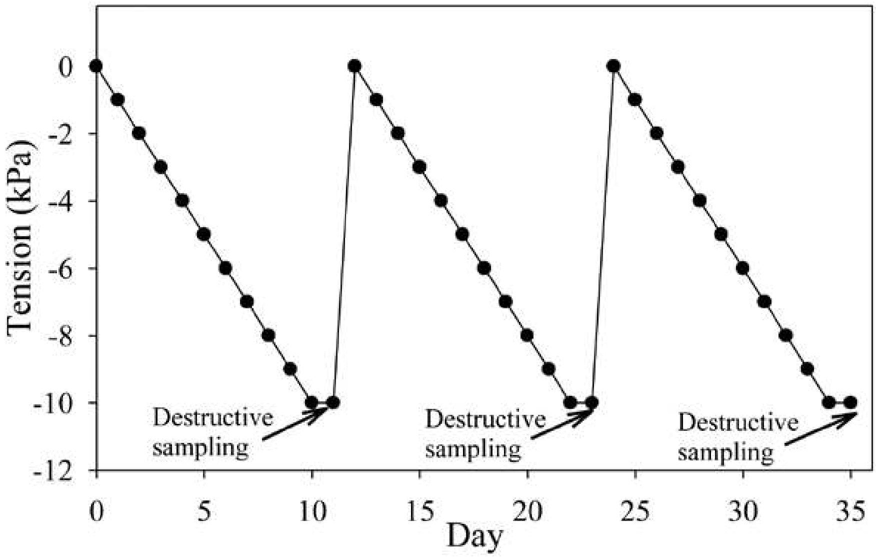


**Figure 1. Schematic diagram showing the changes in soil water tension over the course of the experiment and the points at which destructive soil sampling took place.**

**Results and discussions**

The ratio of $N_2O$ to $N_2$ emitted during denitrification depends on factors such as soil pH, soil water status, $NO_3$-N concentration and C supply (Clough *et al.* 2004). During the 1st drainage cycle nitrification of ammonium (from urine) was limited by restricted $O_2$ diffusion through the cores at high water contents (mean WFPS > 80%) and high DOC levels simulating microbial $O_2$ consumption (with corresponding high $CO_2$ emissions – data not shown) (Fig. 2a) hence, the initial supply of $NO_3$-N for denitrification was low (< 25 µg $NO_3$-N $g^{-1}$) and what little $N_2O$ was produced was further reduced to $N_2$ (Fig. 3, cycle 1).

At the start of the 2nd drainage cycle soil $NO_3$-N concentrations had increased slightly (Fig. 4, day 12) but $N_2$ was still the predominant gaseous N product with similar amounts being emitted from all compaction treatments (Fig. 3, cycle 2). This was likely to be due to limited $NO_3$-N supply, high soil pH and high DOC concentrations (Fig. 2, day 12).

By the 3rd drainage cycle $NO_3$-N concentrations increased further and were no longer limiting denitrification (Fig. 4, day 24) with $N_2O$ the predominant emission product from 0 and 220 kPa compaction treatments. The 400 kPa treatment had similar fluxes of both $N_2O$ and $N_2$ (Fig. 3, cycle 3) due to comparatively higher pH & DOC levels (Fig. 2, day 24) and higher water status at any given tension which favoured $N_2$ emissions. This resulted in significantly more cumulative $N_2$ being emitted from the 400kPa treatment by the end of the third cycle.

Relationships between soil water metrics (VWC, WFPS, AFP, MP and GDC diffusivity) and N gas fluxes ($N_2O$ and total N flux ($N_2O + N_2$)) were analysed for cycle 3 data only. Log transformed flux data was plotted against the soil water metrics and thresholds for anaerobic N gas production were investigated by determining the turning points on the resulting non-linear curves using Loess 3rd order models (Fig. 5). The 0 and 220 kPa compaction treatments displayed turning-points where N dynamics changed from being denitrification dominated to nitrification dominated, the 400 kPa compaction treatment did not reach a turning point and was excluded from this analysis. VWC was the preferred soil water metric as it explained more of the variability (57.5%) in $N_2O$ flux than WFPS (54.9%); $R^2$ values from Loess 3rd order models. More of the variability was explained in total N flux, with WFPS the best predictor on 67.3% (VWC 65.8%). Turning points in VWC for $N_2O$ and total N fluxes were identical for the 0 and 220 kPa compaction treatments at 54.4%. In contrast, turning points for WFPS decreased with increasing compaction with respective values for the 0 and 220 kPa compact treatments of 81.7 and 80.4%, however these differences were not significant.

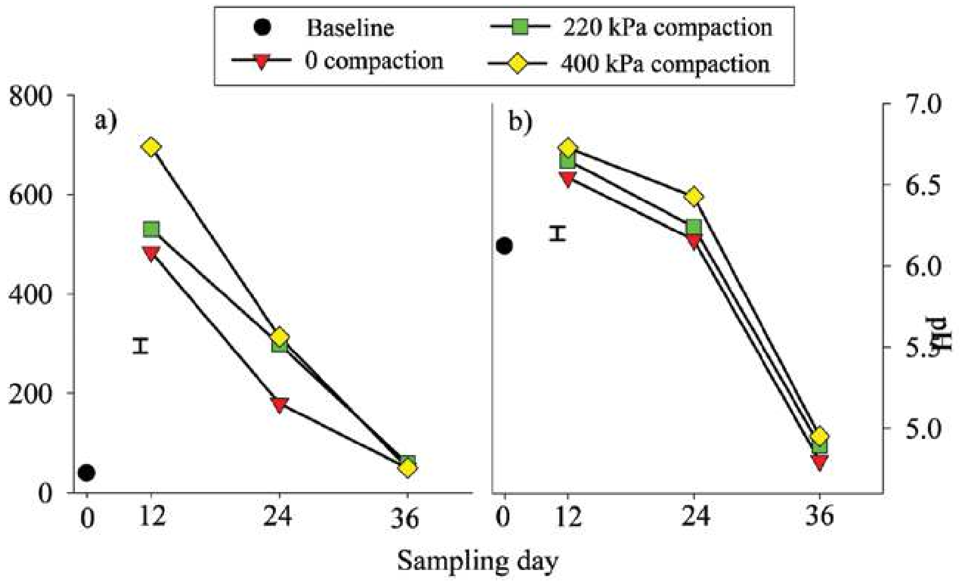


**Figure 2. The effect of sampling time and compaction treatment on a) dissolved organic carbon, and b) pH in urine treated soils. The bars represent the average 5% LSD with 50 df.**

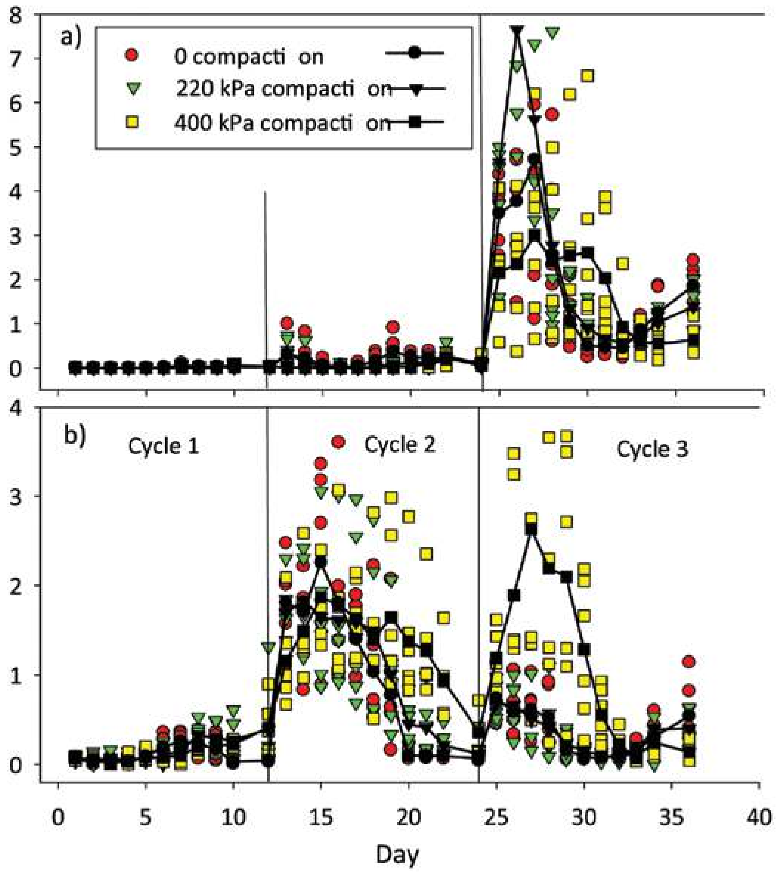


**Figure 3. The effect of drainage cycle and compaction treatment on flux rates of a) $N_2O$ and b) $N_2$ from urine treated soils. Lines are mean values (n = 6). Note the different scales on y-axis.**

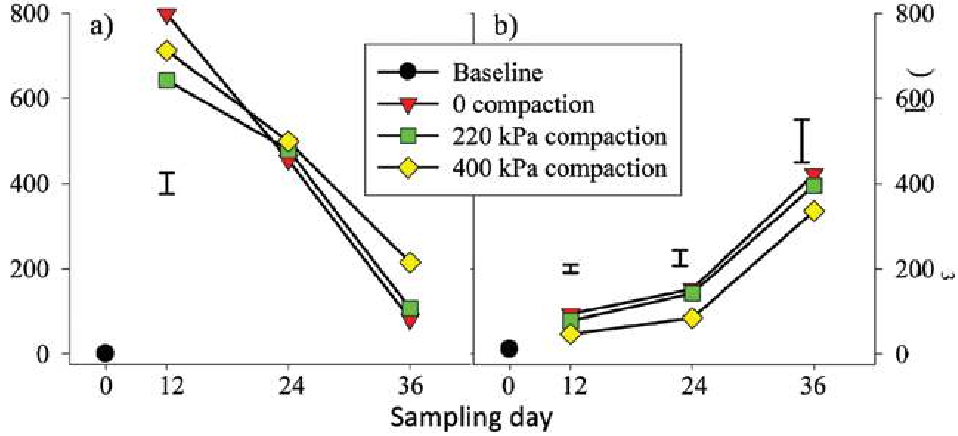


**Figure 4. The effect of sampling time and compaction on a) ammonium and b) nitrate levels in urine treated soils. The bar in graph a) represents the 5% LSD with 50 df, the bars in graph b) are 95% confidence intervals.**

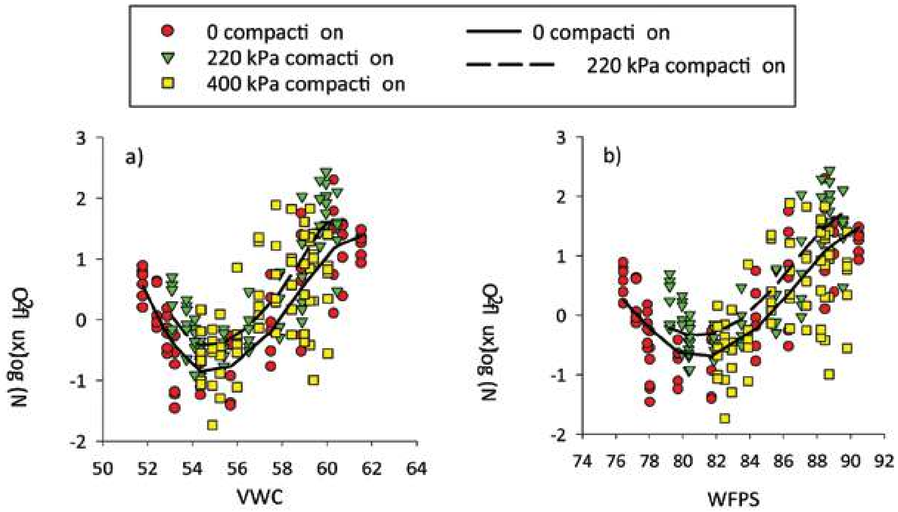


**Figure 5. Fitted 3rd order Loess models of log $N_2O$ flux against a) VWC and b) WFPS for the 0 and 220 kPa compaction treatments. Data limited to emissions measured during the third drainage cycle.**

## Conclusions

Compaction changed the drainage and aeration characteristics of urine amended soils and these changes influenced nitrification rates, the timing of emissions and ratio of N gas products over a series of drainage cycles. Our findings provide some support to the suggestion by Farquharson and Baldock (2008) that the volume fraction of water (VWC) be considered over WFPS where estimates of $N_2O$ emissions are being calculated across soils with varying bulk density.

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Farquharson R, Baldock J (2008) Concepts in modelling N 2O emissions from land use. *Plant and Soil* **309**(1/2), 147-167.

# Influence of scale on SWAT model simulation for streamflow in a non-conservative watershed

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The Soil and Water Assessment Tool (SWAT) was applied to 672 km$^2$ Shibetsu river watershed (SRW), eastern Hokkaido, Japan and its three sub-watersheds, 71 km$^2$ FW, 22 km$^2$ AFW and 10 km$^2$ AW to study the scale effect of SWAT model simulation for streamflow (Q). Streamflow of the studied location was measured for six years (2003 to 2008). Annul external water (EXT) was estimated from water balance calculation, monthly EXT was assumed as a constant proportion (annual ration of EXT/Q) of observed monthly streamflow and this assumption was validated by uncalibrated SWAT model. Internal stream discharge was calculated by subtracting EXT from observed streamflow for calibration and results comparison, then sensitive parameters differentiated by land use and soil type were calibrated and validated at SRW scale with simulation outputs for sub-watersheds FW, AFW and AW. Model performance was analyzed by computing the coefficient of determination ($R^2$) and Nash–Sutcliffe efficiency ($E_{NS}$) for the whole period. Results indicated that SWAT model performed monthly and annual streamflow reasonably well at large spatial scales of SRW and well down scale predicted annual streamflow of sub – basins FW, AFW and AW, however, with lower performance for monthly streamflow at FW, AFW and poor performance at AW. Lower performance can be explained by the uncertainty of monthly EXT assumption and its consequently compensation effects of biased assumption during calibration process, besides highly variable underground water fluxes exchange between AW and other sub-basins was the main reason for the poor performance.

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# Real-time tracking of the soil water balance for precision irrigation scheduling

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### Abstract

Real-time tracking of volumetric soil moisture and matric potential, using wireless soil moisture sensor networks, has been used to directly measure daily soil moisture deficit in spatially defined management zones under a 750-m centre pivot, modified with individual sprinkler control for precision irrigation. The sensors are optimally positioned into the crop using an electromagnetic (EM) survey and terrain attributes derived from the digital elevation map. At this site, the soil texture is uniform sand, but soil profiles are variably influenced by a high water table, with soil moisture typically varying by 0.35 $m^3 m^{-3}$ at any one time in the 68.5 ha area under the pivot. Irrigation timing, amount and placement is varied according to the EM soil management classes to maintain readily available water in the soil profile within the optimal range, withholding irrigation from wetter areas when droughty areas require irrigation, having reached their 100 kPa refill point. Results from two years of trials (2010/2011: maize; 2011/2012: wheat) showed water savings of 14% and 36% and water use efficiency estimates of 17.8 cf. 16.5 and 18.4 cf. 12.3 kg DM/mm for variable rate irrigation (VRI) compared with uniform rate irrigation (URI) respectively, for 2010/2011 and 2011/2012. The greater water savings in the second year reflect more rain during the irrigation season resulting in one management class requiring no irrigation all season.

### Key Words

Soil moisture, wireless sensor networks, EM, terrain attributes

### Introduction

Practitioner decisions to invest in precision irrigation technologies are being driven by the need to improve irrigation water-use efficiency, because water allocations are often restricted and there are increasing societal and regulatory demands for best practice management. Precision irrigation requires (i) an engineering solution to modify or develop an irrigation system with precise control of irrigation amount and placement, e.g. individual sprinkler control (Bradbury 2012), and (ii) a spatial decision support tool for precise irrigation scheduling (Green *et al*. 2006; De Jonge *et al*. 2007). The engineering solution can be put to best use when a spatial decision support tool is available for automated control of the precision irrigation system (Kim and Evans 2009).

Remote and proximal sensing technologies provide the opportunity to map soil variability, define critical management classes, monitor soil moisture status within each class, and send the information through land line or broadband internet or 3G cellular network connections to remote machine users (Hedley *et al*. 2012). High resolution soil moisture status maps derived by soil water balance modelling provide useful information for precision management (e.g. Hedley and Yule 2009). However, site specific soil moisture monitoring in management classes provides a direct spatially explicit measurement of soil moisture deficit for irrigation timing, and is therefore preferable. This direct measurement of soil moisture status precludes the need to estimate evapotranspiration, which requires many inputs not easily obtainable, including relative humidity, solar radiation and crop stage. Modelling approaches can be inaccurate at the site specific scale because they do not fully account for natural variability.

Irrigation control systems based on wireless sensor networks (WSNs) and real-time soil moisture data are potential solutions to optimise precision water management by remotely accessing in-field soil water conditions and then site-specifically controlling irrigation sprinklers (Kim and Evans 2009; Hedley and Yule 2009). Soil moisture data can be collected using an autonomous WSN positioned into the field, which then communicates with another autonomous wireless sensor network system controlling individual sprinklers on the irrigation hardware. Once the field WSN is activated, the most efficient communication path to relay data from each node to the base station is established. Data are processed at the base station, which also acts as a database and web server. This paper presents our progress in developing a spatial decision support tool for automated control of a precision irrigation system.

**Methods**

Two years of trials (2010/2011: maize; 2011/2012: wheat) were conducted in a 68.5 ha arable field irrigated by a 750-m centre pivot, modified for precision irrigation (Bradbury 2012). The area was mapped with an on-the-go EM mapping (Adamchuk *et al.* 2004) system, consisting of Geonics EM31 and EM38Mk2 sensors combined with RTK-DGPS and dataloggers mounted on an all-terrain vehicle. EM, elevation and yield data were kriged on a five-metre grid in the R 2.13.1 statistical environment (R Development Core team 2011) to produce the soil EM and digital elevation (DEM) maps. The SAGA Wetness Index (SWI) was derived from the DEM using SAGA software (Olaya and Conrad 2009). Yield data were obtained for the spatial data analysis, and interpolated onto the same grid. Yield closely mirrors the EM map; both reflected the contrasting soil moisture pattern across our study area, where the uniformly textured sand soils are variably influenced by a high and fluctuating water table.

The spatial data analysis processed these four datalayers (EM31, EM38, SWI, yield) onto a common interpolation grid, and the K-means clustering algorithm was used to derive the management class map (k=3). Replicated intact soil cores (n=3) were collected from each class for soil moisture release characterisation.

A wireless soil moisture sensor network (WSN) was then installed into the management classes, with three nodes, with sensors attached, positioned into each class. Watermark matric potential sensors (effective range between 0 and 200 kPa) were installed at 15 cm, and Decagon SM300 volumetric soil moisture sensors at 15 cm and 50 cm in each of the nine node positions. Soil moisture data were radioed every 15 minutes to the base station at the pivot. Data were remotely accessed, manipulated, and viewed in a web browser via a 3G cellular connection. The WSN enables a direct measurement of soil moisture status providing the precise information necessary for optimal control of the precision irrigation system.

Trials were conducted over 2 years to compare (i) uniform irrigation (URI) to the entire field when the most droughty soil class required irrigation with (ii) variable rate irrigation (VRI), which tailors irrigation scheduling to the specific refill point (at 100 kPa) for each management class. Yield was assessed in the trial plots to determine irrigation water use efficiency.

**Results and Discussion**

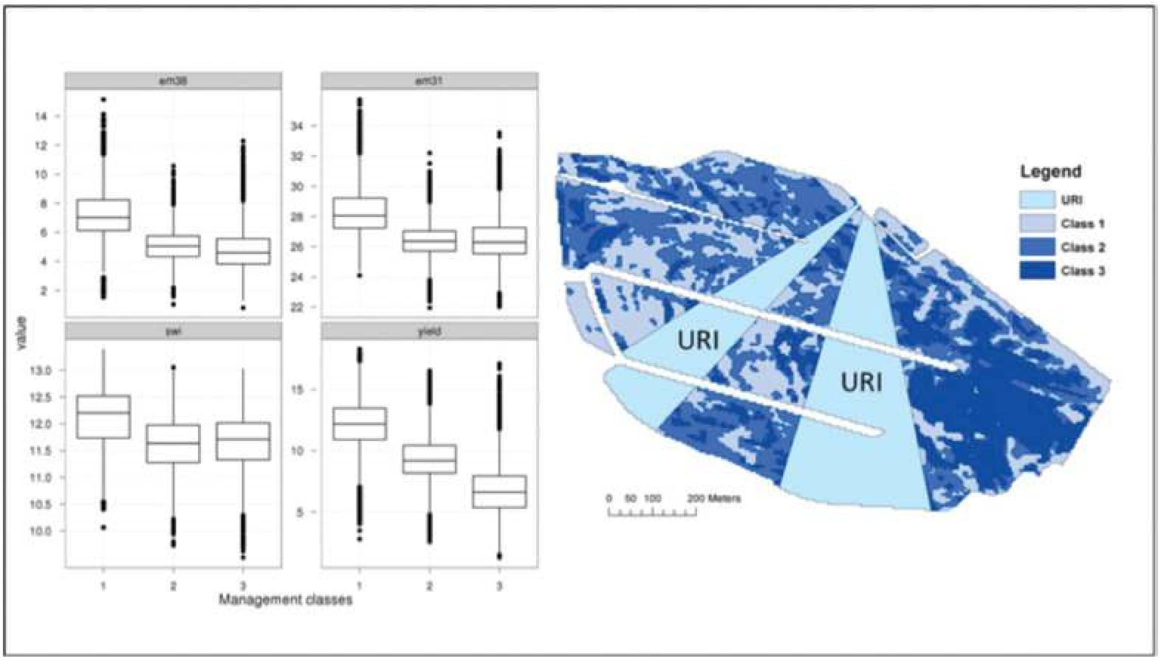


**Figure 1. Spatial data analysis of covariate datalayers was used to define three management classes. Irrigation was varied to these three classes, and applied uniformly to the Uniform Rate Irrigation (URI) plots.**

Figure 1 shows the clustering of the covariates into management classes. Ground-truthing of these classes confirmed that they accurately reflect management issues encountered at this site, including the fact that Class 3 soils remain wetter than other areas during spring because of the high water table in lower lying areas, and therefore require less irrigation.

It is difficult to account for the effect of sub-irrigation from ground water in a soil water balance modelling approach (De Jonge et al., 2007), so instead we used our WSN data to assess soil moisture deficit directly. Figure 2 shows how the wireless soil moisture sensors radio data to a base station, serviced by a standard web browser, which can be accessed from remote internet PCs.

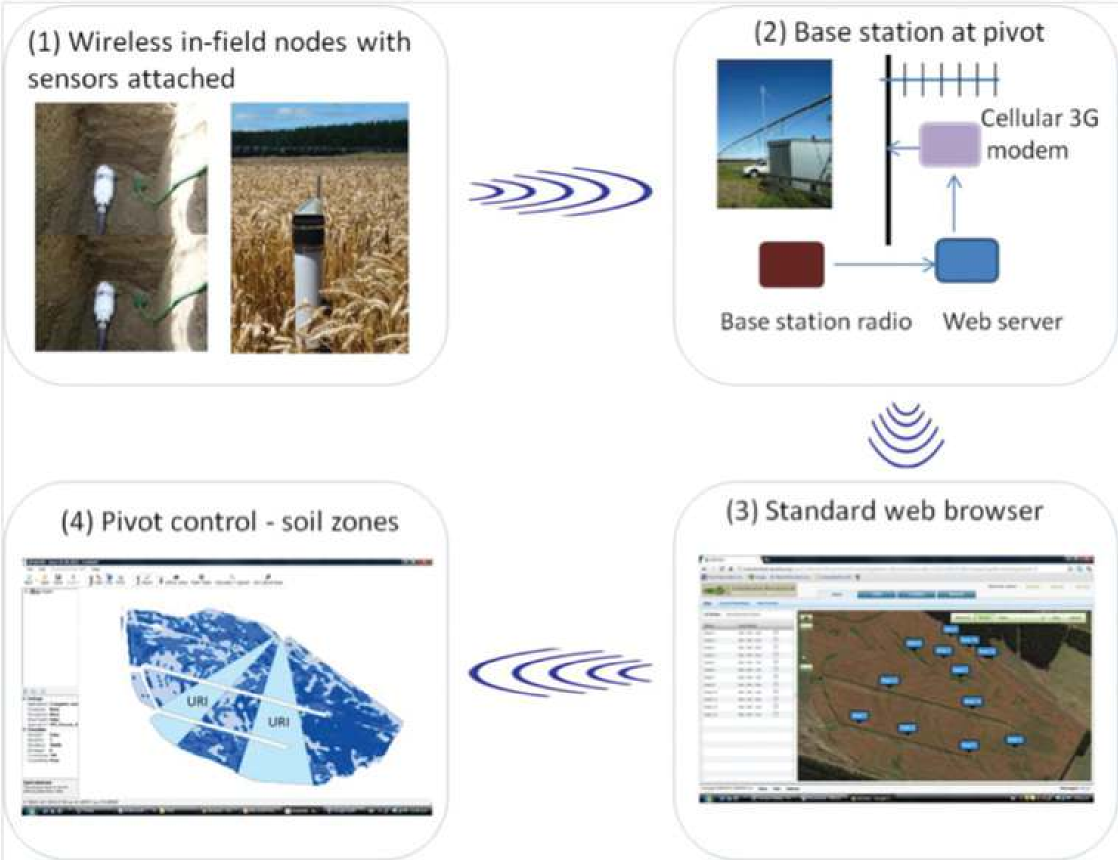


**Figure 2. A wireless soil moisture sensor network was installed into the field for real-time soil moisture monitoring within each management class.**

Soil moisture data for the 2011/2012 irrigation season is shown in Figure 3 for the three management classes. It shows the highly contrasting moisture conditions of the three classes. Class 3 soils were wetter than field capacity for the entire irrigation season, and irrigation was reduced to 20% compared with Class 1 which was within the operational constraint of the pump.

Overall water savings were 36%, compared with 14% in the previous much drier irrigation season (calculated for the same management class areas for both years). Irrigation was withheld when soil moisture was above field capacity (Fig. 3) and then scheduled to maintain soil moisture above the refill line to ensure no yield reduction. Water use efficiency (WUE) estimated as kg dry matter per millimetre 'irrigation plus rainfall' was 17.8 cf. 16.5 and 18.4 cf. 12.3 kg DM/mm for variable rate irrigation (VRI) compared with uniform rate irrigation (URI) respectively, for 2010/2011 and 2011/2012. These are not significantly different values. Daily water loss is also determined from the WSN soil moisture data, received every 15 minutes, which assists irrigation scheduling forecasts, and this direct measure is therefore an improvement on plant stress decision support tools for irrigation scheduling.

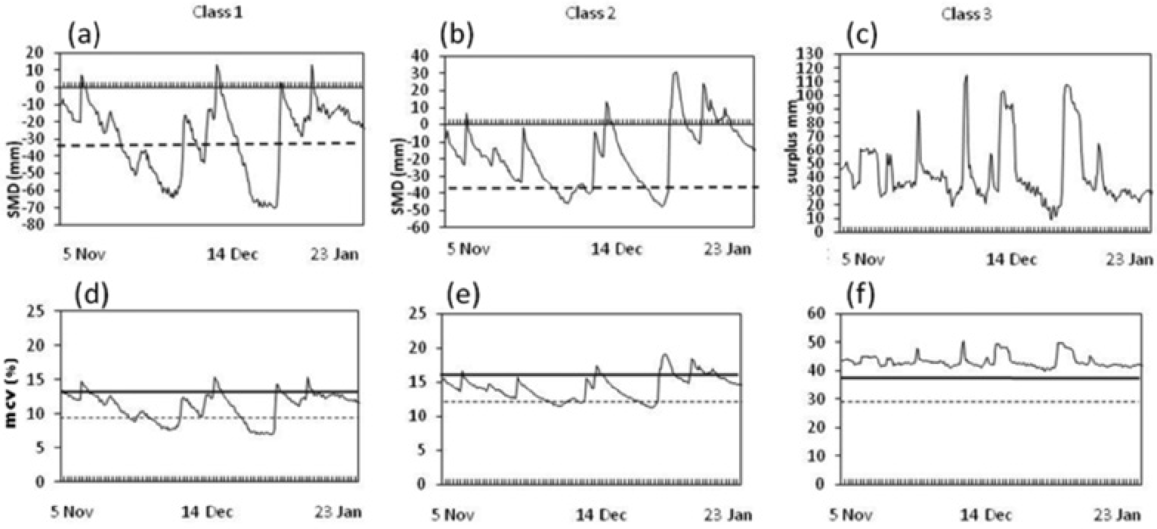


**Figure 3. *In situ* monitoring of daily soil moisture deficit (SMD, mm) for the 2011/2012 irrigation season for (a) Class 1, (b) Class 2, (c) Class 3; and volumetric soil moisture (mcv, %) in the profile to 20 cm for (d) Class 1, (e) Class 2 and (f) Class 3. In all graphs the dashed line is the refill line (100 kPa). In (d), (e) and (f) the solid line is field capacity. The tick marks on the horizontal axis represent days.**

## Conclusions

Our web-enabled, spatially-targeted WSN soil moisture monitoring directly measures daily soil moisture deficit, using volumetric soil moisture and matric potential data, which are then used to manage and vary irrigation to spatially defined management classes. This provides an accurate direct method for precision irrigation scheduling, and aims to refine existing modelling and monitoring approaches.

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# Combining EM mapping with VNIR spectroscopy for farm-scale soil carbon accounting

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On-the-go and on-site proximal sensing technologies are now available for characterising soil variability at a very fine scale. In this study, these technologies are combined to refine traditional methods of assessing soil carbon stocks and stock changes within the landscape, taking into account its intrinsic spatial variability. Visible near infra-red spectroscopy (VNIR) was used as a non-destructive and cost-efficient field method for estimating soil carbon stocks in a 68.5 hectare arable field. Soil carbon measurements (to 0.3 m) at one hundred positions have been modelled and mapped using electromagnetic (EM) survey data to develop a total soil organic carbon (SOC) mapping method.

Soil carbon was estimated by (1) laboratory analysis and (2) chemometric processing of the VNIR soil spectra. To estimate the number of physical samples needed to provide an accurate calibration set for the chemometric processing of VNIR spectra, model performance was repetitively assessed using between 10 and 100% of soil analyses for the calibration set. Our results indicate that VNIR could accurately predict SOC using only 40% of the soil samples as a calibration set.

*Mean* estimations over 10 simulations of the total soil carbon present to 0.3 m depth in this paddock is 3476.2 T for Method 1 and 3555.73 T for Method 2. Method 2 used 60 % less soil carbon laboratory analyses, and only differs from the Method 1 result by 2.29 %.

# Terrons as a planning and management tool for wine production in the Lower Hunter Valley

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This study developed terrons for the Lower Hunter Valley wine-growing region. Terrons, combined soil and landscape classes, are a quantitative and practical approach to developing terroirs, an important land classification for winegrowers. From a survey of 1400 data points covering some 22,077 ha, eight individual soil attributes were identified; namely, three principal components related to the Australian soil classification system, pH data from the subsoil and at 50cm, a drainage index, readily available water and the presence of $CaCO_3$ bearing rock in the soil profile. These data were combined with the following landscape attributes derived from a 25 DEM:- elevation, terrain wetness index, slope, multi resolution valley bottom flatness index, solar insolation and curvature, to create 12 distinct terron classes. The resulting summaries can be used together to determine soil suitability for various crops, and the metrics provided, such as drainage and available water can assist with the planning of irrigation and fertiliser regimes. Furthermore, in the case of wine and other luxury products, terrons themselves can be used as a marketing tool by linking descriptors of the wines e.g. varieties, quality, personality to particular attributes of the soils and landscape from which they come from.

# Improving access and use of national soil data assets

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The Australian Collaborative Land Evaluation Program (ACLEP) supports a vision that "natural resource management in Australia is underpinned by appropriate soil and land resource information and knowledge to ensure sustainable economic and environmental systems".

Recent ACLEP project activity has focused on a soil data user needs analysis, development of standard national soil data products, development of national soil information models and data specifications, provision of soil data as web services and demonstration of on-line access and use of soil data through a mobile device.

These ACLEP approaches provide significant opportunities for improving the Australian soil information infrastructure of the future. This paper examines the needs and opportunities for linking improved national soil data governance, open data sharing, new fine resolution data products and application development to support better policy and land management decisions.

# GlobalSoilMap's Oceania Node: Progress towards a new form of soil map

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*GlobalSoilMap.net* is the international soil science community's response to the need for soil information aimed effectively at the challenges of food security, environmental degradation and climate change. The Oceania Node of *GlobalSoilMap.net* was launched on Feb 7th 2011in Bogor, Indonesia. The Node developed over 2010 - 2011 through a process of workshops and engagement to build involvement in the node, explore approaches in pilot projects, develop an understanding of the information management challenges, develop and deliver training programs and propose a set of principles for the Node's operation. The Node members agreed on a Memorandum of Understanding and are building capacity in soil information and in related skills. The Node is now in its second year and in many places has moved into the building phase. Projects have been established in Australia, New Zealand and Indonesia and an initial project scoped for the Pacific.

Node members have collaborated on innovative methods for information modelling and information services, the development of effective approaches to uncertainty measurement and communication, shared approaches to the disaggregation of traditional polygon mapping, broadscale 'SCORPAN' estimation of soil functional attributes, a new approach to mapping and displaying soil attribute depth relationships and, more generally, in the design of national and transnational approaches to producing a fine-scale grid of soil functional attributes. The new global specifications for GlobalSoilMap owe much to the work within the node.

The end result will be a new comprehensive map covering the lands of Indonesia, Australia, New Zealand and Pacific at a constant fine resolution (100m) with the key soil attributes needed to model and understand the soil's underpinning role in food production, environmental management and climate change adaptation and mitigation. The first year of operation has produced examples across the node that illustrate the potential power of this project and a larger group of highly skilled spatial soil scientists.

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# Digital soil mapping of compositional particle-size fractions (PSFs) using proximal and remotely sensed ancillary data

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The soil particle-size fractions (PSFs) are one of the most important attributes to influence soil physical (e.g. soil hydraulic properties) and chemical (e.g. cation exchange) processes. There is an increasing need, therefore, for high-resolution digital prediction of PSFs in order to improve our ability to manage agricultural land. Consequently, use of ancillary data to make cheaper high resolution predictions of soil properties is becoming popular. This approach is known as digital soil mapping (DSM). However, most commonly employed techniques (e.g. multiple linear regression-MLR) do not consider the special requirements of a regionalised composition, namely PSF; (i) should be non-negative, (ii) should sum to a constant at each location, and (iii) estimation should be constrained to produce an unbiased estimation, in order to avoid false interpretation. Previous studies have shown that the use of the additive log-ratio transformation (ALR) is an appropriate technique to meet the requirements of a composition. In this study we investigate the use of ancillary data (i.e. electromagnetic (EM), gamma-ray spectrometry, Landsat TM and digital elevation model (DEM) to predict soil PSF using both MLR and generalised additive models (GAM) in a standard form and with an ALR transformation applied to the optimal method (GAM-ALR). The results show that the use of ancillary data improved prediction precision by around 30% for clay, 30% for sand and 7% for silt for all techniques (MLR, GAM and GAM-ALR) when compared to ordinary kriging (OK). However, the ALR technique had the advantage of adhering to the special requirements of a composition, with all predicted values non-negative and PSFs summing to unity at each prediction point and giving more accurate textural prediction.

# Progress in national ground cover monitoring for Australian agriculture

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## Abstract

A nationally agreed basis for monitoring ground cover using satellite imagery is being implemented for Australia. This will assist with the assessment of Australia's soil resources and agricultural productivity at national, state and regional scales. A MODIS-based vegetation fractional cover product has been selected for this purpose. It is being validated and improved using a national network of field sites. The results of the initial validation and improvement in fractional cover estimates are presented. Fractional cover is separated into photosynthetic and non-photosynthetic vegetation and bare soil components. The current focus is on establishing the accuracy of the product for agricultural land uses, in particular grazing in the rangelands, and for use in erosion models. Summary statistics are produced from the fractional cover product to report ground cover levels and trends over time. A national example is given using the seasonal mean ground cover levels for the 12 month period from autumn 2011. Further developments of the fractional cover product and associated metrics are outlined.

## Key words

ground cover, remote sensing, erosion, agriculture

## Introduction

Ground cover data has been identified as critical for assessment of environmental targets related to soil erosion and land management by natural resource management agencies (McKenzie and Dixon 2006). For Australia to manage its current soil resources, develop policies to help land managers adjust to a changing climate and demonstrate the effectiveness of current land management policies and investments, a clearly defined process is required for monitoring and reporting on ground cover at regional, state and national scales (Leys *et al.* 2009).

Ground cover can be monitored consistently over months, seasons and years for large spatial extents, such as Australia's rangelands and broadacre cropping areas, using remote sensing. From a remote sensing perspective, ground cover is the fractional cover of the non-woody vegetation and litter near the soil surface. Fractional cover comprises living (photosynthetic) vegetation, senescent or dead (non-photosynthetic) vegetation and bare soil or rock.

A collaborative partnership (DAFF 2010) is implementing a nationally agreed, reliable and cost-effective project for mapping ground cover using satellite imagery to provide regular updates of ground cover conditions across Australia. This will assist the monitoring and reporting of ground cover at national, state and regional scales to assess soil resources and agricultural productivity.

This paper describes the progress in delivering a nationally validated ground cover product for Australia.

## Methods

The Moderate Resolution Imaging Spectroradiometer (MODIS)-based vegetation fractional cover product of Guerschman *et al.* (2009) was selected for monitoring ground cover across Australia (Stewart *et al.* 2011). The product is available nationally every 8 days as 16-day composites from March 2000 onwards at 500 m resolution. It separates the photosynthetic and non-photosynthetic vegetation components from the bare soil —important for improving erosion model outputs. More extensive validation of the product is required.

A stratified sampling approach has been adopted to prioritise site selection (Malthus *et al.* in prep) to validate the fractional cover product. Grazing (predominately in the rangelands) and broadacre cropping areas were selected considering factors such as different ground cover levels and soil colours. The sites selected will contribute to a network of sites that are sensor independent (that is suitable for use for fractional cover products derived from satellites in addition to MODIS). Each site has been described and fractional cover measured on the ground according to national standards (Muir *et al.* 2011). These standards have been adapted from the Queensland Statewide Landcover and Tree Study with site description attributes conforming to the *Australian soil and land survey field handbook* (NCST 2009). Site description and transect measurements were recorded on paper and/or electronic forms and compiled into a national database (Rickards *et al.* 2012).

Summary statistics (metrics) that describe the maximum, minimum, mean and variation for each of the three fractional components (photosynthetic, non-photosynthetic and bare soil) are produced for each month, season, year and the whole archive.

Data products are available at the National Computational Infrastructure (NCI 2012).

**Results**

The algorithm for the MODIS-based vegetation fractional cover product (Guerschman *et al.* 2009; version 2.1) was originally developed for the northern savannas of Australia and then applied nationally. Using data available for 359 sites (567 observations) from Queensland, New South Wales and South Australia, version 2.1 was validated and the algorithm recalibrated to produce version 2.2 (Guerschman *et al.* 2012). The recalibration eliminated the bias in underestimation of non-photosynthetic vegetation and overestimation of bare soil of version 2.1. It also reduced the root mean square error (RMSE) in the estimates of the three fractions. The RMSE of version 2.2 is 14.7 % for photosynthetic vegetation, 20.6 % for non-photosynthetic vegetation and 17 % for bare soil. A lower RMSE indicates less difference between the modelled fractional cover data and the data collected at field sites.

ABARES (2012a) have produced metrics for version 2.2 to summarise the MODIS-based vegetation fractional cover product. These are used for monitoring and reporting on ground cover levels and trends.

Figure 1 shows seasonal change in mean ground cover levels, for each 500 m pixel across Australia, under non-forested agricultural land uses from autumn 2011 to summer 2012. Conservation and indigenous protected areas and other non agricultural land uses were excluded using the *Catchment scale land use of Australia–Update March 2010* (ABARES 2010). Areas with more than 20 % tree cover were excluded based on *Forests of Australia 2008* (ABARES 2008). Ground cover is the sum of the photosynthetic and non-photosynthetic vegetation fractions covering the soil. The mean ground cover for each season was then classified into four classes based on soil erosion control targets. These are: 0 to 30% ground cover, as areas where ground cover levels are 30 % or less are at high risk of soil loss; 30 to 50 %; 50 to 70 %, as targets of 50 % are promoted to prevent wind erosion and 70 % (50% for Western Australia) to prevent water erosion; and 70 to 100 %, as these areas are at very low risk (Leys *et al.* 2009). The high ground cover shown across most of Australia in 2011 is due to very high rainfall during 2010 and 2011 (BOM 2012).

**Discussion**

Further improvement of the algorithm is recommended by Guerschman *et al.* (2012) to achieve a RMSE of lower than 10-15 %. For erosion modelling, a precision of +/- 15 % for the bare soil component of the fractional cover product is required (Malthus *et al.* in prep.). Possible approaches to further improve the algorithm include:

- having variable endmembers, as defined in Guerschman *et al.* (2009), depending on environmental conditions
- incorporating more bands or indices to increase the ability of the model to estimate fractional cover
- increasing the number of field visits where spectral measurements are taken.

Comprehensive collection of field data representing vegetation conditions, preferably at internally homogeneous sites and with coincident Landsat imagery, will help validate the extent that these approaches improve the model. Also to be resolved are the effect of soil colour and soil moisture on model performance.

A minimum of 495 additional sites will be used in the validation and improvement of the algorithm in 2013. Data collection from these sites is funded under the current project and collected by collaborating state and territory partners. Extra site data for the rangelands should also be available for this exercise through the Terrestrial Ecosystem Research Network's AusPlots. Methods of data collection have been aligned to ensure suitability for use with remote sensing. Other state-based activities in Queensland and New South Wales use the same transect sampling methods as described in Muir *et al.* (2011) and will also be sourced. These additional sites will improve the spatial accuracy of the MODIS-based vegetation fractional cover. Selected sites will need to be revisited to assess the temporal accuracy of the product.

ABARES is producing a report providing products for monitoring and reporting of ground cover levels and trends. It is anticipated that these products will be available via the Monitor, an on-line reporting tool (ABARES 2012b). Future developments will seek to automate the production and delivery of the MODIS-based vegetation fractional cover product and its associated metrics to enable real-time decision making.

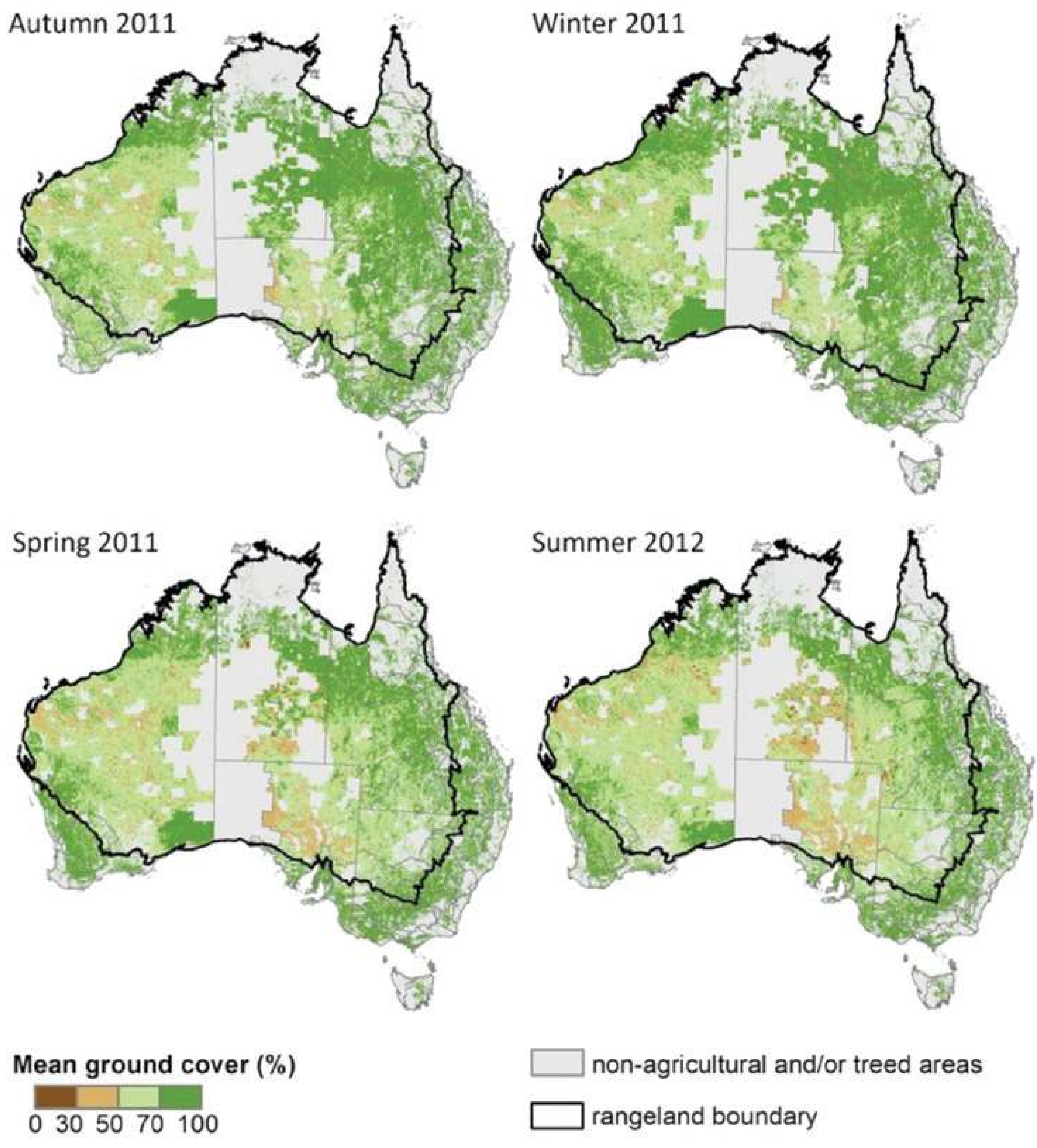


**Figure 1. Seasonal ground cover levels for agricultural areas derived from the MODIS-based vegetation fractional cover metrics (version 2.2).**


## Acknowledgements

The Department of Agriculture, Fisheries and Forestry is supporting this activity. Collaborating organisations are: Commonwealth Scientific and Industrial Research Organisation (CSIRO) through the Sustainable Agriculture Flagship, the Terrestrial Ecosystem Research Network, the New South Wales Office of Environment and Heritage, the Northern Territory Department of Natural Resources, Environment, the Arts and Sport, the Queensland Department of Science, Information Technology, Innovation and the Arts, the South Australian Department of Environment, Water and Natural Resources, the Tasmanian Department of Primary Industries, Parks, Water and Environment, the Victorian Department of Primary Industries, and the Western Australian Department of Agriculture and Food.

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# Using digital soil mapping for enterprise suitability assessment in support of Tasmanian irrigation development

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## Abstract

Digital Soil Mapping(DSM) was used to model enterprise suitability in recently commissioned irrigation schemes in Tasmania, in support of Government agricultural expansion policy. The Wealth from Water pilot program commenced in 2010 within a 70,000 ha irrigation district in the Meander Valley and Midlands of Tasmania. The modelling requires comprehensive soil, climate and terrain parameters to rate the suitability of land for twenty enterprises, including alkaloid poppies; carrots; hazelnuts; barley; blueberries; pyrethrum and industrial hemp. Digital soil mapping was used to produce soil information for pH, EC, clay and stone content, drainage, and depth to sodic and/ or impeding layer based on sampled soil cores and explanatory spatial data. Sites were located using a stratified random sampling design, with explanatory covariate datasets including a digital elevation model and derivatives, gamma radiometrics, legacy soil mapping, surface geology, and satellite imagery. Individual soil properties were predicted using Mid Infrared (MIR) spectroscopy analyses. Temporal climatic grid inputs of frost risk, growing season, chill hours, and rainfall intensity were generated using field temperature sensors and explanatory terrain data, and extrapolated to historical long-term averages. The land suitability for each enterprise was determined by interrogating each soil and climate parameter with a series of suitability rules. The soil, climate and suitability datasetsare planned for public release in December 2012 through the Tasmanian Government web-based Spatial Portal, (the LIST, www.thelist.tas.gov.au).

## Introduction

The Wealth from Water Pilot Program commenced in November 2010 in support of irrigated agricultural expansion through land suitability assessment. It is a partnership between the Department of Primary Industries, Parks, Water and Environment (DPIPWE), the Department of Economic Development, Tourism and the Arts (DEDTA), the Tasmanian Institute of Agriculture (TIA), and The University of Sydney (through an Australian Research Council Linkage Project). The enterprise suitability process was piloted in two irrigation areas in Tasmania; the Meander Valley (43,000ha) and Midlands (27,000ha). Suitability rules using soil, climate and terrain parameters were developed by TIA for 20 different enterprises including alkaloid poppies, carrots, hazelnuts, barley, blueberries, pyrethrum and industrial hemp. Existing 1:100,000 soil mapping and available site data was not at the scale or quality to produce reliable soil property surfaces at the resolution required. A DSM approach was implemented, which can often produce superior continuous and quantitative soil property estimates to conventional soil maps, with statistical validation and associated model uncertainty provided (McBratney *et al.* 2003). An "Enterprise Suitability Model" was developed using these soil inputs together with generated climate surfaces for each individual enterprise.

## Material and Methods

The Tunbridge and Midlands pilot areas (Figure 1) were chosen to test the methodology on a wide range of soil and climatic parameters. These include Tertiary sediments of the Launceston Tertiary Basin (mainly Sodosols), Triassic sandstone sequences near the Oatlands township (Chromosols and Sodosols), Tertiary Basalt (Ferrosols) at Deloraine and complex alluvium at Meander (Hydrosols), (Spanswick and Kidd 2001; Spanswick and Zund 1999).

McBratney *et al*. (2003) introduced the "SCORPAN" approach for predictive soil mapping, which states

$$Soil\_{(property/\ class)}=f(S,C,O,R,P,A,N) \qquad (1)$$

where the prediction of a soil type or attribute is a function of soil data (S), organisms (O), relief (R), parent material (P), age (A)and spatial position (N). The Wealth from Water DSM methodology analysed soil sample cores to train and validate various modelling approaches using explanatory spatial datasets (covariates).These covariates (Table 1) were assembled and processed to a common 30m spatial resolution for the study area using SAGA GIS (System for Automated Geoscientific Analyses, http://www.saga-gis.org).Terrain derivatives (e.g. hill-shade, aspect, slope/mid-slope position, plan curvature, topographic wetness index and MrVBF (multi-resolution valley bottom flatness index) and products from satellite imagery (e.g. Normalised Difference Vegetation Indices and Fractional Vegetation Cover) were used in the DSM process. The existing 1:100,000 soil mapping was extrapolated into unmapped areas using a rule-based classifier. A ground-based gamma radiometric survey was undertaken by CSIRO Land and Water to complete partial existing Mineral Resources Tasmania (MRT)coverage in Meander West.

**Table 1: SCORPAN covariates used in the Wealth from Water DSM methodology**

| Dataset | Dataset |
|---|---|
| 1:100,000 soil maps (Meander, Oatlands) | SPOT 2009 multispectral satellite imagery |
| 1:100,000 land capability map (Meander) | RapidEye 2010 multispectral satellite imagery |
| Shuttle Radar DEM (Geosciences Australia, 2010) | LandSat 2010 multispectral satellite imagery |
| Gamma radiometrics (U, Th, K and TC), (MRT) | 1:50,000 land use (DPIPWE 2010) |
| 1:25,000 TASVEG vegetation (DPIPWE 2012) | 1:25,000, 1:250,000 geology (MRT) |

A "conditioned Latin hypercube" (cLHS) sample design was undertakenfor the Meander Valley Irrigation Area (MVIA) to maximise stratification of the full covariate distribution (Minasny and McBratney 2006). Of the 260 cores collected for an initial 20,000 ha district of the MVIA, 200 of these were sampled using this design. An additional 60 sites (30%) were sampled from fuzzy k-means clusters (derived from SCORPAN covariates) for validation assessment, where each cluster was used to stratify the covariates. In the remaining 50,000 ha of the Tunbridge and MVIA areas combined, both training (469 sites) and validation (201 sites) was undertaken using the fuzzy k-means sample design.This approach offered greater flexibility than the cLHS approach, as sites could be manually shifted within clusters where original locations were subject to access and sampling constraints. This maintained sampling stratification and maximised the chances of obtaining a good sample distribution across the full covariate feature space. All validation samples were generated independently of the sampling design to improve validation rigour (Brus *et al.* 2011).

Samples were collected to a depth of 1.5 m using a 50 mm diameter percussion soil corer, and sub-sampled by horizon. Cores and surrounding landscape position were described according to Australian Soil and Land Survey guidelines (National Committee on Soil and Terrain 2009). MIR spectroscopy was undertaken on all soil samples by CSIRO Land & Water and the DPIPWE to predict required soil properties. Twenty percent of scanned samples were selected and analysed for chemical and physical properties (pH, EC, exchangeable cations, N, P, K, organic carbon, and particle size distribution) at CSBP Laboratories in Western Australia to calibrate MIR spectra with soil properties. Depth splines (Malone *et al.* 2011) were fitted to all sampled profiles for each MiR-predicted soil property to allow depth-specific queries and creation of inputs for land suitability modelling. Spatial predictions of soil properties were generated using the MiR predictions and a range of DSM techniques and covariate data (Table 2).

**Table 2: Spatial soil prediction techniques and associated software**

| Spatial prediction technique | Software |
|---|---|
| Regression Trees (RT) | Cubist®/ See5®(Rulequest Research) |
| Artificial Neural Networks (ANN) | JMP®JMP, Version 9. SAS Institute Inc., Cary, NC, 1989-2010 |
| Regression Kriging (RK) | SAGA GIS(System for Automated Geoscientific Analyses, http://www.saga-gis.org) |
| Random Forests (RF) | R(R Foundation for Statistical Computing - http://www.R-project.org) |

Climate data was not available at a sufficient resolution or sample density to reliably inform the suitability mapping at 1:50,000 scale. RK was used to produce detailed climate surfaces using data recorded by 271 additional temperature sensors (0.4 sensors/km$^2$) and 6 new climate stations, along with a suite of terrain covariates.These predictions were then adjusted to long-term averages from historical Australian Bureau of Meteorology (BoM) data using multiple regression analyses between all BoM climate stations and digital elevation.Suitability rules were developed by the TIA for each enterprise from information on Tasmanian agricultural research trials, existing literature, and consultation with industry experts. Table 3 shows the predicted soil, climate and terrain parameters, and the suitability requirements for blueberries.

**Table 3: Suitability Parameter Ratings (blueberries)**

| Suitability Rating | Soil Depth | pH (0- 15cm) | Drainage Class | Stone % (0-15cm) | Slope % | % Frost Risk Years (Sep-Oct) | Mean Max Temp $^{o}$C (Oct-Mar) | Chill Hours |
|---|---|---|---|---|---|---|---|---|
| Well-Suited | >10cm | 5 – 5.7 | Well, Mod-Well | <20% | <12 | <10% | 15-26 | 800-1200 |
| Suited | >10cm | 4.5 – 5.0<br>5.7 – 6.7 | Imperfect | <20% | 12-20 | 10-30% | 15-26 | 700-800 |
| Moderately-Suited | >10cm | 6.7 – 7.5 | Imperfect | <20% | 12-20 | 30-50% | 26-30 | NA |
| Unsuited | <10cm | <4.5<br>>7.5 | Poor, V. Poor | >20% | >20 | >50% | >30 | <700, >1200 |

To assess the suitability ranking of each parameter for each 30 x 30 m pixel, ESRI Model Builder® was used to automate and apply a series of SQL queries based on the suitability rules and associated parameter ranges. A"most limiting factor" approach was used (Klingebiel and Montgomery 1961), which applies the lowest rating of any one parameter to the overall enterprise suitability rating for that pixel. The enterprise suitability model produces an output surface for each enterprise, along with the suitability rating for each individual parameter so that any soil or climate limitations can be identified.

### Results

Soil properties including pH, ECe (saturated extract), clay %, depth to sodic layer, depth to impeding layer, stone % and drainage class were spatially predicted using a variety of DSM techniques. For example, clay % (0-15 cm) was predicted and validated using RT, ANN and RK. Two hundred training sites were used for both training and validation (MIR scanning of validation sites was incomplete at the time of publication), with a stratified random hold-back of 60 training sites for validation. Remaining soil property predictions were tested using comparable techniques, which should improve during 2012 as more training and validation sites are sampled in adjacent areas.Table 4 shows the example comparison of different modelling methods for clay % estimates (0 to 15 cm) based on preliminary soil sampling and analyses.

**Table 4: Comparison of modelling methods for clay % (based on preliminary results).**

| Method | R-squared Training | R-Squared Validation | Validation RMSE |
|---|---|---|---|
| RK | 0.73 | 0.57 | 4.1 |
| RT | 0.61 | 0.43 | 4.4 |
| ANN | 0.43 | 0.32 | 5.7 |

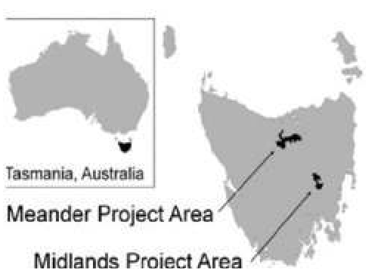


**Figure 1: Project Areas**

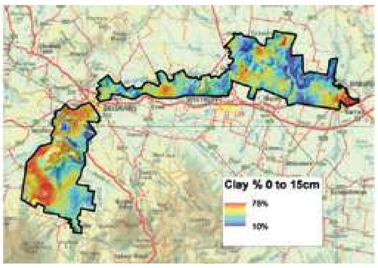


**Figure 2: Predicted Clay% (0 to 15cm)**

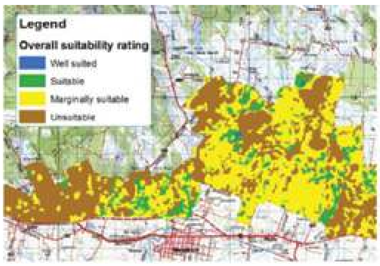


**Figure 3: Prototype suitability (blueberries)**

Prototype suitability maps were produced for each enterprise and uploaded to "The LIST" (an on-line spatial web portal, www.thelist.tas.gov.au).This web portal enables interrogation of all parameter suitability rankings, and identification of any limitations. Figure 3 shows a prototype suitability output for blueberries.

## Discussion

Initial prediction and validation results (e.g. clay % for 0 to 15cm) suggest that the available spatial covariates are satisfactory predictors of certain soil properties. The best preliminary predictions were achieved using Global Universal Kriging, a form of RK (Hengl *et al.* 2004). The RMSE of less than 5% was considered an acceptable degree of uncertainty, or predictive error. Soil spatial prediction methods implemented to date have worked satisfactorily for some soil properties and not for others. These methods will be further trialled for all properties, and those having superior validation used as final inputs to enterprise suitability models.

The enterprise suitability predictions using the "most limiting factor" approach is reliant upon the qualities and uncertainties of soil and climate predictions. For the example of blueberries (Figure 3), the greatest limitation to suitability in the modelled area was shown to be soil drainage, which spatially aligns with known flood plains, lower river terraces, and drainage depressions, consistent with documented major soil types of the area. Prediction uncertainties will compound for each input parameter prediction in the model and will be captured in the overall modelled suitability uncertainty or value range for each pixel. The enterprise suitability predictions and associated uncertainties should improve as soil sampling and analysis data for training and validation purposes becomes available.

## Conclusions

The Wealth from Water enterprise suitability assessment is one of the largest digital soil mapping projects undertaken in Australia, and is a good example of academic research moving into operational Government business. There are many components to such an assessment process, with the overall suitability outputs reliant on the quality of the predictions that they are based on. Preliminary outputs are positive, with satisfactory soil and climate predictions achieved, which should only improve as training and validation sampling is finalised during 2012. Many of the available covariate datasets appear to be good predictors of required soil properties, while modelled enterprise suitability products are aligning with expert landscape knowledge and land uses within the pilot area. The Wealth from Water project will assist Tasmanian irrigators to maximise returns on their irrigation investment through the generation of high resolution soil and climate data and their associated uncertainties by providing a quantitative measure of confidence in these outputs that is often missing from conventional soil mapping.

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# Space-time modelling of soil moisture

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**Abstract**

Space-time geostatistics provides a valuable insight into joint space-time variability of soil moisture, namely, to account for spatial and temporal variation of model residuals when a physical trend is available. Adding time to assessment models provides for a better understanding of soil moisture dynamics and, consequently, crop growth or even response to climate change. This paper presents a space-time model for soil moisture using the Tarrawarra dataset as a case study. The final aim is to illustrate the usefulness of this approach for identifying flaws in the physically-based trend model. In general terms, model results can also be used for designing monitoring schemes and to improve soil moisture mapping.

## Introduction

Soil moisture assessment is a key issue for reservoir management, early warning of droughts, irrigation scheduling, and crop yield forecasting. Moisture variability impacts many different processes acting at various scales ranging from the local agricultural field up to the global catchment scale. At the field-scale, soil moisture has an impact on the generation of runoff and erosion, plant growth and the chemical behaviour of fertilizers, which is important to agriculture and the environment (De Lannoy *et al.* 2006). At the catchment-scale, flood prediction and water-balance are influenced by the average soil moisture conditions. Monitoring soils moisture is the way to determine how much water is present, to prevent soils being either over- or under-irrigated. Besides the water use efficiency, excess soil moisture conditions above field capacity can result in excess drainage and leaching of nutrients (increased production costs, groundwater pollution and eutrophication of wetlands). On the other hand, deficient soil moisture places the plants under stress and leads to poorer crop yields (poor rooting, reduced nitrogen uptake and earlier maturity).

Given the importance of soil moisture to our agricultural and natural ecosystems, accurate models to describe its variation in space and time are of paramount importance. In general, studies related to soil moisture have focused on the spatial domain where there are many observations in space for one time period or the temporal domain where the same site is repeatedly measured. While this is useful, of more interest for management is the joint variation in space and time of soil moisture. This is due to its dynamic nature as it responds temporally to rainfall but this response is dependent on variation of soil properties through space.

Therefore an idealized model should consider space and time. The variation of soil moisture can be defined by equation 1:

where $s$ is the spatial location and $t$ is the time interval. The trend can be described by a model with a physical basis (e.g. bucket model) or a regression relationship with readily available covariates, e.g. rainfall and the topographic wetness index. The residual can be described by a space-time variogram which models the joint spatial and temporal variation of the residuals (Heuvelink and van Egmond 2010).

In the soil science literature, there are few studies of the joint space-time variation of soil moisture, and the few that do exist either focus on modelling soil moisture directly with space-time geostatistics or with a trend model. However the advantage of using a modelling approach as described in Equation 1 is that combining both together not only improves the predictions but perhaps of equal importance, it allows a deeper understanding of why our trend or physical-based model is giving poor predictions in certain time periods or at particular locations in space.

Therefore the aims of this research are to present a space-time model of soil moisture using the Tarrawarra dataset as a case study and illustrate the usefulness of this approach for identifying flaws in our physically-based trend model.

## Materials and methods

### *Description of study site and soil moisture dataset*

The case study dataset is the Tarrawarra Soil Moisture Dataset (Western and Grayson 1998; http://people.eng.unimelb.edu.au/aww/tarrawarra/tarrawarra.html). The Tarrawarra catchment (southern

Victoria, Australia) is an experimental site covering 10.8 ha that was extensively monitored for spatial and temporal soil moisture variation between September 1995 and March 1997. According to Western and Grayson 1998, the Tarrawarra catchment is representative of landscapes with relatively shallow soils of low to moderately high lateral permeability with an impending layer at depth, for which topography plays a significant role in routing water through the landscape, and for climates ranging from temperate to subhumid. Land uses consist of perennial improved pastures for dryland dairy cattle grazing. The Tarrawarra dataset consists of 13 sets of Time Domain Reflectometry (TDR) soil moisture surveys. All surveying work was carried out using GPS total stations (Western and Grayson 1998). Soil moisture surveys were performed using TDR and probes were inserted to a depth of 30 cm. The majority of the TDR surveys (11 out 13 dates) were collected on a 10 x 20 m grid with the 20 m axis orientated perpendicular to the main drainage direction. Of the remaining two surveys, the 10/11/96 TDR survey was collected on a 10 x 10 m grid with single probe and the 02/05/96 TDR survey was collected on a 10 x 20 m grid with four probes set at the edges of a 1.7 x 1.9 m rectangle. In this work we do not consider these as they are not collocated in space. The histogram of moisture values measured for all time sets and their statistical distribution per sampling period are presented in Figure 1. Soil moisture shows a clear variation between sampling campaigns, with higher moisture contents registered from July to September and lower values measured in February, March and November. Moisture was most variable in terms of the range in September (in 1995 and 1996).The smallest individual moisture values were measured in February and in November whereas the highest was in September 1996.

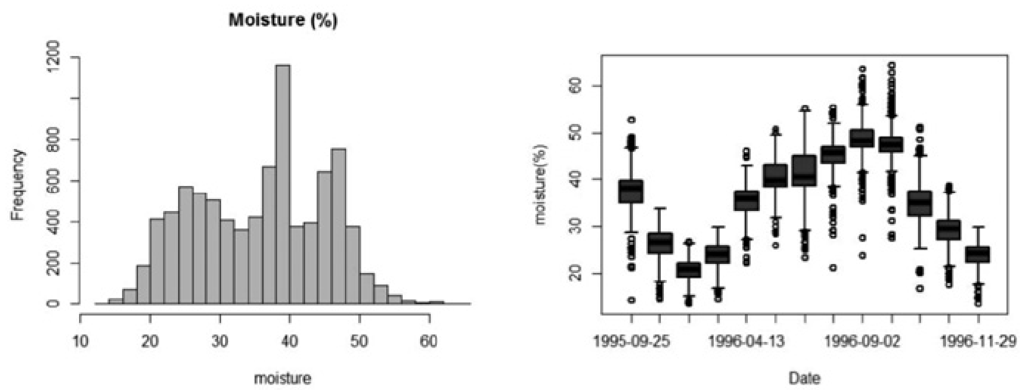


**Figure 1. Histogram of all moisture data and box-plot through time.**

*Modelling the trend*

In this study we use empirical covariates with a physical basis to model the space-time trend in the form of a multiple linear regression model. Recent work by Wang *et al*. (2011) presented a discounting factor for stream flow which was used to model sediment concentrations. The main idea of using a discount factor is that the effect of stream flow on concentration is not solely dependent on the current flow but antecedent conditions. We used the same principle here but used it for rainfall and consider discounting factors (df) of 0.5, 0.7, 0.9, 0.95, 0.99 and 0.999. These range in interpretation from being equal to the current rainfall (df = 0.999) to being equal to the average rainfall (df = 0.5). In addition, we used rainfall at time of sampling as a potential covariate. To account for a trend in space, we included clay content, elevation and topographic wetness index in the trend model.

*Modelling the residuals*

Space-time geostatistics was used to model the residuals. In a first step, the spatial and temporal variability of the residuals is analysed using a space-time variogram. To obtain a space-time representation of the variogram, the R package “spacetime” (Pebesma 2011) was used considering a space lag of 25 m and a time lag of 15 days (the output variogram presented in Figure 2 is averaged over all time lagged observation sets).

Both the predictions from the trend and residual model were summed together to produce final predictions in space and time.

**Results and discussion**

The best prediction model using backwards elimination was found to include rainfall and all of the df variables, together with the terrain wetness index (twi) and elevation. The relationship between soil moisture and the relevant environmental factors for the Tarrawarra study site is expressed by the multiple linear regression model of equation 2.

Moisture (%) = 22.5 + 4.82(rain) – 27.04(df50) + 42.18(df70) – 42.01(df90) + 30.73(df95) – 9.45(df999) + 0.89(twi) – 0.24(elevation)   (2)

The space-time variogram obtained for the model residuals is presented in Figure 2. The first temporal lag shows higher variability across space. This possibly indicates that, for the first time lag, the trend model presents also higher variability that could affect model prediction of soil moisture.

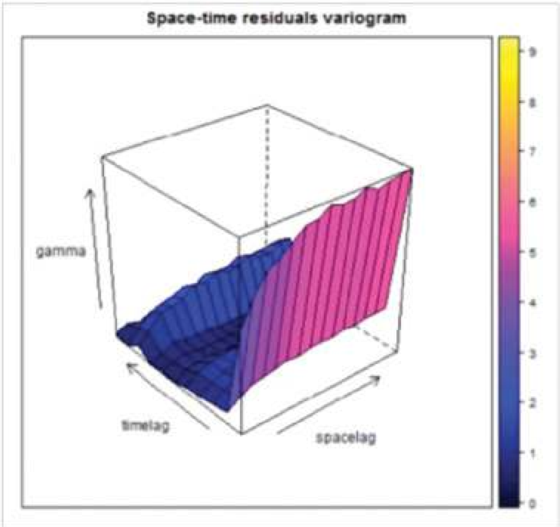


**Figure 2. Space-time residuals variogram.**

Using the fitted space-time variogram parameters, space-time kriging is performed to the residuals. The outputs are shown in Figure 3(a). An example of the final output for soil moisture distribution is presented in Figure 3(b) for the first time interval (representing a period of 15 days for September 95).

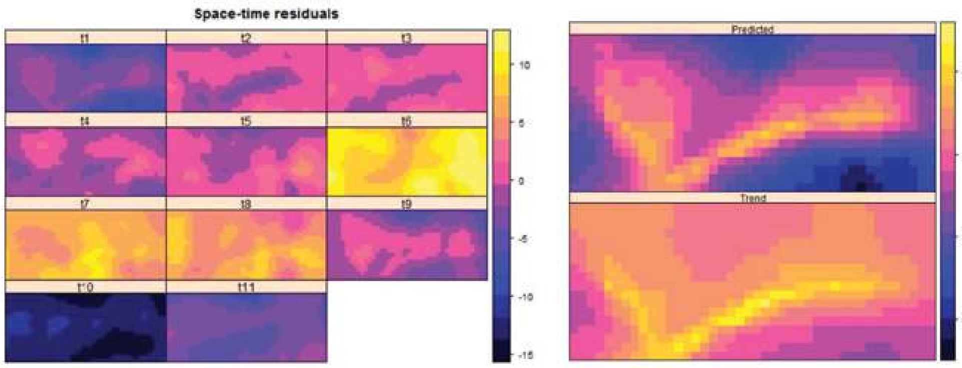


**Figure 3. Space-time residuals (a); trend and model predictions for time interval t1 (b)**

The maps of space-time residuals (Fig. 2a) are illuminating in where the trend model performs poorly in both space and time. Based on this output, we concluded that the trend model explains successfully the spatial distribution of moisture and its variation through time intervals t2 to t5. According to the distribution of experimental data in Figure 1, t2 to t5 correspond to low and intermediate soil moisture values measured in the spring and summer periods. High values of soil moisture measured for t6 to t8 seem to be underestimated by the trend. As for t10, where soil moisture is overestimated, this could be the effect of the number of observations that were twice as higher for this sampling period. Figure 2b shows an example of a model prediction based on the trend and integrating the space-time variation of residuals obtained for time t1. For this specific example, it is noticeable that the trend reproduces quite well the pattern observed for low elevation values with highest twi values. The variability observed in the space-time experimental variogram could be the result of this apparently over-influence of the spatial covariates.

## Conclusions

To better explain the variability of soil moisture in the Tarrawarra dataset, we choose to build a model able to integrate the influence of key environmental factors and to analyse its predictive performance. The most relevant key factors are rainfall and the periodicity of rainfall events as well as elevation and the terrain wetness index. The choice of these covariates was based on the results of a multiple linear regression model. The predictions given by this model represent the trend behaviour of soil moisture.

To evaluate the performance of this trend model, we analysed the joint spatial and temporal variation of model residuals (Fig. 2a) using space-time kriging. Based on these results, the model seems to under-predict soil moisture when the catchment is wet and over-predicts in drier conditions. This can help identify where improvements are needed in the model structure in terms of different covariates. Despite this, the space-time kriging of the residuals is also useful as it can be used to improve the model predictions as shown in Figure 2b which shows the trend predictions and the overall prediction found by summing the trend and residuals.

In conclusion, the use of space time models to account for joint spatial and temporal variability of the residuals provides a valuable contribution to evaluate the performance of the trend model. To implement space-time models, time monitoring of soil moisture and other soil properties is a requirement. In the near future, it is expected a continuous increase of soil data available with the widespread of remote sensing products and better and cheaper measurement devices. Hence, more and better data will be available to support space-time analysis.

The results presented for the Tarrawarra catchment could be used as a preliminary assessment of locations for installing instruments to measure soil moisture continuously in time. The number of space sampling locations could be easily optimized based on variogram analysis. Regarding time, the use of a mixture of closely spaced samples in time combined with a regularly spaced sampling will improve the estimation of the temporal model and the temporal predictions.

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## VALUATION OF NATURAL CAPITAL AND ECOSYSTEM SERVICES

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# Applying MODIS time series data to infer and map soil ecosystem services and assist in soil condition monitoring

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### Abstract

Soil landform maps are static representations of soil and land qualities that are then interpreted for land use planning, land capability assessments and agricultural development. Ecosystem services provided by soil are dynamic expressions of soil processes and their environmental interactions. Soil ecosystem services have been described by many authors over the last 20 years and there is growing agreement about what these are and how they depend on key soil properties and functions. The task of quantifying and mapping these services presents many challenges. Although logical pathways (properties-processes-functions-services) may be scientifically accepted, creating a spatially explicit representation of ecosystem services is confounded by complex interactions and limited by available spatial data. Interactions between soil and land use influence the degree to which services are delivered or impaired. Land use data, particularly land use practices, as well as hypotheses that describe or quantify these interactions, are needed in order to theoretically map ecosystem services from soil. A conceptual framework linking notions of soil as natural capital, concepts of soil health and soil quality, land capability classification, soil susceptibility to degradation, and ecosystem services provides a starting point for identifying data requirements for mapping ecosystem services. Time series EVI data available from MODIS can serve as a surrogate to map delivery of ecosystem services related to net primary production and to classify the areas with respect to potential differences in soil condition.

### Key Words

Soil assets, SOC, erosion, land capability, land use, land cover

### Introduction

The work reported here resulted from the Victorian Department of Sustainability and Environment's (DSE) need to examine how the asset diversity of soils might be mapped, valued, and used for prioritising investment in soil in order to protect it and other natural, social and economic assets. The DSE define soil as, "*a biophysical asset that is valued due to the influence it has on two key services, the provision of food and fibre and the provision of biodiversity. It is also valued for its contribution to many other less understood but potentially equally important ecosystem services, such as carbon sequestration, pollination, water and waste filtration*." (Annett and Adamson 2008). A review of the literature reveals a plethora of terms that are used to describe soil 'as an asset' or as 'natural capital'. Soil natural capital has been defined by Robinson *et al*. (2009) as "the stocks of mass, energy and their organisation (entropy) within soil". A schematic diagram to show the relationships between some of these terms within a broad 'stocks and flows' framework is shown in Figure 1. The dynamic relationships that exist between the stocks and the flows is mediated by land use and land management which serve to deliver services but may also degrade the stocks and result in disservices and poor soil health or condition. Provision of an ecosystem service from soil cannot be easily separated from the influence of land management.

Mapping the potential delivery of ecosystem services and distinguishing areas with potentially different soil condition requires an understanding and quantification of these relationships, as well as the relevant spatial data for soil quality, land capability, land use and land management. In previous work the Department of Primary Industries (DPI) has used the Land Use Impact Model (LUIM) to map threats to the soil assets (McNeill and MacEwan 2007) so we have adopted some of the logic of that approach in our methods to map ecosystem services. Soil and land data that can be interpreted for land capability (Rowe *et al*. 1991) and for soil susceptibility to degradation processes are readily available, but reliable spatial information on land use and land management practices is harder to obtain. However, the DPI is now creating an annual statewide land use and land cover map by using ground truth for land cover to calibrate and validate data freely available from NASA's Moderate Resolution Imaging Spectroradiometer (MODIS) (http://modis.gsfc.nasa.gov/data) (Morse-McNabb 2011). The Enhanced Vegetation Index (EVI) data available from the MODIS provide opportunities to interpret land cover with respect to primary productivity, potential for soil cover for protection from erosion, and relative inputs for soil sequestration of carbon (Fig. 2).

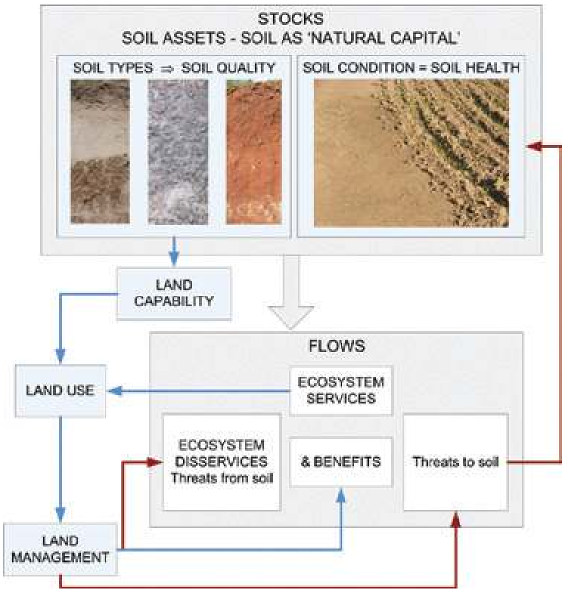


Figure 1: Representation of the relationships between soil assets, ecosystem services, land management and threats in the context of stocks and flows.

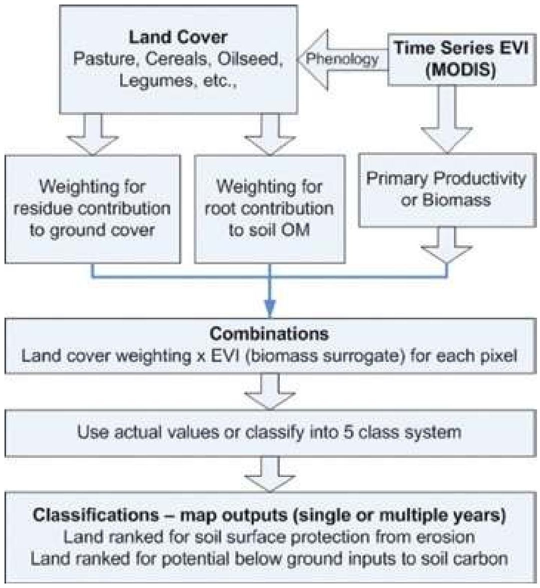


Figure 2: Flow diagram showing use of MODIS enhanced vegetation index (EVI) to classify land cover and to create maps of potential soil surface protection and root contribution to soil carbon.

## Method and Results

### Data sources

Two primary data sources were used (1) MODIS imagery from 2001 to 2009 and (2) soil information, as soil profile and map data, held by the Department of Primary Industries in the Victorian Soil Information System. Processing was carried out in ArcGIS software (ESRI 2011).

*Land cover*

The following land cover classes were adopted from data generated in the 2009 season for the Victorian Land Use Information System: woody vegetation, cereals, legumes, oilseed, pasture, bare ground and fallow. The 2009 algorithms were applied to the satellite data from 2001-2008 to generate land cover maps for those years. Classification was at the pixel scale (nominally 250x250 m).

*Primary Production*

Net primary production can be interpreted directly from EVI which serves as a surrogate for, and is linearly related to, green biomass. Time series EVI data from 2001 to 2009 were used to classify average relative productivity over the nine year period on a pixel basis (6.25 ha pixels). A statewide classification shows a strong relationship to rainfall with EVI values increasing from the north west of Victoria to the south east. Mapping the EVI data within a region of reasonably homogeneous rainfall to provide greater visual contrast reveals some sharp boundaries which coincide with land parcels and are assumed to be responses to management or soil condition rather than climate or soil type. This output provides a basis for site selection for investigating soil condition and management history. It also forms the basis for use in mapping the potential contributions to soil organic carbon (SOC) and potential residue for soil protection between harvest and crop establishment in the following season.

*Soil Organic Carbon Inputs*

We cannot predict or assume anything regarding post harvest management of above ground residues and the fate of carbon contained in them, however, it is fair to assume that the roots of the cover would remain below ground and contribute directly to SOC. We used values for root:shoot ratios from the literature to select weighting of pixel EVI values, based on the cover type, to provide a value that represented relative below ground biomass and therefore relative input to SOC. The following weights were used: Legume (4); Pasture (3); Oilseed (3); Cereal (2); failed crop and fallow (2). Statewide map output based on cumulative data for all nine years, 2001-2009, is shown in Figure 3.

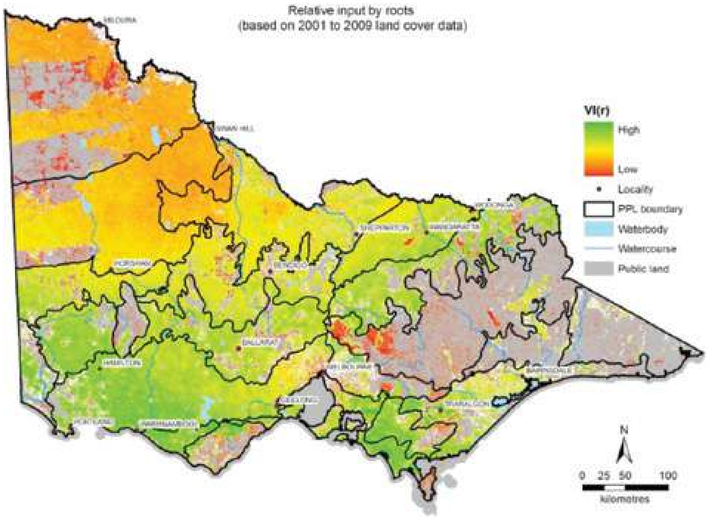


**Figure 3: Relative contribution of below ground biomass to soil organic carbon (green= high; red=low) based on 9 years land cover data, EVI weighted according to cover type and assumed root:shoot ratios. Heavy line boundaries (PPL) are Victoria's 'Primary Production Landscapes' or agro-ecological landscapes.**

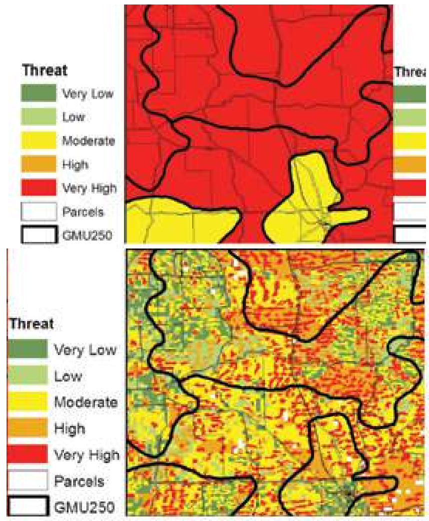


**Figure 4: Examples of wind erosion threat output using susceptibility attribution based 1:250,000 geomorphological GMU250 land units (top) and average VI(e) values or on disaggregated components and individual pixel VI(e) values (lower).**

*Threat of soil erosion*

The threat of soil erosion is made up of two factors: ground cover to protect the soil surface from wind and water, and the inherent susceptibility of soil and land to wind or water erosion. The innovation in this project was to classify the MODIS pixels for the degree of soil surface protection during the summer and autumn. Assumptions were made about the summer soil cover due to land cove types, and weightings given to each land cover as follows: Woody Vegetation (5); Pasture (5); Cereal (4); Oilseed (3); Legume (2); Failed Crop and Bare or Disturbed Ground (1). The yearly sum of EVI (total biomass) for each pixel was multiplied by the weight appropriate to its land cover class and this was done for each of the 9 years of MODIS data, producing a weighted EVI data set, VI(e), for the state for each of the 9 years. The mean weighted EVI was then calculated using the 9 years of data to create one statewide dataset for use as an input to represent soil protection in the 'erosion threat' model.

Erosion threat was classified by intercepting the soil protection layer with the 1:250,000 geomorphological (GMU250) soil and land units attributed for susceptibility and statewide divisions for aridity and rainfall erosivity. The steps are outlined below:

Step 1: A soil susceptibility rating for each erosion process is assigned to each GMU250 soil-landscape unit, based on dominant soil class and average topography, or to a finer resolution land component where that exists (e.g. the Victorian Mallee land components have been disaggregated using a 10 m digital elevation model).

Step 2: For wind erosion, the State is stratified into three regions of Aridity. For water erosion, the State is stratified into 3 regions of Rainfall Erosivity.

Step 3: The soil susceptibility and the Aridity / Rainfall Erosivity classes are combined in LUIM using a classification table to produce an overall susceptibility rating for each GMU250 polygon.

Step 4: A weighted vegetation index, VI(e), according to the land cover potential to protect the soil from erosive forces is produced.

Step 5: Each GMU250 unit is assigned a relative (Very good to very poor) soil protection class based on the average VI(e) pixel value within the unit.

Step 6: A classification table (table 1) was used to combine the soil susceptibility and the soil protection class of each GMU250 unit or finer scale land component to produce an erosion threat rating for that unit. Example outputs are shown in Figure 4.

**Table 1. Erosion threat rating matrix to relate susceptibility of soil to erosion (by water or wind) and the degree of expected summer ground cover contribution from the land cover class.**

| Soil Protection | Soil Erosion Susceptibility | | | | |
|---|---|---|---|---|---|
| | Very Low | Low | Moderate | High | Very High |
| Excellent | Very Low | Very Low | Very Low | Low | Moderate |
| Good | Very Low | Very Low | Low | Moderate | High |
| Fair | Very Low | Low | Moderate | High | Very High |
| Poor | Very Low | Low | Moderate | High | Very High |
| Very Poor | Low | Moderate | High | Very High | Very High |

## Discussion

In mapping below ground SOC contributions and potential ground cover during summer, due to different land cover types, we have had to draw on the available literature and make some large assumptions. However we are confident that the method offers a sound way to model and map these two factors that reflect and contribute to ecosystems services from soil. The input data will be improved by quantifying:

- EVI values and actual biomass (t/ha). We are confident that EVI is linearly related to green biomass but currently have no data for our land cover types and conditions.
- Land cover biomass and post harvest crop residue (t/ha and %ground cover)
- Root:shoot ratios for major cover types and crop varieties. These data are time consuming to collect but would be essential for SOC models, particularly those using satellite imagery.

## Conclusion

The freely available MODIS data has value in providing a spatial understanding of the relative contribution of soils to ecosystem services and functions related to primary production, in particular the methods reported here can:

- Distinguish high value soil assets from poorer ones
- Indicate areas where soil may be in a degraded state or poor health
- Map the threat of soil erosion
- Provide a correlative input for modelling SOC sequestration
- Guide spatial selection of areas and design for long term, landscape scale monitoring of farming systems, productivity and soil condition

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# An ecosystem services approach to the evaluation of soil conservation in New Zealand hill country pastures

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Over the last 50+ years, investment in soil conservation in New Zealand to reduce soil erosion and associated downstream costs has run into many millions of dollars. Current evaluation of such ecological infrastructure investment is largely limited to reduction in soil loss and sediment. The value of the soil ecosystem services retained by preventing or reducing erosion has never been included.

This study uses a recently developed framework considering thirteen soil services to quantify and value the loss of ecosystem services following an erosion event and the impact of an investment in soil conservation on the protection and restoration of such services.

Previous use of the framework to quantify and value soil services under a dairy farm operation has shown that the value of provisioning services (US$3,430/ha/yr), including the provision of food and support, was less than the value of regulating services (US$9,090/ha/yr), including flood mitigation, greenhouse gases regulation, filtering, detoxification and pests regulation.

The same methodology is used to quantify soil services under a typical sheep and beef operation in hill country affected by erosion, and assess the cost-efficiency of an investment in soil conservation. Quantitative information was scaled up using satellite imagery and valued using neo-classical economic valuation techniques.

Understanding how current investments in built capital and current and future investments in ecological infrastructure are likely to change the flow of ecosystem services from managed landscapes is critical to assess the efficiency, cost-effectiveness and sustainability of resource management policies, and to increase political and public awareness of the value of land.

# Mapping soil natural capital

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Soil natural capital has potential as an alternative to conventional land evaluation because it enables soil to be assessed in relation to its ability to provide ecosystem services. This allows soil natural capital to be used as a more direct estimate of soil assets of a farm or region for a given land use. We used the stock adequacy method to estimate a soil natural capital index for a suite of representative soils on the Canterbury Plains, New Zealand. The estimation was made with reference to intensive dairying and considered regulating services; nitrogen filtering, water supply, aeration, and drainage. The suite of soils comprised two soil depth sequences from deep loamy stone free material to deep stony material, predicted to vary in their ability to filter nitrogen and store water and provide drainage. . One sequence was in younger soils with low clay and the other in older soils with high clay. The results expressed the comparative capacity of the soils to provide regulating services.

# Setting of nutrient loss limits

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The different approaches for the allocation of a nitrogen (N) leaching discharge allowance from land to protect the quality of receiving water bodies is currently a major environmental policy topic in New Zealand.

This paper compares several options that have been considered in arriving at a discharge allowance, including: placing limits on intensive land use; benchmarking; nutrient use efficiency; input-based controls; best management practices; grand-parenting (i.e. the capping of N leaching losses at levels based on emissions from current land uses); allocation based on land area in the catchment; and the use of a capital "natural capital" approach.

In the short-term, placing a limit on the N leaching losses from existing intensive land uses only offers an option for stopping further decline in water quality. This assumes that there is no further expansion of intensive agriculture. The use of a single discharge value for each hectare in the catchment addresses this weakness, offering a simple, effective and seemingly "equitable approach" for all landowners in the catchment. This approach assumes that all land is equal, which is clearly not the case as soil scientists know. Most landscapes contain a diversity of soils differing in natural capital and varying in their provision of ecosystem services (e.g. the ability to attenuate N losses). In developing policy for setting discharge allowances to protect water quality clear recognition needs to be given to the implications of the policy to limit, or allow future land use options. Land is a finite resource.

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# Using a spatially explicit ecosystem service approach to analyse agricultural landscapes of northern Victoria

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The true value of soil ecosystem services is rarely considered in land management planning and policy development. Under-appreciation of soil value can lead to practices that decrease soil capital and the associated capacity of the soil to provide ecosystem services. In this study, we will quantify and map the supply of selected soil ecosystem services within case study landscapes that encompass 305 km2 in northern Victoria, Australia. Historical clearing and intensive agriculture in these landscapes have contributed to degradation of soil capital through erosion, compaction and localised salinisation. However, a 25 year plan (Future Farming Landscapes, VicSuper) allows for extensive reconfiguration of land uses to improve both productivity and environmental protection. These landscapes are therefore ideal for examining changes in soil ecosystem services under future land use scenarios. Our experimental approach involves an innovative combination of diverse methods, including soil sampling according to a geostatistically-valid design, soil analysis using mid-infrared spectroscopy for key characteristics, application of the APSIM (Agricultural Production System Simulator) suite of models to estimate the supply of key soil properties and services at sampling points on the landscapes, and application of spatial geostatistical methods (e.g. ordinary kriging) within ArcGIS to produce maps of soil service distribution. Our study will develop maps of ecosystem service supply under current conditions and under future scenarios (defined by policy drivers, land use practices/regimes, and possible future climate). This information will support planning and policy decision making.

# Soils, food production, ecosystem services and ecological infrastructure

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Ongoing growth in both population and consumption is placing unprecedented, and in many cases unsustainable, pressure on the world's soil and water resources, which has resulted in increased demands for food production systems that incorporate the value of ecosystem services. We argue that a more proactive approach, including significant investment in ecological infrastructure (EI), is needed. The ecosystem services approach (ESA) attempts to identify and value those processes and functions, or 'services', that are provided by ecosystems and exploited for human benefit but not valued economically in terms of decision making. Despite its increasing adoption in natural resource management and agriculture, the ESA is fundamentally flawed, from both ecological and economic perspectives, and has been criticized by economists, environmentalists and even some of its own proponents. A major gap in the ESA is the neglect of the underlying EI, which provides the ecosystem services and sustains their provision into the future. In this paper we conceptualize EI as consisting of key elements, systems, processes and functions and the all important connectivity among and within them. We outline the conceptual and practical differences between identifying and valuing discrete 'services' and investing in the underlying infrastructure that provides them, and argue that investing in EI is at least, if not more important, than investing in our built infrastructure. We conclude by highlighting the importance of pore connectivity and soil water flow and transport as essential features of a robust and resilient soil EI that can be maintained and enhanced, through carbon investment strategies.

# Friday 7 December

## FORENSICS

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# Soil as significant evidence in a sexual assault case

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## Abstract

In November 2007, a 10 year old girl was kidnapped in her Paralowie home in Adelaide, South Australia and sexually assaulted by a male in the nearby Paralowie school grounds. The girl escaped from the suspect and was located by family 100 m from her home as she ran back from the school at about 4.30 am. The suspect was caught after using a credit card he stole from the victim's house. Shoes were collected from the suspect's residence, which is within 500 m of the victim's home and the Paralowie School. The suspect denied any contact with the victim or with the crime scene. The whole area is residential, except for the school grounds, which has un-grassed playing fields and adjacent woodlands (i.e. where the victim was sexually assaulted twice). The soil type underlying the area comprises Red-Brown Earths (i.e. Red Chromosols or Red Duplex Soil). Soil was collected from: (i) the suspect's shoes (two questioned samples), (ii) the school grounds at two crime scenes (four control samples) and (iii) a range of likely alibi sites near both the victim's and suspect's homes (nine alibi samples). Analysis using soil morphology, microscopy and X-ray diffraction indicated that one specific alibi human-made soil collected from the exposed strip between road and footpath at the victim's home and soil on the top of the sole of suspect's shoes were remarkably similar (i.e. had an extremely strong "degree of comparability" of being from a single location). This alibi soil had a reddish brown colour and is clayey, had a markedly similar mineralogy (i.e. all nine minerals identified had similar composition and crystallinity: quartz, albite, orthoclase, mica, kaolin, calcite, hematite, chlorite and dolomite) to the soil from the top of the sole of the shoe. The suspect pleaded guilty to the attack and was sentenced to a non-parole term of seven years for abducting a child from her house and raping her.

**Key Words**

Forensic soil science, questioned samples, control samples, alibi samples, human-made soils, soil morphology, mineralogy, x-ray diffraction.

## Introduction

Soil can be used to indicate or compare provenance, and therefore be used as a source of police intelligence and subsequently trial evidence to both narrow areas of search during an investigation and aid convictions. Evaluative comparison of soil on one article of evidence compared to another, or compared to a known location, can and has been used as evidence in courts of law (e.g. Fitzpatrick, 2009; Fitzpatrick *et al*., 2009; Fitzpatrick and Raven 2012a; Murray, 2011; Pye, 2007; Ritz *et al*., 2008; Ruffell and McKinley, 2008). Forensic soil examination can be complex because of the diversity and heterogeneity of soil samples. For example, the diversity, heterogeneity and complexity of human-made soils, especially in residential areas that contain transported or manufactured materials such as road metal materials, enables forensic soil examiners to distinguish between soils which may appear to be similar (e.g. reddish soil materials, which may contain minor amounts of road metal comprising specific minerals such as dolomite). Forensic soil scientists must determine if there are unique features of questioned soils crucial to an investigation that enables these soils to be compared with control soils or soil/geological map units from known locations.

Soil science expertise was used to help solve a double murder by identifying the similarities between soil and clay assemblages on a shovel and from a quarry (Fitzpatrick and Raven, 2012a). The soil material and clay mineralogy had a common provenance and revealed the location of two buried bodies. This successful case led to the formation of the Centre for Australian Forensic Soil Science (CAFSS) in 2003. Since then, the Centre has advised on over 100 forensic investigations. Problems in recognising the potential of forensic soil science can be overcome by sharing successful case examples to make soil scientists and police aware of the value of soil evidence. For example, Dr Raymond Murray in his book entitled "Evidence from the Earth" (Murray, 2011) presents several high-profile legal cases in which geological materials and methods have contributed significantly to solving cases from around the world. To draw attention to this underutilized forensic tool, we present in this paper some important ways by which soil evidence was proven useful in helping to solve a complex rape case in 2009. We also highlight the critical importance of: (i) sampling soils from a range of likely alibi sites and (ii) identifying the presence of an unusual mineral, which at a specific location had become mixed with the common soil underlying the entire crime investigation area.

**Materials and Methods**

In November 2007, a 10 year old girl was kidnapped in her Paralowie home in Adelaide, South Australia and sexually assaulted twice by a male in the nearby Paralowie school grounds (Figure 1: locations of victims house and crime scenes). The girl escaped from the suspect and was located by family 100 m from her home as she ran back from the school at about 4.30 am. The suspect was caught after using a credit card he stole from the house. Upon questioning by investigators, the suspect denied any contact with the victim or with the crime scene. Police collected shoes containing soil [Figures 2(a)] from the suspect' s home (Figure 1) and submitted them to the CAFSS for examination, along with two control soil samples (CAFSS_037.1, 2) taken near footprints where the alleged sexual assault took place in the school grounds (Figure 1). These two questioned and two control soil samples "appeared similar" because of their distinctive reddish-brown colours and clayey textures. However, it was established that the school grounds and whole adjacent residential area is also underlain by a common reddish-brown soil type, namely Red-Brown Earths (Northcote *et al.,* 1960-68) or Red Chromosols (Isbell, 1996), which includes the suspect's residence that is within 500 m of the victim's home and the Paralowie School. Consequently, the area was revisited on 17th February 2008 to collect additional control and alibi samples from crime scenes and a series of likely alibi sites. As shown in Figure 1, the following twelve soil materials were sampled from: (i) four control sites (Figure 2 where samples of loose - possibly transferable soil were taken at or in very close proximity of crime locations) and (ii) nine alibi sites [samples were collected on the surface (0 to 3 cm)] to determine whether or not the suspect had transferred soil materials from near his house and/or the house of the victim. The alibi samples were used to determine differences or similarities with the control samples (e.g. Figure 2).


**Figure 1** Google earth image showing the localities of the four control sites and nine alibi sites (from Fitzpatrick and Raven, 2008).


**Figure 2**

**Figure 2. From top to bottom photographs of: (a) top of rubber shoe sole with tooth-like edge moulding showing brown (7.5YR4/4 moist) clayey soil (Questioned soil; CAFSS_037.4), (b) of brown (7.5YR4/4 moist) clayey soil in the wooded area at crime scene (Control soil sample CAFSS_037.7), (c) & (d) reddish brown clayey soil with white quartz and dolomite gravel (Alibi soil sample, CAFSS_037.16) exposed in bare patches of lawn between the pavement and road in front of the victims house (Fitzpatrick and Raven, 2009).**

Forensic soil characterisation uses descriptive steps for rapid characterisation of whole soil samples for screening (i.e. Soil Morphology; Stage 1), and detailed characterisation and quantification of composite and individual soil particles after sample selection, size fractionation (< 50 microns) using mineralogical and organic matter analyses with advanced analytical methods such as X-ray powder diffraction (XRD) (Stages 2 and 3). Morphological soil descriptors are arguably the most common and simplest; it is for this reason that all composite samples are characterized first using morphological descriptors - using standard Australian (McDonald and Isbell, 2009) and international (Schoeneberger *et al*., 2002) soil morphological methods. XRD methods are arguably the most significant for both qualitative and quantitative analyses of solid materials in forensic soil science.

## Results and Discussion

*Stage 1: initial characterization of composite soil particles in whole samples for screening*

The soil samples from the shoes were completely different in morphology (colour, texture and structure) to the following seven alibi soil samples taken from: (i) the bare patches of lawn in the back yard near the back door and front entrance-strip between road and footpath at the accused's house (brown sandy soils), (ii) in the bare front lawn at the victim's house (grey sandy soil) and (iii) the walkways near and in the Paralowie school (gravelly soils) (Figure 1). Hence, the seven samples collected from these localities have a "low degree of comparability" with samples from the shoes and are not likely to have been sourced from the same locality (Fitzpatrick and Raven 2009). Consequently, because of screening only nine samples with similar morphology (Munsell colour, texture and consistence) were analysed by XRD (Stages 2 & 3; Table 1).

*Stages 2 and 3: detailed characterization of composite and individual soil particles*

Sufficient descriptive and mineralogical (XRD) data were acquired on all soil samples to determine the major similarities and differences between the samples using "Categories of Comparability" as defined by Fitzpatrick and Raven (2012b).We established the "degree of comparability" of the soil materials from the shoes (two samples) being "comparable" or "not comparable" to the 15 representative control and alibi soil samples from: (1) Paralowie school oval (CAFSS_037.1, 2) and adjacent wooded area park (sexual abuse location; CAFSS_037.7), (2) wooded area adjacent to an Oval in the Paralowie school (sexual abuse location CAFSS_037.10), (3). Side yard lawn, back yard lawn and front entrance-strip between road and footpath at the accused's house) and (4) the front lawn and strip between road and footpath at the Victim's house (Fig 1).

The brown (7.5YR 4/4) clayey soil on the top of the sole of the shoes had markedly similar soil morphological properties (soil colour and texture) and mineralogy (i.e. all nine minerals identified have similar composition and crystallinity: quartz, albite, orthoclase, mica, kaolin, calcite, chlorite, hematite and dolomite (and does not contain amphibole) (Table 1). The fine detail expressed in the XRD patterns from these two soil samples are remarkably similar as shown in Figure 3 (i.e. likened to "finger print comparisons' showing subtle similarities and differences). The brown (7.5YR 4/4) clayey soils found at the two control sites (i.e. sexual abuse locations in the Paralowie school grounds) also had similar soil morphological properties (soil colour and texture) and mineralogy (quartz, albite, orthoclase, mica, kaolin, calcite, amphibole and hematite; both do not contain dolomite) to the soils found on the bottom of the sole and top of the sole of the shoes.

**Table 1: Summary of mineralogical composition from XRD analysis for samples with similar morphological properties**

| CAFSS (IDENT and description) | Q | Alb | Orth | Mi | Kt | Chl | Amp | Ct | Dt | Ht |
|---|---|---|---|---|---|---|---|---|---|---|
| 037.1 Paralowie school, surface[2] | D | M | T | T | T | T | T | T | - | T |
| 037.2 Paralowie school, subsoil[2] | D | M | T | T | T | T | T | T | - | T |
| 037.3 Bottom of sole[1] | D | M | T | T | T | T | - | M | - | T |
| 037.4 Top of sole[1] | D | M | M | M | T | T | - | T | T | T |
| 037.5 Nodule from 037.2[2] | CD | T | T | T | T | T | - | CD | - | T |
| 037.7 Paralowie school, adjacent park[2] | D | M | T | T | T | T | T | T | - | T |
| 037.10 Paralowie school, surface[2] | D | M | T | T | T | T | T | T | - | T |
| 037.12 Accused's home back yard[3] | D | M | T | T | T | - | T | T | - | T |
| 037.16 Victim's home footpath[3] | D | M | T | T | T | T | - | T | T | T |

**Where**: Q= Quartz; Albite = Alb; Orthoclase = Orth; Mica = Mi; Kaolin = Kt; Chlorite = Chl; Amphibole = Amp; Calcite = Ct; Dolomite = Dt; Hematite= Ht

D – Dominant (>60%), CD – Co-dominant, SD – Sub-dominant (20% to 60%), M – Minor (5% to 20%), T Trace (<5%).

[1] Questioned shoe samples (CAFSS_037.3 and CAFSS_037.4) were not sieved because they are approximately <50 micron.

[2] Control samples (Paralowie school) were sieved (<50 micron fraction). [3] Alibi samples were sieved (<50 micron fraction).

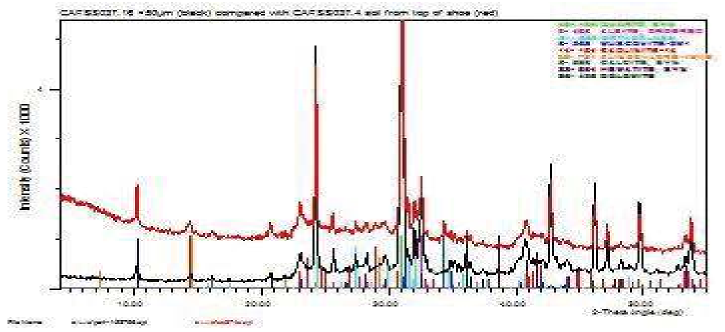

**Figure 3: Comparison between X-ray diffraction (XRD) patterns of the questioned soil sample from the top of the right shoe sole (CAFSS_037.4) (red) and the reddish clayey alibi soil between road and footpath at victim's home (CAFSS_037.16) (<50 micron fraction) (black).. Samples were both ground using an agate mortar and pestle before being lightly pressed into aluminium sample holders for XRD analysis. XRD patterns were recorded with a Philips PW1800 microprocessor controlled diffractometer using Co Ka radiation, variable divergence slit and graphite monochromators.**

## Conclusions

*Stage 4: integration and extrapolation of soil information from one scale to next*

The soil recovered from the top of the sole of the shoe (questioned sample) has: (1) an extremely strong degree of comparability to the alibi soil sample taken from near the victim's house between the foot path and road (CAFSS_037.16); (2) a moderately strong degree of comparability to the 4 control sites (i.e. sexual abuse locations) on the Paralowie school oval (CAFSS_037.1; 2) and adjacent wooded areas (CAFSS_037.7; 10) and (3) a moderate degree of comparability to the sandy clay loam alibi soil material recovered from the back yard of the of the accused house (CAFSS_037.12).

The alibi soil taken from near the victim's house between the foot path and road (CAFSS_037.16) differed in containing dolomite compared to other control and alibi soils, which also had reddish-brown colours and were clayey, i.e. control samples from the Paralowie school and an alibi sample from the accused back yard. The occurrence of dolomite supported the interpretation that this alibi soil and the soil on the top of the sole of the shoe had very likely originated from a specific location on the footpath at the victim's house where the natural reddish brown clayey soil and dolomite-rich road metal had mixed to form a somewhat unique "human-made soil". The evidential significance of the shoe deposit aided in the police investigation and led to the accused (Allan John Hopkins) confessing to abducting the 10 year old child from her house and raping her. On 16th July 2009, he was sentenced to a non-parole term of seven years.

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# Does organic matter in soil forensic science ?

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Forensic science can potentially tap into a wide range of modern analytical methods to analyse trace evidence including soil. Traditionally soil forensic practitioners have largely relied on identification of mineral particles to provide information on potential geological provenance and on pollen, spores and plant fragments to provide complimentary information on relating to vegetation. Recently attention has turned toward soil organic matter and the associated microbial community to address questions of a forensic nature. We compared the potential of organic chemistry (n-alkanes, fatty-alcohols, mid-infra red spectroscopy) and molecular biology (bacterial, fungal, archaeal TRFLP) to discriminate between soil sources of contrasting land-use. Soil fatty-alcohol and fungal DNA analyses complemented mineral analyses to the greatest degree, providing the most promising indicators of broad-scale land-use. Based on a mock scene of crime study, the impact of transfer, persistence, and cross-transfer on organic and biological profiles were explored. The key forensic considerations required to promote organic and molecular profiles from the research context towards robust forensic application are discussed.

# Development of soil forensic methods and databases from targeted locations in Tasmania to assist Police

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Across Australia, Police rely heavily on DNA evidence and witness testimony to solve crime. Historically, this provided enough evidence to charge an offender. But in recent major crime investigations, soil evidence compared and characterised using the approach and methods outlined by CAFSS (Fitzpatrick and Raven 2012), has proved crucial in successfully assisting police and convicting offenders in Australian Courts of Law. However, most Police forensic laboratories are ill-equipped to analyse forensic soil evidence. A rising tide of unsolved 'cold case' major crime, with no DNA evidence or reliable witnesses, is 'gathering dust' on Police books. With assistance from Tasmania Police, forensic soil data from specific soil types at the following targeted locations will be gathered, to assist with new avenues of inquiry:

- Key soil catenary sequences near Richmond, Hobart will be used to test distinctness of mixed pit spoil and single pit spoils via "credible scenarios of crime scene approach".
- Human-made or human-transported soils (Anthroposols) in a high-crime area of Hobart.
- Organic-rich 'peaty'soil (Organosols) from high-altitude in Tasmania.
- Coastal and inland Acid Sulfate Soils from Tamar Estuary, Launceston.
- Cultivated soil from commercial opium poppy plantations across Tasmania.
- Use of spectrophotometry to quantify colour of a diverse range of Tasmanian soil types.

Soil samples will be characterised according to the four-stage process developed by CAFSS, including detailed soil morphology characterisation of soil, mineral and organic composition using a combination of X-ray diffraction (XRD), magnetic susceptibility, heavy mineral and magnetic fractionation. Soil-regolith conceptual models and maps will complete analysis of each site. The capacity of different soil properties to discriminate between not only diverse soil types, but very similar soil samples from close geographic locations, as well as soil trace evidence, shall be investigated.

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# Soil forensics aid in Tasmanian murder case

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Tasmania has a very diverse range of soil formation environments due to wide variations in geology, landforms, landscape history and climatic setting. Parent materials range from mafic to highly siliceous, with some carbonates; mean annual rainfall from 400 to 3000 mm/year while soil formation periods range from a few thousand to several millions of years. This setting leads to a wide range of soil types with vastly differing mineralogy, chemistry and stratigraphy occurring in relatively close proximity. This diversity can assist in criminal investigations and prosecutions via soil evidence helping to both eliminate and/or locate crime scenes from an investigation or to support presence or absence of incriminating additions or modifications to soil materials.

We used X-ray diffraction and a sequence of soil profile descriptions at a crime scene to show earthy lime rich materials found above the murder victim in a deep pit (4 m) were exotic to that environment. The local soil type, in which the victim was located, was a very deep Acidic Red Ferrosol formed from dolerite in the Tasmanian highlands. The subsoils of the soil profiles described adjacent to, upslope and down slope of the murder site had acid reaction (pH $<$ 5.0), were deeply weathered, clay textured and red to dark reddish brown in colour. X-ray diffraction analysis undertaken identified the light coloured, highly alkaline (pH 11.8) lime rich materials as being composed of 90 – 95% calcium carbonate with traces of calcium hydroxide (“lime”). These properties supported the incompatibility of the “lime” with the soil, showing it to be exotic to the site. This fact helped to link the perpetrators to the crime scene.

# Static Posters

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# The Land and Soil Management Community of Practice – past, present and future

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A community of practice (CoP) is broadly defined as a group of people who share a craft and/or a profession. The land and soil management CoP (LSMCoP) was set up in 2004. The concept is a group of like minded practitioners and peers in the soil and land assessment area sharing information. It was set up as an in house program within the former NSW Department of Infrastructure Planning and Natural Resources. The membership has become increasingly diverse due to departmental splits, individuals changing employers and requests to join the LSMCoP from persons outside of the original department. The LSMCoP now encompasses 177 people from 46 different organisations. In practice it is a large e-mail list to which technical and professional information is circulated. It enables soil and land management parishioners to: share soils info; pose soils questions to the group; and to distributed information on up coming soils and land management events. The aim is to keep the passion for soil science and land management going. Hopefully it also provides the feeling of a 'soils community'. After 8 Years of operation it is timely to review the future direction of the LSMCoP, the type and nature of information to be shared and to sound out potential new members.

# Soil peels for soil education

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It is challenging to excite people about soil. Yet soil, the foundation upon which we live and the material that sustains life needs to be promoted as a valuable asset and resource to the community. To achieve this, soil specialists need to prepare and present creative, innovative and attention grabbing tools to demonstrate their field of science. At secondary school levels, geology, landscape and soil science are all topics taught through the national geography school curriculum and teachers less equipped to present on specific areas, often call upon specialists. A soil extension tool and technique being used extensively in Western Australia by the Department of Agriculture and Food, are Soil Peels (soil held together with cloth material and soluble latex). Soil peels are an exciting, easy to use and inexpensive technique for collecting an exact reproduction of soil profiles and are a practical way of bringing the paddock to the class room. While preparing and mounting each soil peel can be time consuming, the final product creates considerable interest by demonstrating a functional and striking display that can be used in schools, universities, farming community open days and workshops or even hung in the local art gallery promoting soils to a wide audience.

# Land management impacts on soil carbon levels in four Tasmanian soil types

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The objectives of this research were to determine the effects of land management practices on soil organic carbon in a range of cropped and pastoral soil types in Tasmania. A total of 276 sites were selected on Dermosols, Ferrosols, Vertosols and a group referred to as texture contrast soils (Chromosols and Sodosols). Information was collected using farmer surveys of land management practices over the preceding ten years at each site including data on; crop rotations, type and rates of fertilisers applied, number of tillage operations, grazing and residue management, irrigation application and amendments, such as lime, applied. The land management data were collated into quantifiable groupings for statistical analysis which included quantifying the regularity of; cropping, irrigation, hay cutting, fallow, types of grazing management, types of tillage, and average fertiliser applications. The impact of the cropping was quantified by attributing numerical weightings to different crops based on the perceived impact on soil carbon. A similar tillage figure was calculated by attributing numerical weightings to different tillage practises. Initial analysis has identified irrigation to be negatively associated to soil organic carbon concentration and the frequency of growing perennial pasture to be positively correlated with soil organic carbon concentration. Additional results will be presented in the conference paper.

# Salinity reduces the ability of soil microbes to utilise cellulose

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Soil salinity has been shown to affect some microbes more than others and may therefore also affect the ability to decompose carbon substrates. To determine the response of soil microbes to salinity when supplied with different carbon forms, one non-saline and three saline soils of similar texture (sandy clay loam) with electrical conductivity ($EC_{1:5}$) 0.1, 1.1, 2.3 and 4.2 dS $m^{-1}$ were amended with 2.5 and 5 g C $kg^{-1}$ (5C, 2.5C) as glucose and cellulose, soluble N and P were added to achieve a C/N=20 and C/P=200. Microbial biomass C and available N and P were determined on days 2, 7, 14 and 21. Cumulative respiration on day 21 was higher with 5C than with 2.5C and higher with glucose than with cellulose. Cumulative respiration decreased with increasing EC, but whereas the decrease was gradual with glucose, there was a sharp drop in cumulative respiration from the non-saline soil to EC1.1 with little change at higher EC.  Microbial biomass C and available N and P concentrations were highest in the non-saline soil but did not differ among the saline soils. Microbial biomass C was higher and available N was lower with 5C than with 2.5C. With glucose, microbial biomass was highest on day 2 and then decreased whereas available N was lowest on day 2 and then increased. With cellulose, microbial biomass C increased gradually over time and available N decreased gradually. It is concluded that salinity reduced the ability of microbes to decompose cellulose more than glucose utilisation.

# Turnover times of soil organic carbon in aggregates based on radiocarbon measurements

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Radiocarbon (C-14) is a useful tool for studying carbon dynamics in soil aggregates. We hypothesized that (1) the turnover of soil organic carbon (SOC) in macroaggregates is more rapid than that of microaggregates (2) agricultural land uses, such as cropping, preferentially remove younger carbon from soil aggregates. Contrasting land uses were investigated on sites with Dermosol soils (equivalent to Alfisol in the US Soil Taxonomy) supporting native pasture, crop-pasture rotation and woodland, and sites with Ferrosol soils (equivalent to Oxisol in the US Soil Taxonomy) with improved pasture, cropping and forest land uses. Soil aggregates were separated into macro- (>250 μm) and microaggregates (<250 μm) by wet sieving. Signatures of C-14 in macro- and microaggregates were determined by Accelerator Mass Spectrometry (AMS). Radiocarbon contents in both macro- and microaggregates were >100 percent modern carbon which indicated the presence of post-bomb carbon in soil. The turnover time of SOC in both macro- and microaggregates was >50 years longer in Ferrosols as compared with Dermosols. SOC in microaggregates had a longer turnover time than that of macroaggregates in Dermosols. Microaggregates in native pasture and cropping had 40 and 19 years longer SOC turnover time, respectively, than that of other land uses. Higher SOC turnover time in cropping indicated removal of younger SOC during cultivation.

# Identifying opportunities to reduce greenhouse gas emissions from agricultural production: A Life Cycle Assessment approach

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**Keywords:** Life Cycle Assessment, greenhouse gas emissions, wheat production, cotton production, legume rotations, nitrous oxide

The use of Life Cycle Assessment (LCA) to determine environmental impacts of agricultural production is increasing, especially to determine greenhouse gas emissions. We have undertaken LCAs of wheat production in Central Zone (East) NSW (Figure 1) and cotton production at Narrabri NSW (Figure 2) and found the emissions profiles to be dominated by the production and use of synthetic nitrogenous fertilisers, at 67% of the emissions profile for wheat and 68% for cotton. When we studied the effect of replacing these fertilisers with biologically fixed N, emissions from a legume-based system were found to be 39% of those from a non-legume system (Table 1). Other factors which greatly influence calculated emissions are yield and choice of direct nitrous oxide emissions factor.

We are primarily taking a regional approach to LCA and accommodating variability within regions. As it is not possible to build case-study scale LCAs across all farms in NSW, robust regional-scale LCAs provide a basis for advising producers on potential emissions reductions. Also, by showing inter- and intra-regional variability they provide a stronger platform for testing national policies, than coarser-scale studies.

Whilst most emissions from livestock enterprises are not soil-related, we found direct and indirect $N_2O$ emissions from animal wastes to contribute 10% of the emissions profile for a regional wool production system. Therefore, further field-based research into these emissions would be valuable.

Our future work will involve regionalising our cotton LCA and studying the opportunities to reduce emissions from cropping systems, especially through the use of legume rotations.

**Table 1. Total greenhouse gas emissions for canola-wheat and chickpea-wheat production systems with different levels of fertilizer N, i.e. zero (0N) or 80 kg N/ha (80N)**

| Rotation[A] | Pre-farm and on-farm emissions year 1 (kg $CO_2$-e/ha) | Pre-farm and on-farm emissions year 2 (kg $CO_2$-e/ha) | Total emissions per ha over 2 years (kg $CO_2$-e) |
|---|---|---|---|
| Canola (80N)–wheat (80N) | 768 | 780 | 1548 |
| Chickpea (0N)–wheat (80N) | 306 | 720 | 1026 |
| Chickpea (0N)–wheat (0N) | 306 | 297 | 603[B] |

[A]The canola and chickpea crops yielded 1.8 t/ha; the three wheat crops yielded 3.0 t/ha; [B]Total emissions from chickpea (0N)–wheat (0N) are 39% of those from canola (80N)–wheat (80N); using Ecoinvent and the Australasian LCI database updated May 2012.

# Field decomposition rates of single and mixed litter species under two contrasting climatic zones – wet tropical and semi arid

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Soil Organic Carbon (SOC) stock assessment remains a complex research frontier because of diversity and distribution of vegetation types, their associated carbon inputs and interactions with the soil and climatic environment that govern decomposition rates. In the context of mine land rehabilitation, there is an opportunity to establish vegetation communities with a strategic aim to rapidly enhance SOC accumulation. Hence, as an initial step towards determining the effect of vegetation type on the rate of SOC accumulation, this study compared field decomposition rates of leaf litter from a range of forest and pasture vegetation species under contrasting climatic regimes; wet tropical (Java, Indonesia) and semi-arid (Central Queensland). Litter from dominant vegetation community types, with and without nitrogen-fixing species, was placed in mesh litter bags and pegged onto the soil surface for 8 weeks (Java) and 12 weeks (Queensland), at each of a forest and pasture site, in each country location. Australian litter was taken to Java but not vice versa. Soil types at the sites were Vertosol (Central Queensland) and red Ferrosol (Java). Decomposition, measured by litter bag mass loss at set time intervals showed that under wet tropical conditions, Australian litter decomposed up to 41% faster than Indonesian in a forested situation and up to 17% slower in a pasture situation while mixed litter decomposed faster than single species by up to 18% in forest and 21% in pasture. Under semi arid conditions mixed litter decomposed faster than single species and litter in pasture decomposed faster than in forest.

# The availability of residual legume and fertilizer nitrogen to a subsequent wheat crop under elevated $CO_2$ concentration

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## Abstract

The effect of elevated carbon dioxide concentrations ([$CO_2$]) on the residual contribution of legume and fertilizer nitrogen (N) to a subsequent crop was investigated. Field pea was labeled *in situ* with $^{15}$N under ambient and elevated (700 μmol mol$^{-1}$) [$CO_2$] in controlled environment glasshouse chambers. Barley was also labeled under the same conditions by addition of $^{15}$N-enriched urea to the soil. Wheat was subsequently grown to physiological maturity in soil containing either $^{15}$N-labeled field pea residues or $^{15}$N-labeled barley plus fertilizer $^{15}$N residues. Elevated [$CO_2$] increased the total biomass of field pea (21%) and N-fertilized barley (23%), but did not significantly affect the biomass of unfertilized barley. Elevated [$CO_2$] increased total biomass (11%) and grain yield (40%) of subsequent wheat crop regardless of rotation type in the first phase. Irrespective of [$CO_2$], the grain yield and total N uptake by wheat following field pea were 24% and 11%, respectively, higher than those of the wheat following N-fertilized barley. The residual N contribution from field pea to wheat was 20% under ambient [$CO_2$], but dropped to 11% under elevated [$CO_2$], while that from fertilizer did not differ significantly between ambient [$CO_2$] (4%) and elevated [$CO_2$] (5%).

**Key Words**

Elevated [$CO_2$], $^{15}$N labeling, below-ground legume N, residual legume N, residual fertilizer N

## Introduction

The concentration of atmospheric carbon dioxide ($CO_2$) has increased from 280 μmol mol$^{-1}$ in 1800 to around 390 μmol mol$^{-1}$ now, and is expected to reach about 550 μmol mol$^{-1}$ in 2050 (Houghton *et al*. 2001). Total N uptake by crops and N removal in grain also generally increase under elevated [$CO_2$] (Kimball *et al*. 2002; Miyagi *et al*. 2007), and this increase in N demand in cropping systems would be expected to gradually reduce soil N reserves unless replenished.

Legume or fertilizer residues can contribute to subsequent N supply of crops (Ladd and Amato 1986), and alleviate N limitation under ambient [$CO_2$]. While elevated [$CO_2$] may affect the quantity, quality and decomposition rate of crop residues (Torbert *et al*. 2000; Kimball *et al*. 2002), the availability of residual legume and fertilizer N to subsequent crops under elevated [$CO_2$] remains unclear. Such information is critical for N management practice for soil N replenishment under elevated [$CO_2$]. The objective of this study was to investigate the interactive effects of elevated [$CO_2$] and residual N (legume or fertilizer N) on subsequent wheat growth and N uptake.

## Materials and Methods

### *Glasshouse chambers*

This study was conducted in potted soil in a set of four naturally light glasshouse chambers (3.1 m long ´ 2.4 m wide ´ 2.6 m high) located at the Department of Primary Industries Grains Innovation Park in Horsham (36*°43'S, 142°10'E*), Victoria, Australia. Two glasshouse chambers had ambient [$CO_2$] (390 μmol mol$^{-1}$), and two had elevated [$CO_2$] (700 μmol mol$^{-1}$). The average day/night temperature of the chamber throughout the growing season was 24°C/21°C.

### *Experimental design*

The effect of elevated [$CO_2$] on the contribution of residual legume N and residual fertilizer N from a previous barley crop to subsequent wheat crops was investigated in pots under three different 2-phase rotations, viz. field pea-wheat, N-fertilized barley-wheat, and barley (no fertilizer)-wheat (control). In rotation phase 1 we determined the effect of elevated [$CO_2$] on the accumulation of legume N and the partitioning of N derived from fertilizer in the crop-soil system. In rotation phase 2 we examined the recovery of residual legume N and fertilizer N by a following wheat crop under ambient or elevated [$CO_2$]. The experimental design was two $CO_2$ concentrations ´ three rotation types ´ two rotation phases, with four replications, totaling 48 pots (PVC tubes, 15 cm diameter, 45 cm long, sealed at the bottom). The PVC pots within each chamber were completely randomized.

*Soil and PVC pot preparation*

A Vertosol soil (Isbell 1996) was collected from the plough layer (0–20 cm) of an undisturbed area 8 km south-west of Horsham. The soil had a pH (soil:water ratio of 1:5) of 8.10, and contained 1.10% organic C, 0.12% total N, 2.0 mg ammonium-N $kg^{-1}$, 4.1 mg nitrate-N $kg^{-1}$, 7 mg Colwell P $kg^{-1}$, 630 mg available K $kg^{-1}$ and 37% clay. The soil was air dried, crushed into < 1 cm fractions, and mixed thoroughly.The 48 PVC pots were filled with 7.4 kg (dry weight) of the Vertosol. A basal nutrient application of 20 kg P $ha^{-1}$, 2.5 kg Cu $ha^{-1}$and 5 kg Zn $ha^{-1}$ was added to the top 2.0 kg portion, and the contents of this portion were thoroughly mixed.

*Rotation phase 1*

Field pea (*Pisum sativum* L. cv. Kaspa) plants (two per pot) were labeled with $^{15}N$ urea (3 mL, 0.5% w/w urea, 98.26 atom% $^{15}N$) at growth stages of 12$^{th}$, 16$^{th}$ and 20$^{th}$ nodes after emergence. The tip (2 mm) of two tendrils per field pea plant was cut off and the remainder of the tendrils was immersed in the labeling solution in sealed ziplock bags (4 cm by 6 cm) for 24 hours, by which time the solution was mostly absorbed (de Graaff *et al*. 2007). Barley (*Hordeum vulgare* L. cv. Gairdner) plants (four per pot) were grown in the PVC pots of the barley-wheat rotation. $^{15}N$-labeled urea (10.22 atom% $^{15}N$) was applied to the PVC pots at 50 or 0 kg N $ha^{-1}$ as a band with the seeds to minimize losses by volatilization.All the pots were watered with reverse osmosis water to constant weight (80% of field capacity) every two to three days.

*Rotation phase 2*

The dried shoot material obtained from the harvest of the first phase was cut into segments of 2–3 cm length, and added to the 24 PVC pots that had not been destructively harvested. The soil and root (0–10 cm) was lightly 'ploughed' to incorporate the residue material into the top 10 cm of soil to simulate field practice. Reverse osmosis water was added to each pot to constant weight (80% of field capacity) every week for one month before sowing of the subsequent wheat crop. After a wetting / drying cycle of one month, wheat (*Triticum aestivum* L. cv. Yitpi) plants (four per pot) were grown in the soil to maturity.

*Chemical analysis and $^{15}N$ calculations*

The finely ground plant and soil samples in both phases were analyzed for total C, total N and $\delta^{15}N$ values by isotope ratio mass spectrometry.Total below-ground legume N was calculated as described in Khan *et al*. (2002). The percentages of $^{15}N$ applied that were recovered in crop and soil were calculated according to Hauck and Bremner (1976). The percentage of residual field pea N or fertilizer N in the first phase recovered by the subsequent wheat ($\%N_{wheat}$) was calculated according to McNeill (2001).

*Statistical analysis*

Data were analyzed with MINITAB 16 statistical package using a general linear model analysis of variance with a level of significance of *p<0.05 unless otherwise stated*.

## Results and Discussion

*Rotation phase 1*

Elevated [$CO_2$] increased ($p< 0.001$) the total biomass of field pea and N-fertilized barley by 21% and 23%, respectively, but the [$CO_2$]-induced increase (9%) in the total biomass of barley receiving no N fertilizer was not significant (Table 1). A similar N-dependent growth response of barley to elevated [$CO_2$] was also observed by Thompson and Woodward (1994) and Fangmeier *et al*. (2000). In our study, the N-dependent effect of growth response to elevated [$CO_2$] was associated with plant N uptake. Increasing [$CO_2$] resulted in a 25% increase in total N uptake by field pea ($p< 0.001$), but had no effect on total N uptake by N-fertilized barley and reduced that of unfertilized barley by 10% ($p< 0.001$) (Table 1).The reduction in plant N uptake by unfertilized barley under elevated [$CO_2$] indicates that soil N availability was insufficient to meet plant demand under elevated [$CO_2$]. A declining N availability was shown to eliminate the $[CO_2]$-induced enhancement in leaf area index of cereal (Kim *et al*. 2003), thereby restricting the $CO_2$ fertilization effect on plant growth. In contrast, we found that this limitation of $CO_2$ fertilization effect on growth did not occur when N-fertilizer was applied to barley, or for field pea, which could source its N via $N_2$ fixation.

**Table 1: The effect of elevated [CO2] on total biomass, total plant N and residue C:N ratio of field pea, N-fertilized barley and unfertilized barley. Values are means of the four replicates for each treatment.**

| | $^{15}$N-fed field pea | | $^{15}$N-fertilized barley | | unfertilized barley | | $[CO_2]$ | Species | $[CO_2]$ ´ species |
|---|---|---|---|---|---|---|---|---|---|
| | Ambient $[CO_2]$ | Elevated $[CO_2]$ | Ambient $[CO_2]$ | Elevated $[CO_2]$ | Ambient $[CO_2]$ | Elevated $[CO_2]$ | | | |
| Total biomass (g core$^{-1}$) | 16.9 | 20.4 | 18.2 | 22.3 | 13.3 | 14.5 | *** | *** | ** |
| Total plant N (mg N core$^{-1}$) | 421.3 | 525.5 | 146.8 | 151.6 | 90.2 | 81.3 | ** | *** | *** |
| C:N ratio | 44.1 | 51.9 | 114.1 | 135.4 | 129.3 | 127.1 | * | *** | 0.08 |

Significant effects are indicated as ***$p< 0.001$, **$p< 0.01$ and *$p< 0.05$

*Rotation phase 2*

The grain yield and shoot biomass of wheat following field pea were 24% ($p< 0.001$) and 21% greater ($p< 0.001$), respectively, than those of wheat following N-fertilized barley, regardless of $[CO_2]$ (Fig. 1a). These results suggest that N fertilizer application does not have a substantial residual effect (Ladd and Amato 1986; Ladha *et al*. 2005), whereas that derived from legume residue N is a more accessible N source for subsequent crops, especially over the longer term (Ladd and Amato 1986). The supply of residual N to the subsequent crop depends on various biotic and abiotic factors including the quantity and quality of residue, and the interaction between plants and microbes under elevated $[CO_2]$ (Reich *et al*. 2006).Elevated $[CO_2]$ increased the amount of residues produced from the preceding field pea resulting in a greater mass of N being present for the following crop (Table 2), however this residue also had a greater C:N ratio (Table 1) favouring immobilization. This suggests that soil N availability to the subsequent wheat crop was reduced under elevated $[CO_2]$ which partly explains why whole plant N uptake of the subsequent wheat was not higher under elevated $[CO_2]$ (Fig. 1b). Thus manipulation of post legume N management at higher $[CO_2]$ may produce substantial agronomic gains and warrants future research.

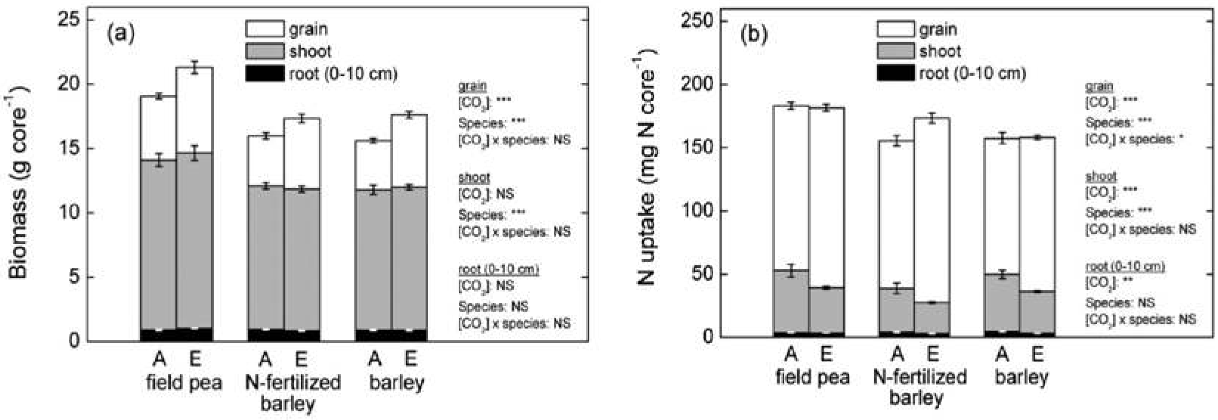


**Figure 1. Biomass (a) and N uptake (b) in different wheat parts following field pea, N-fertilized barley, and unfertilized barley under ambient $[CO_2]$ (A) and elevated $[CO_2]$ (E) in rotation phase 2. Values are means of the four replicates for each treatment. Vertical bars indicate standard errors. Significant effects are indicated as *$p < 0.05$, **$p< 0.01$ and ***$p< 0.001$. NS, not significant**

The contribution of field pea N to subsequent wheat N (11–20%) was within the range of the recovery of N contained in grain legume residues (2–27%) by various first succeeding crops (Fillery 2001). Although reduced under elevated $[CO_2]$, the contribution of field pea residue N to subsequent wheat (11%) was significantly greater than that of fertilizer residue N (5%) in our study(Table 2). While soil microbes prefer higher qualitysubstrates (Kuzyakov 2002), the lower C:N ratio of the field pea residues than that of barley residues may explain the difference in residual contribution between field pea N and fertilizer N.

Unaffected by elevated [$CO_2$], the recovery of residual fertilizer N by wheat in the present study was within the range of the generally low fertilizer N recovery (2–5%) in the first succeeding crop under various N application methods and residue management practices reported by Ladha *et al*. (2005). These results suggest that N uptake from legume residue was not enhanced under elevated [$CO_2$] even though biomass was. Therefore manipulation of post legume N management may yield significant agronomic gains (yield and protein) under future $CO_2$ environments.

**Table 2: The amount of 15N excess and percent recovery of residue N in various parts of wheat following field pea and N-fertilized barley.Values are means of the four replicates for each treatment.**

| | $^{15}$N-fed field pea | | $^{15}$N-fertilized barley | | [$CO_2$] | Species | [$CO_2$] ´ species |
|---|---|---|---|---|---|---|---|
| | Ambient [$CO_2$] | Elevated [$CO_2$] | Ambient [$CO_2$] | Elevated [$CO_2$] | | | |
| Total excess $^{15}$N added from phase 1 (mg) | 9.30 | 12.33 | 4.01 | 3.75 | 0.08 | *** | * |
| Grain $^{15}$N (mg) | 1.37 | 1.05 | 0.11 | 0.15 | 0.07 | *** | * |
| Shoot $^{15}$N (mg) | 0.475 | 0.260 | 0.039 | 0.024 | * | *** | * |
| Root (0–10 cm)$^{15}$N (μg) | 48.9 | 37.8 | 6.4 | 5.0 | 0.06 | *** | NS |
| Recovery in above-ground parts (%) | 19.8 | 11.2 | 3.8 | 4.6 | ** | *** | *** |

Significant effects are indicated as ***$p$< 0.001, **$p$< 0.01 and *$p$< 0.05. NS, not significant

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# BMC Soil Carbon Model – A predictive tool for assessing the benefits of building soil carbon in the Upper Hunter Valley

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In January 2012, Bengalla Mining Company (BMC) commenced a four year project focussed on community engagement and improving land-use and land-use outcomes in the Upper Hunter Valley. The objectives of the project are to work with landholders to build soil carbon on their properties, measure and demonstrate the benefits of increased soil carbon via improved management practices and biochar application and produce an assessment model, the BMC soil carbon model. The model will allow landholders to predict the likely impact of particular management practices on soil carbon levels and equate these increases to potential productivity increases. Additionally, the carbon sequestered in the soil has the potential to be traded under the Carbon Farming Initiative (CFI); the likely* *process is incorporated in the model.*

*Productivity increases on regional properties arising from land management practices that increase soil carbon are not only of great benefit to individual landholders in terms of profitability and increased land value but are directly offsetting the loss of agricultural productivity in the region as a result of BMC's operations. Furthermore, BMC's requirement to deal with GHG emissions liabilities under the Clean Energy Future legislation is aided by the potential 'local' market for soil carbon sequestration and expected abatement of emissions of nitrous oxide due to fertiliser use efficiencies as a result of biochar application.*

**No process (methodology) currently approved.*

# Long-term prescribed burning does not affect the abundance of denitrifier communities in a wet sclerophyll forest of Australia

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### Abstract

A 35 year old repeated prescribed burning trial, with three treatments (no burning as control, 2 yearly burning and 4 yearly burning), was selected to explore how prescribed burning affects the $N_2O$ flux, key soil properties (inorganic N, dissolved organic C and N, pH), N-associated functional gene abundance and their interactions. Overall, our results indicate that biennial burning reduced the microbial biomass significantly but did not change the abundance of genes of denitrifying communities, whereas quadrennial burning did not affect either biomass or gene abundance, which reflected the resilience of microbial community.

### Key Words

Prescribed burning; $N_2O$ flux; denitrification; microbial functional genes; real-time PCR

### Introduction

Denitrification is a microbiological process in which nitrate ($NO_3^-$) or nitrite ($NO_2^-$) is reduced to gaseous nitric oxide (NO), nitrous oxide ($N_2O$), or molecular nitrogen ($N_2$) under oxygen-limited conditions (Zumft 1997). Denitrification is of great importance since it can lead to significant N losses and greenhouse gas (i.e. $N_2O$) emission (Nakicenovic and Swart 2000). There is a large diversity of bacteria, archaea, and fungi involved in denitrification, and their abundance and composition are likely to be affected by different environmental changes. Fire is one of the key drivers of dynamics of diversity and function of terrestrial ecosystems (Orians and Milewski 2007). Overall, fires immediately lead to the loss of C and N as gases and particulates into the atmosphere from the ecosystem (e.g. Carter and Foster 2004; Galang *et al.* 2010). With respect to the $N_2O$ flux, it is generally very low and not always significantly affected by prescribed burning (Livesley *et al.* 2011; Dannenmann *et al.* 2011). Fires can lead to a drastic reduction in soil microbial biomass in the short term and cause a shift in bacterial and fungal communities in forest soils (e.g. Pietikäinen *et al.* 2000). Nevertheless, the understanding of the impact of long-term repeated prescribed burning on the interactive links of N-cycling associated microbial communities, soil environmental factor and $N_2O$ flux are still incomplete.

### Material and Methods

The research site is located in Peachester State Forest, southeast Queensland (26º50'S, 152º53'E) and was described in detail by Guinto *et al.* (1999). The prescribed burning trials were established in 1972 and consist of three treatments: (1) 2 yearly burning (burning every 2 years, 2YB), (2) 4 yearly burning (burning every 4 years, 4YB) and (3) no burning (CK), giving a total of 12 plots, 4 of each treatment. These plots were arranged in a complete randomized manner. Soil samples were taken in January and April 2011, respectively. For each sampling, approximately 10 cores of surface soils (0-10 cm) were collected using a corer (7 cm in diameter) from each plot and combined to produce one composite sample. Soils were sieved (4 mm) and kept at 4ºC prior to analysis. Subsamples for molecular analyses were sieved (2 mm) and stored at -80°C.

Soil properties were determined according to the methods described by Chen *et al*. (2004). DEA was analyzed by the method of Alef and Nannipieri (1995). $N_2O$ gas fluxes were determined using nonvented static PVC chambers consisting of a permanent base and a removable lid with a rubber septum for gas sampling. The base was inserted into soil by 10 cm two months before the gas sampling to minimize the effects of disturbance. Gas samples (26 ml) were collected using gas-tight syringes at around 9-10 am on 25/01/2011, 09/03/2011, 12/04/2011, 16/05/2011, 15/06/2011 and 12/07/2011 and analyzed using gas chromatography (Varian Inc., Middelburgh, Netherlands). Soil DNA extraction was conducted with the MoBio Powersoil™ DNA Isolation Kit (Carlsbad, USA).Quantitative PCR analysis of functional genes was carried out in an Effendorf Mastercycler® (Effendorf, Hamburg, Germany) ep realplex real-time PCR system in triplicate. The 20 µL PCR mixture contained 10 µL of SYBR green PCR Master Mix (Takara SYBR® Premix Ex Taq™), 0.4 µL of each primer (10 µM) and 12.5 ng of DNA. The primers and thermal conditions for all the functional genes are described in Table 1.

## Results

*Soil basic chemical and biological parameters*

Both 2YB and 4YB treatments significantly increased soil pH compared with the CK in both January and April (Table 2). The 2YB decreased soil TC and TN contents compared with the 4YB and the CK treatments at both sampling times (Table 2), while there was no significant difference between the CK and the 4YB treatments. The prescribed burning did not affect concentrations of $NH_4^+$-N at both sampling times, while concentrations of $NO_3^-$-N were significantly lower in 2YB treatments (Table 2). Concentrations of DOC, TSN and MBC and MBN were significantly lower in the 2YB treatments than in both the 4YB and CK treatments.

*Soil $N_2O$ flux and denitrification enzyme activity (DEA)*

*In situ* $N_2O$ fluxes ranged from 0 to 8.8 g $N_2O$-N $ha^{-1}$ $h^{-1}$ (Figure 1a) and it varied from month to month, with the highest rate observed in April and the lowest in July (Figure 1a). Overall, the 2YB had lower $N_2O$ fluxes than the CK treatments, but these differences were significant only in January and June. DEA, expressed as overall denitrification production - $N_2O$, was generally lower in the 2YB compared with the CK and 4YB treatments (Figure 1b).

*Abundance of N-associated functional genes*

In January, no significant difference was observed in most of the genes between burning treatments, except for archaeal *amo*A, *nos*Z and *nir*K genes (Figure 2a). The copy numbers of archaeal *amo*A, *nos*Z and *nir*K genes were significantly lower in the 2YB compared with the CK and the 4YB treatments (Figure 2a). The abundance of all other functional genes (except for *nif*H and *nir*S) and 16S rRNA gene tend to be lower in the 2YB compared with the CK and the 4YB treatments (Figure 2a), but the differences were not statistically significant. Similar trends in the gene copy number were observed across different functional genes and different burning treatments in April (Figure 2b).

## Conclusions

Two and 4 yearly burning did not change the abundance of genes of denitrifying communities despite 2 yearly burning significantly reduced microbial biomass, indicating the resilience of microbial community in response to repeated prescribed burning.

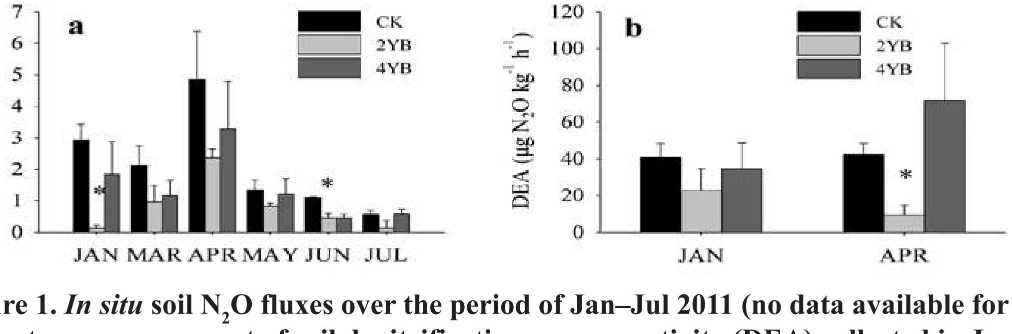


**Figure 1. *In situ* soil $N_2O$ fluxes over the period of Jan–Jul 2011 (no data available for Feb) (a) and laboratory measurement of soil denitrification enzyme activity (DEA) collected in Jan and Apr 2011(b) under different prescribed burning regimes at the Peachester site. Mean values ± standard error are shown (n= 12). The star above the bars indicates significant differences between different treatments (*$P < 0.05$*).**

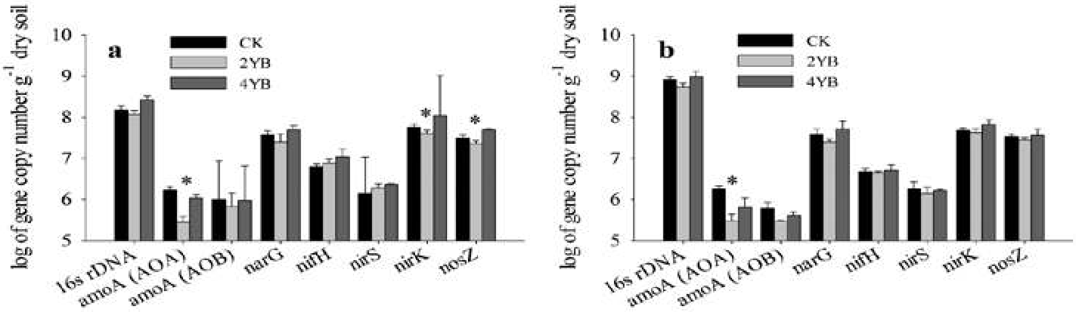


**Figure 2. Abundances of the 16S rRNA, archaeal *amo*A, bacterial *amo*A, *nar*G, *nif*H, *nir*S, *nir*K, and *nos*Z gene in soils under different burning regimes in Jan (a) and Apr (b), expressed as the number of gene copies $g^{-1}$ dry soil. The star above the bars indicates significant differences between different treatments (*P* < 0.05).**

**Table 1: Thermal conditions and primers of qPCR reactions used in this study**

| Target gene | Primer name | Reaction conditions | | | | |
|---|---|---|---|---|---|---|
| | | Denaturation time at 95˚C (s) | Annealing time and temperature | Elongation time and temperature | Number of cycles | qPCR efficiency |
| 16S rRNA | 338F, 518R | 15 | 55˚C,15s | 72˚C,30s | 30 | 0.91 |
| *nif*H | POI-F, POI-R | 15 | 54˚C,30s | 80˚C,30s | 35 | 0.82 |
| *amo*A(AOA) | CrenamoA23F,A616R | 15 | 55˚C,30s | 72˚C,45s | 40 | 1.00 |
| *amo*A(AOB) | amoA1F, amoA2R | 15 | 55˚C,30s | 72˚C,45s | 40 | 1.05 |
| *nar*G | narGG-R, narGG-F | 15 | 58˚C,30s | 80˚C,30s | 35 | 0.89 |
| *nir*K | nirK876, nirK1040 | 15 | 58˚C,30s | 80˚C,30s | 35 | 1.07 |
| *nir*S | nirS4QF, nirS6QR | 15 | 58˚C,30s | 80˚C,30s | 35 | 0.77 |
| *nos*Z | nosZ2F, nosZ2R | 15 | 60˚C,30s | 80˚C,30s | 35 | 0.77 |

**Table 2: Soil parameters as affected by different prescribed burning regimes**

| Parameter | January | | | April | | |
|---|---|---|---|---|---|---|
| | CK | 2YB | 4YB | CK | 2YB | 4YB |
| pH | 4.6 (a) | 5.50 (b) | 5.1 (ab) | 4.56 (a) | 5.20 (b) | 5.07 (b) |
| EC (µs $cm^{-1}$) | 39.2 (a) | 19.7 (b) | 29.5 (ab) | 29.6 (a) | 15.9 (b) | 23.4 (ab) |
| Moisture (%) | 36.9 (a) | 21.6 (b) | 36.8 (a) | 37.2 (a) | 23.3 (b) | 37.6 (a) |
| Total C (%) | 6.28 (a) | 3.53 (b) | 6.55 (a) | 6.23 (a) | 3.70 (b) | 6.83 (a) |
| Total N (%) | 0.250(a) | 0.10 (b) | 0.30(a) | 0.30 (a) | 0.13 (b) | 0.33 (a) |
| $NH_4^+$ (mg N $kg^{-1}$) | 8.02 (a) | 7.44 (a) | 7.61 (a) | 6.59 (a) | 7.35 (a) | 9.57 (a) |
| $NO_3^-$ (mg N $kg^{-1}$) | 7.26 (a) | 0.62 (b) | 4.18 (ab) | 6.75 (a) | 0.50 (b) | 2.94 (ab) |
| $DOC_{hw}$ (mg C $kg^{-1}$) | 730 (a) | 409 (b) | 625 (a) | 555 (a) | 481 (b) | 486 (ab) |
| $TSN_{hw}$ (mg C $kg^{-1}$) | 83.5 (a) | 41.3 (b) | 68.1 (a) | 74.8 (a) | 35.2 (b) | 55.6 (ab) |
| DOC:TSN ratio | 8.8 (a) | 10.0 (a) | 9.4 (a) | 8.4 (a) | 9.5 (a) | 9.1 (a) |
| MBC (mg C $kg^{-1}$) | 499 (a) | 290 (b) | 518(a) | 426 (a) | 264 (b) | 422 (a) |
| MBN (mg N $kg^{-1}$) | 71.8 (a) | 48.4 (a) | 70.5(a) | 67.8 (a) | 49.1 (a) | 65.9 (a) |
| Microbial C:N ratio | 6.9 (ab) | 6.2 (b) | 7.4 (a) | 6.3 (a) | 5.4 (a) | 6.5 (a) |

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# Novel instrumentation for CN determination in Soils – the vario MAX cube

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Organic elemental analysis usually requires complete conversion of the sample of interest into the gas phase at approx. 1200 K. Afterwards the mixture of combustion gases is chemically converted to defined species, separated to single components and most commonly subjected to a thermoconductivity detector (TCD). Carbon and nitrogen determination in soils according to the Dumas method are hampered by large amounts of ash residues causing frequent maintenance intervals. The well-established *"vario MAX"* already set standards by means of reusable steel crucibles as sample containers which are automatically inserted and taken out of the furnace via a patented, vertical robot arm. Consequently, ashes are also removed after combustion is completed.

With the newly designed *"vario MAX cube"* Elementar further advanced this proven technique. Benefits in comparison to the *"vario MAX"* are optimized chemicals and gas consumption, faster time of analysis, lower price per analysis, modern electronics and improved software. Experimental results will be discussed in full detail.

Special attention was given towards the decreasing availability (and thus an increase in price) of helium traditionally used as carrier gas in Dumas analysis. The *"vario MAX cube"* permits utilization of either argon or helium. Here, we will present a detailed comparison of data obtained on different soils based on helium or argon operation and discuss limitations of both approaches. All results gained with argon are in good agreement with those of helium. However, the lower thermoconductivity of argon in comparison to helium demanded development of a novel TCD with increased sensitivity.

# Effects of chemical properties of dissolved organic matter derived from tropical plantation tree species on soils sorption

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To better understand the sorption behaviour of dissolved organic matter (DOM) to tropical soils, chemical assays of DOM derived from litters of three tropical plantation tree species were conducted. The sorption of nine DOM samples which consist of three species; *Acacia mangium, Eucaryptus pellita and Pinus merkusii* divided into three decomposed stages on six soil samples; each two layers of sandy acrisol, clayey acrisol and ferralsol was studied in batch experiments. Chemical assays as pH, huminification index, 280nm molar absorptivity, $^{13}C$ NMR spectroscopy and hydrophobicity were conducted on DOM before and after sorption. Soil properties of pH, soil texture and organic carbon content did not vary and no difference in sorption amount among the soil types was observed. Maximum sorption capacity estimated by the modified Langmuir sorption isotherm on the same soil differed in species and decomposed stages. DOM from more decomposed litter was high in huminification index and sorbed more to soil than DOM from fresh litter. This was caused by the higher sorption affinity of more oxidized and polymerised DOM. Maximum sorption capacity decreased in the order of *P. merkusii*, *A. mangium* and *E. pellita* and the reasons of these differences were discussed based on the results of structural characteristics of DOM using $^{13}C$ NMR spectroscopy and the XAD fractionation method, which suggested that DOM with rich hydrophobic fraction such as lipidsubstances and lignin-derived aromatic substances sorb more to tropical soils.

# Investigation of compost and plant associated microbes for use in mine spoil rehabilitation

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Open cut coal mining is a significant temporary land use in the Hunter Valley of NSW, Australia. Mining companies are required to rehabilitate mined lands. In some cases, the rehabilitation objective is to establish a sustainable native ecosystem on these lands. To achieve this, both plant and soil systems must be successfully performing essential functions.

In many cases, soil resources for rehabilitation are minimal. This has led to the use of spoil (the parent rock material left after mining) and other alternative rehabilitation media. Mine spoils and often, post-mining soils are in a degraded state and contain little to no organic Carbon. Organic Carbon contributes to soil structure (or aggregation) thereby enabling root penetration, water filtration, gas exchange, nutrient availability and provision of habitat for soil flora and fauna. In addition, organic Carbon aids litter decomposition (by providing energy for microbes). Thus, the absence or impairment of these essential soil functions (due to low organic Carbon content in soils and spoils) may hamper rehabilitation efforts.

This research proposal utilises mine spoil amended with compost and microbes as a rehabilitation medium. The ameliorating properties of municipal waste compost are recognised but short-lived. However, if combined with plant-associated microbes (like mycorrhizae, and *Rhizobium*) and dark endophytic fungi, the potential may exist to boost soil organic Carbon levels. This would enable processes like litter decomposition, and spoil aggregation to occur. The developing system may then commence other essential functions, and become self-sustaining. It is proposed to measure the effects of municipal waste compost, microbial inoculation and native plants on spoil organic Carbon levels, root litter decomposition and spoil aggregation through field and laboratory experimentation.

# Climate, soil order and land use influences on soil carbon in Tasmania

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Tasmania has a diverse range of soil types due to variations in landforms, landscape history, lithology and climatic setting. However, land development and clearing for agricultural uses has occurred mostly on the more versatile soil types and gentler slopes. The objectives of this research were to determine the effects of both environmental parameters and land use history on soil organic carbon concentrations and carbon stocks in agricultural soils of Tasmania. The 282 Sites were selected on Dermosols, Ferrosols, Vertosols and a group referred to as texture contrast soils, dominantly Chromosols and Sodosols.

The land uses were “Pasture”, which required sites to have had eight or more years of pasture grown during the preceding ten years, and “Cropping” which required sites to have had five or more years of cropping during the preceding ten years. A strong relationship was found between total organic carbon and both mean annual rainfall over the past five years and long term (30 year) rainfall under pasture. The relationship was stronger with mean summer rainfall (November - March) than either mean winter and spring rainfall (April - October) or total mean annual rainfall. The relationship for cropping sites was weak or non-significant. Carbon stocks in the 0 – 30 cm depth were greater under pasture than under cropping on all soil orders sampled. Clay rich soils contained the greatest carbon stocks under pasture with Ferrosols containing a mean of 150 MgC ha-1, Vertosols 107 MgC $ha^{-1}$, and Dermosols 102 MgC ha-1. Texture contrast soils under pasture contained a mean of 67 MgC $ha^{-1}$.

# Encouraging microbial soil C sinks – important points for land managers

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Recently the IPCC have provided resounding evidence that atmospheric methane ($CH_4$) concentrations have been increasing due to anthropogenic sources and are currently ~ 150 % greater than at the start of the Industrial Revolution. There are many components of the global C cycle one of which is soil methane ($CH_4$) oxidation, performed by the enzymatic conversion of $CH_4$ to $CO_2$ and biomass by methanotrophs. These soil bacteria thereby create a sink for $CH_4$, as on a molecule for molecule basis its GWP is 21 times greater than that of $CO_2$, hence the process is a less recognised means of C sequestration. Globally the soil $CH_4$ sink is considered to be approximately 30 Tg $y^{-1}$ and studies have shown the soil $CH_4$ oxidation sink can offset up to 8 % of the increase in New Zealand (NZ) agricultural emissions since 1990.

Methanotrophs are generally found in most soils and with the right understanding by land managers their effectiveness as a means of sequestering C can be enhanced. The $CH_4$ oxidation process is generally found in the top 100 – 150 mm of soils and is predominantly regulated by soil water content and to a lesser extent temperature. They are very sensitive to disturbance which can be physical i.e. changes in soil structure, and chemical i.e. by the addition of (N) salts and any alterations to the native soil pH as well as vegetation cover. Some guidelines to encourage greater $CH_4$ oxidation rates in soils in NZ will be presented.

# Soil type and landform are useful indicators of soil organic carbon in the freehold northern Mallee of Victoria

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We investigated the usefulness of soil type and landform in explaining the distribution of soil organic carbon (SOC) in the cropping area of the northern Mallee region of Victoria. Relationships between SOC, landform and soil type (Australian Soil Classification) were examined to test the following hypotheses, that:

(i) landform at a land element level (e.g. dune v swale) can explain major differences in SOC.

(ii) landform morphological unit (e.g. slope position) does not have sufficient data to explain the spatial variation in this study area.

(iii) finer classification of soil type (i.e. Sub-Order, Great Group and Sub-Group) better represents the distribution of SOC than Order alone.

As part of the national Soil Carbon Research Program (SCaRP), around 100 sites were selected according to two dominant soil types i.e. Calcarosols and Tenosols, and encompassing a range of topographic positions. Relationships between SOC values and topographic position and field texture were assessed. We discuss SOC distribution in terms of both landscape and soil type and the degree of resolution of these data to predict the distribution of SOC in this landscape.

# Estimating turnover times of soil organic carbon fractions based on natural C-14 abundance

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The knowledge of turnover times of different well defined and meaningful soil organic carbon (SOC) fractions is valuable to better understand SOC dynamics in contrasting land use systems. The aim of the present study was to determine turnover times of SOC fractions as separated by a combination of physical and chemical methods. Soils from three different land uses (woodland, native grassland and cropped land) and two soil types (Chromosol and Vertosol) were collected from northern cropping region of New South Wales, Australia. Each soil sample was first physically fractionated into particulate organic carbon (POC) (>53 µm) and silt+clay (s+c) (<53 µm) fractions by dispersion with 5% sodium hexametaphosphate followed by sieving. Second, the chemically resistant SOC fraction or inert organic matter (IOM) was separated from s+c fraction by oxidation with 6% sodium hypochlorite (NaOCl). Natural C-14 abundances in the SOC fractions were measured by accelerator mass spectrometer (AMS).The POC fraction had the shorter turnover times (10 to ~370 yr) compared to those of other SOC fractions. The s+c fraction (excluding IOM fraction) had intermediate turnover times (230 to ~3100 yr), with radiocarbon ages ranging from modern to ~1100 yr BP (before present). The IOM had the oldest carbon compared to that of other SOC fractions, with radio carbon ages ranging from modern to ~1950 yr BP. Organic carbon in almost all the fractions in cropped soils were older compared to native woodland and grassland. In cropped soils, turnover times of different SOC fractions were longer compared to woodland and native grassland.

# Effects of repeated low-intensity fire on soil carbon fractions in a mixed eucalypt forest in south-eastern Australia

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Prescribed fire or planned burning has been practiced on Victorian public lands to reduce the risk of wildfires and for silvicultural purposes for the last 50 years. The impact of repeated prescribed fires on soil carbon (C), nitrogen (N) and organic matter fractions is being evaluated at five Fire Effects Study Areas in the Wombat State Forest, north-west of Melbourne. Low intensity fire treatments have now been applied since 1985 in spring and autumn at three year and ten year intervals to long-unburnt forest.

After 13 years there was a clear decline in surface soil (0-2 and 2-5 cm) Total C and Total N in the 3-year burn interval treatments, but no significant decrease in the 10-year interval burnt forests (Hopmans 2003). Additional measurements on these soils found that Particulate Organic Carbon (POC, 50-2000 µm) was about 90% of Total C regardless of treatment.

Soils from the study were re-sampled in 2012, after 26 years of burning treatments. Results will be presented from size fractionation of these soils (wet sieving <50, 50-2000 µm) and analysis of Total C and Char C content that will be used to initialize RothC model estimates of soil carbon cycling (Skjemstad *et al*. 2004). Proposed methodologies for analysis of Char C include acid digestion, pyrolysis – gas chromatography mass spectrometry. Mid infra-red spectra will be collected for all soils to assist with interpretation of changes in soil organic matter functional groups.

# The effect of aged biochar on surface charge characteristics and adsorption behaviour of Cd and P in two contrasting soils

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Biochars are considered to develop negative charge from oxidation with ageing in soils. However, this finding is based on limited analysis of black carbon in soils. In the present study, surface charge characteristics, and Cd and P adsorptive behaviour of an aged biochar were investigated in two variable charged soils, a Ferrosol and a Tenosol, by comparing (i) unamended soil; (ii) soils amended with fresh biochar (450 °C) and (iii) soil amended with biochar (450 °C) and aged for 12 months. The biochar was applied at a rate of 2% w/w in both soils. Surface charge characteristics were assessed using the "index" ion adsorption method, with a LiCl electrolyte. Adsorption studies of Cd and P were conducted using batch experiments. In contrast to previous studies, the results provided no evidence oaf an increase in CEC as a consequence of biochar ageing. The Cd adsorption study demonstrated an increase in Cd adsorption in the presence of aged biochar, where there was a 1.5-fold increase in Cd adsorbed in both soils, at the highest equilibrium Cd concentration. However, with regards to P adsorption, biochar ageing led to a decrease in P adsorbed in both soil systems. In the Ferrosol amended with aged biochar, there was a 6% reduction in P adsorbed at the highest equilibrium P concentration. The Tenosol had an extremely low P adsorption capacity in comparison to the Ferrosol. Overall, this study has highlighted that the adsorptive behaviour of aged biochar varies, depending on the nature of adsorbing ion and soil type.

# Chemical status of Si and Al in soil profiles of Andosols in New Zealand and Japan: Determination of allophanic constituents by solid-state NMR

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Solid-state nuclear magnetic resonance (NMR) spectroscopy is a powerful tool for characterizing Si and Al in soils. Especially, for soils derived from volcanic ejecta, Si NMR can identify allophanic constituents, and Al NMR spectra is a good indicator for the degree of chemical weathering (Hiradate, 2004). In the present study, we analysed some soil samples from Japan and New Zealand by using solid-state NMR. For a pumice soil from Waihora, New Zealand, the form of Al is found to be mostly tetrahedral, indicating that pumice included in this soil has not been weathered much. The Si NMR spectra indicated that allophanic constituents have not been formed in the soil. A similar result was obtained for a Japanese tephra horizon, which erupted in AD 915 (Towada A) and collected from San-nohe, Aomori, Japan. For another soil from Ngahinapouri, New Zealand, it was found that around 30% of Al was transformed into the tetrahedral form. From the Si NMR spectra, no evidence was observed for the presence of allophanic constituents, although the Ngahinapouri soil has been considered to be allophanic. It has been suggested that some non-allophanic Si fractions might be dissolved by acid-oxalate, resulting in erroneous amount of allophanic materials in soils (Hiradate et al., 2006). It could be possible that the Ngahinapouri soil is such a case. Thus, solid-state NMR can provide useful information about soils.

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# Pedology around a 6700 year old Neolithic ring ditch system in Germany

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The cultural landscape in Europe is the product of many different human activities and natural processes. Geoarchaeology tries to reconstruct this cultural landscape using geoarchives of different times. Soils can be used as such archives but there is a lack of ancient in situ soils throughout Europe.

During archaeological excavations in eastern Bavaria late 2008 a new Neolithic ring ditch system was discovered which dates back to 4700 BC. Situated in the loess belt of eastern Bavaria calcic luvisols form the native soils of the region. However, black soils from older floodplains, the so-called Tschernitza, are described some hundred meters beside the excavation area. In addition black sediments formed by anthropogenic fire activity fill the pits of a nearby Neolithic settlement. Interestingly, the backfills of the contemporaneous ring ditch system differ. The 2 m wide and 2.5 m deep ditch functioned as an ideal trap for naturally eroded sediments. The backfills of the ditch indicate different phases of sedimentation. Furthermore, the analysis of thin sections and laboratory data (such as RFA, RDA, pedogenic oxides , magn. suszept., etc.) together with geophysical measurements point to the existence of a fully developed calcic luvisol around 6700 years ago. At the same time within some hundred meters of distance and under the same meso-scale climatic conditions black soils formed and black sediments were produced by the Neolithic settlers. We compare this situation with other sites known in Bavaria and present the meaning of our results for the landscape history.

# Cation exchange as influenced by the type of cations in different clay minerals and soils

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Cation exchange capacity (CEC) is a general indicator of soil/clay storage capacity for available positively-charged plant nutrients such as calcium, magnesium, potassium and sodium. The range and mean values of CEC for natural and soil clays differ when different cations are used in the exchange reactions. We examined 9 clays from natural deposits and 4 Australian soils with contracting mineralogy. Homoionic soil/clay samples were prepared by treating a portion of the soil/clay suspensions with a 1 M solution of the monovalent (K and Na) and a 0.5 M solution of the divalent (Ca and Mg) chlorides.

Comparing the CEC of monovalent cationic soil/clay samples, K-saturated samples have lower CEC's than Na-saturated with the exception of Wyoming and Texas smectite and smectitic Claremont soil. CEC of K-vermiculite is vastly lower than that of Na-vermiculite. This is probably because of the great affinity of the vermiculite interlayer for potassium. For the divalent cationic soil/clay samples, Ca-saturated samples have lower CEC's than Mg-saturated with the exception of illite, Texas, Otay and Wyoming smectites and illitic McLaren soil. Our data show that CEC depends on the cation used partly because of its effect on particle size. The changes in size observed correlate well with zeta potential measurements. The effect is much stronger in 2:1 clays (like montmorillonite and illite) than in 1:1 clays (like kaolinite and halloysite).

# Using vis–NIR spectra to translate between soil classification systems

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Visible–near infra red spectroscopy (vis–NIR) has been identified as a fast and cost-effective way to gather soil data and answer the soil data crisis. First only found in the laboratory and used on dried, sieved spectra, this technology is now moving into the field, thus providing soil scientist with the opportunity to collect a lot more data, at a small cost. VNIR spectra can be correlated to numerous chemical and physical soil parameters, which illustrates the staggering quantity of soil information that can be collected by a single spectra. In this study, we propose a method to use this quantitative, hyper-dimensional space as a conversion framework for soil data in one classification system (WRB soil classification) to another classification system (soil taxonomy classification).

Examples are presented using the GSSL, which is a global dataset, but conversion between Australian and New Zealand soil classification systems will also be presented.

# Mapping sub-surface acidity for targeted land and water management strategies on a coastal acid sulfate soils floodplain

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Sulfidic sediments and acid sulfate soils (ASS) commonly underlie coastal floodplains in eastern Australia. These soils are characterised by high spatial and temporal heterogeneity. On coastal floodplains, ASS are of moderate to high salinity, with salts derived mainly from either connate marine sources or oxidation of biogenic sulfides and the subsequent increases in soluble ions (e.g. $SO_4^{2-}$) and acidity that follow oxidation. Enhanced acidity also increases the mobilisation of pH-sensitive trace metals such as Fe, Al, Mn, Zn and Ni and contributes to increasing apparent salinity.

Accurate soil maps with high spatial resolution are required to develop appropriate management strategies for ASS and other soil types associated with low-lying coastal floodplains. However, collection of this data using standard soil sampling techniques is often time consuming, laborious and expensive.

Ground-based geophysics using electromagnetic induction (EMI) techniques have been used successfully and extensively in inland areas of Australia to rapidly and accurately map soils for salinity management and precision agriculture. This study presents results from an investigation of the use of EMI techniques as a proxy to map the surface and sub-surface spatial distribution of acid sulfate soils and associated acidic groundwater. Soils were sampled to calibrate the EMI survey and analysed for pH, EC, soluble and exchangeable salts and metals, particle size and total actual acidity (TAA).

The classes identified form sensible soil management zones across the study area related to defined geomorphic units. EMI data can then be used to build below-ground models to inform practical targeted management strategies on coastal floodplains to improve land and water quality outcomes.

# Measuring spatial variability of soil's plant available water capacity using electromagnetic induction

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## Abstract

Soil's plant available water capacity (PAWC) is an important yield determinant and knowledge of its spatial variability within a field is useful for site-specific management. However, field measurements are costly and time consuming. We measured spatial variability of apparent electrical conductivity ($EC_a$) using EM38 at the wet and dry soil profile to measure PAWC. The simple regression model showed significant relationship between $EC_a$ measured at the wet and dry profiles and profile volumetric moisture to 1.5m. Spatial variability of predicted PAWC with measured PAWC showed reasonably good agreement and areas with low PAWC matches reasonably well with high concentration of subsoil Cl and high surface soil exchangeable sodium.

## Introduction

Measuring PAWC requires measuring two limits; drained upper limit (DUL) i.e. soil moisture at field capacity and crop lower limit (CLL) i.e. the lower limit of plant water extraction (soil moisture after the crop harvest). PAWC measurements can be obtained either from field measurements (Dalgliesh and Foale 1998) or approximated under laboratory conditions. Field measurements of DUL involves wetting up an area of soil until it has reached saturation, allowing time for drainage whereas CLL involves erecting rain exclusion tent over a portion of the vigorously growing crop at the time of flowering and left in place until the crop has reached maturity. Laboratory measurements involve destructive sampling at sowing and harvest. Both the procedures are time-consuming and labour-intensive. Further, the spatial and temporal variability of soil water makes it difficult to measure soil water with accuracy at a point and sub-paddock scale. We used electromagnetic induction (EMI) because of its speed, ease of use, spatial coverage and its non-invasive nature. EMI has become an important tool to assess soil spatial variability, especially in site-specific management. The functional relationship between bulk $EC_a$ and soil volumetric water content (θ) can be described as (Cook and Williams 1998):

$$EC_a = EC_s + EC_w * \theta * \eta;$$

where $EC_w$ (dS/m) is the electrical conductivity of the soil solution, $EC_s$ is the solid phase contribution to $EC_a$, η is the tortuosity, which is determined by soil texture and moisture content. The relationship between $EC_a$ and θ appears to be very complex, however, other studies (e.g. Kachanowski *et al.* 1988; Huth and Poulton 2007) have shown a linear relationship existed between $EC_a$ and θ in non-saline soils. The objective of this study was to determine if such a linear calibration can be used to monitor changes in soil moisture to enable mapping spatial variability of PAWC at sub-paddock scale especially in soils with high concentrations of subsoil salts.

## Methods and Materials

Two EMI surveys were carried out on a 196 ha paddock in northern NSW (29.03o S, 149.81o E) with negligible topographic variation (< 1 % slope; figure not shown) (Dang *et al.* 2011) at wet profile (May 2009) and at dry profile after the harvest of barley crop (October 2009), using Geonics EM38® linked to data acquisition computer and dGPS. The dGPS equipment was mounted on a vehicle, with the sensor towed behind on a conveyor belt mat. The instrument was also kept covered in perspex box during data collection to protect it from direct sunlight (Abdu *et al.* 2007). Before collecting $EC_a$ data, EM38 was calibrated according to the manufacturer's specification (Dang *et al.* 2011). The EM38 was operated in both horizontal ($EC_H$) and vertical ($EC_V$) dipole mode, providing an effective measurement depth of approximately 0.75 m and 1.5 m, respectively. The $EC_a$ data were collected across field along 24-m wide transects at 1-s intervals, and travel speeds of 30-35 km/h. Seasonal variation in soil temperature was corrected using temperature correction factors calculated for a range of locations in eastern Australia (Huth and Poulton 2007). All position corrected $EC_a$ data were transformed into MGA94 Zone 55co-ordinate set and kriged to a 5 m x 5 m grid with the software Vesper to produce maps of soil $EC_a$ (Whelan *et al.* 2001). The kriged $EC_a$ data for each field was converted to raster and stratified into five classes according to the natural breaks in ArcGIS v. 9.3 and a minimum of five locations were selected within each class for soil sampling. For selected points, the $EC_a$ values were obtained using nearest neighbour interpolation.

Thirty soil samples were taken at each EMI survey using a 50-mm diameter tube and a hydraulic sampling rig. Soil samples were extruded onto a plastic liner and then sub-sampled into a surface interval (0.0–0.1 m), then successive 0.2-m intervals to 1.5 m depth and dried at 105$^0$C to constant weight for measuring gravimetric moisture. Separate samples were also obtained at ten selected locations and analysed for soil Cl and ESP (Rayment and Higginson 1992). To facilitate relationships between $EC_a$ and volumetric moisture, a weighted profile mean value was obtained for each core to 1.5 m (Hedley *et al.* 2004). To determine the relative contribution of soil properties to $EC_a$ with depth, four aggregated sets of data were created: (i) 0–0.1 m (surface soil), (ii) 0.1–0.5 m (shallow subsoil), (iii) 0.5–0.9 m (deep subsoil), and (iv) 0.9–1.5 m (very deep subsoil).

Plant-available water (PAW) at each location was obtained by subtracting volumetric moisture at harvest, representing crop lower limit (CLL) from volumetric soil moisture at the time of sowing, representing drained upper limit (DUL) using the method of Dalgliesh and Foale (1998). Briefly, for determining CLL, a rain-exclusion tent at two sites was erected over a portion (3m x 3m) of the vigorously growing barley crop, at the time of flowering, and was left in place until the crop reached maturity during 2004-05. DUL was determined by wetting up an area of soil until it reached saturation, allowing time for drainage, and then sampling for soil water content. Soil moisture at sowing and harvesting barley in 2009 was compared with measured DUL and CLL in 2004-05.

## Results and Discussion

### *Spatial variability of soil properties*

Across selected soil points within the paddock, average clay content was generally uniform at a given location as well as vertically averaged clay content was spatially uniform with a CV of 7%. Compared to clay content, average Cl concentrations increased with soil depth from 419 to 2725 mg/kg to a depth of 1.5 m, and vertically averaged Cl was more spatially variable with a CV of 12%. Average volumetric moisture content measured before sowing was generally higher and uniform in 0.1 to 1.5 m soil depth as compared to average volumetric moisture measured after the harvest of barley crop (Figures not shown).

### *Plant available water*

Soil moisture measured at sowing in 2009 and DUL measured in 2005 were not significantly different (Fig. 1). However, there were some differences in soil moisture at crop harvest in 2009 and measured CLL in 2005. The differences in CLL can be expected due to soil, environment and fertilizers applied (Ritchie 1981).

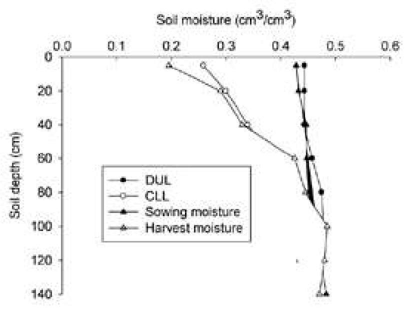


**Figure 1. Soil moisture at DUL, CLL, and at crop sowing and harvest.**

### *Relationships of soil $EC_a$ with measured soil properties*

Stepwise regression showed that soil moisture was generally the principal determinant of variation in $EC_a$, for the profile as a whole, and for the individual depth layers to 0.9 m (Table 1). In the very deep subsoil (0.9–1.5 m), $Cl^-$ was the principal determinant of the variation in the $EC_a$, followed by soil moisture. The coefficient of determination was highest at 0.1-0.5 m soil depth.

**Table 1. Step-wise multiple regressions relating soil properties and $EC_a$ measurements (SM, soil moisture; Cl, chloride concentration; CEC, cation exchange capacity)**

| Soil depth (m) | Regression equation; $EC_a$ (LL) | $R^2$ | RMSE | *P* |
|---|---|---|---|---|
| 0–1.5 | -744.3 + 1.76 SM | 0.87 | 20.2 | <0.0001 |
| 0–0.1 | -406.1 + 7.01 SM+ 3.23 CEC | 0.72 | 31.2 | <0.0001 |
| 0.1–0.5 | -482 + 4.0 SM + 0.06 | 0.98 | 8.3 | <0.0001 |
| 0.5–0.9 | -514 + 4.53 SM | 0.53 | 38.0 | <0.0001 |
| 0.9–1.5 | -711 + 0.03 + 4.95 SM | 0.80 | 41.8 | <0.0001 |

*Relationships between apparent electrical conductivity and volumetric soil moisture*

Measured soil water in 0-1.5 m depth at the wet and dry profiles had positive relationship with both profile average $EC_V$ and $EC_H$. The values of coefficient of determination between measured soil water and profile average $EC_V$ and $EC_H$ were substantially higher in the dry profile than the wet profile (Figure 2). This was possibly because the paddock had high concentration of subsoil Cl restricting the ability of roots to extract moisture in the subsoil (Dang *et al.* 2008) thereby increasing lower limit of crop's available soil water (Dang *et al.* 2006). The relationships between measured soil moisture to 1.5 m depth for both $EC_V$ and $EC_H$ were substantially improved by considering both the wet and dry profiles together (Figure 2a, 2b), however, the values of coefficient of determination was higher and RMSE were lower for $EC_V$ than $EC_H$. Therefore, only $EC_V$ model was used to estimate spatial variability of PAWC.

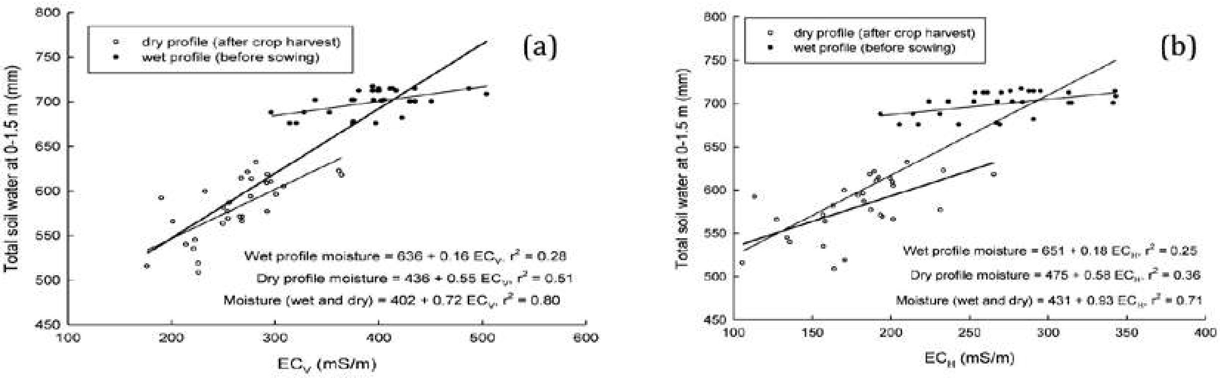


**Figure 2. Relationships between measured soil water at 0-1.5 m and average profile apparent electrical conductivity measured in (a) vertical dipole mode, and (b) horizontal dipole mode.**

*Spatial variability of PAWC and subsoil constraints*

Predicted PAWC in 0-1.5m soil depth based on $\theta_{1.5m} = a+b*(EC_V)$ model, showed reasonably good agreement with the measured PAWC in 0-1.5 m with most of the values were near 1:1 line (Figure 3).

A comparison of spatial variability of predicted PAWC (Figure 4a) with measured PAWC (Figure 4b) shows reasonably good agreement. Areas with low PAWC matches reasonably well with high concentration of subsoil Cl (Figure 4c) and high surface soil exchangeable sodium (Figure 4d). High Cl in the subsoil restricts the ability of the roots to extract moisture and nutrients from the subsoil, high ESP in surface soil results in soil crusting, water-logging, and poor germination (Dang *et al.* 2006).

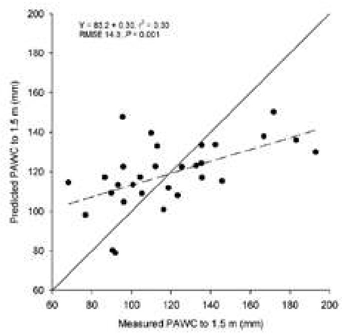


**Figure 3. Relationship between predicted and measured water content for the θ1.5m = a+b* (ECV) model. Solid line represents the 1:1 ratio.**

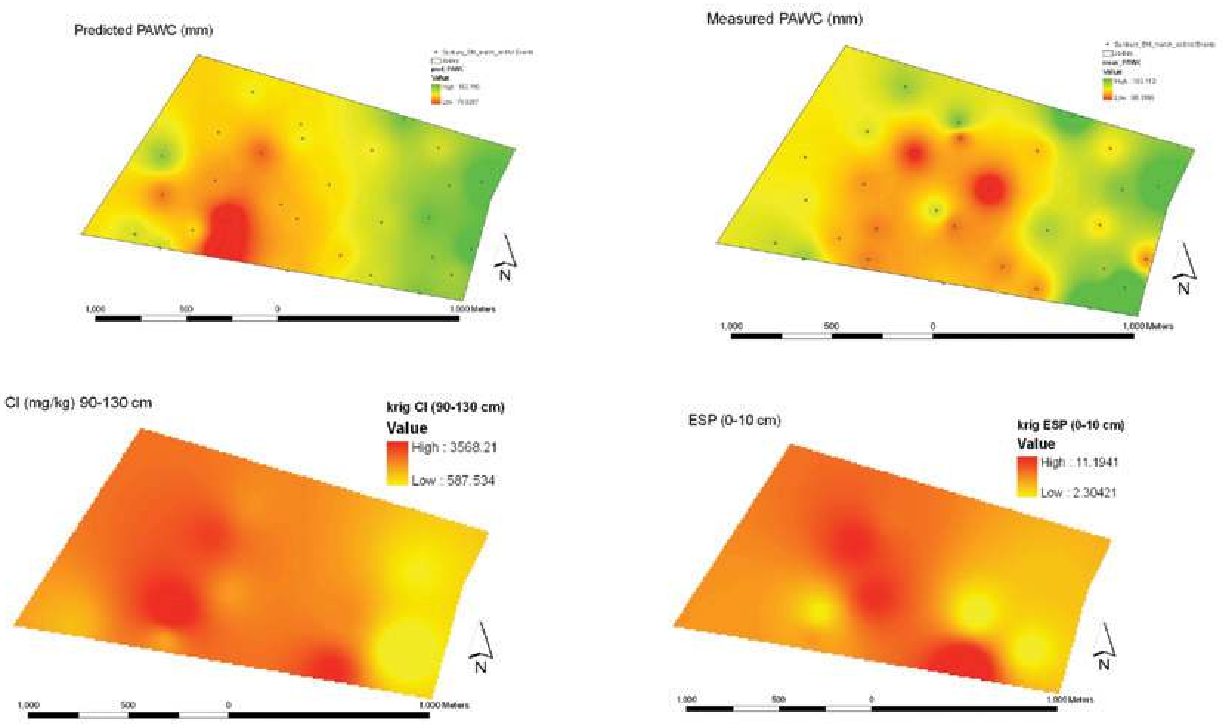


**Figure 4. Comparison of spatial variability of (a) predicted PAWC and (b) measured PAWC with (c) subsoil Cl and (d) surface soil exchangeable sodium percent.**

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# Characterizing soil spatial variability using bi-dimensional empirical mode decomposition

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## Abstract

Soil spatial variability evolves from the combined action of soil physical, mineralogical, chemical and biological processes that operate with different intensities and at different scales. Soil spatial variability is therefore scale-dependent and variation is often nonstationary and nonlinear. Empirical mode decomposition (EMD) has recently been used in one dimension to separate the overall soil spatial variability into various components, each representing different scales. The technique is known to accommodate nonstationary and nonlinear data series. In this research, we used the two dimensional extension of EMD, known as bi-dimensional EMD (BEMD), to separate multi-scale spatial variability in a two dimensional field. To demonstrate the approach, we used a digital elevation model (DEM) of the Simmons Creek catchment in New South Wales, Australia. The BEMD separated the overall variations in the DEM into 6 different bi-dimensional intrinsic mode functions (BIMFs) according to their characteristic scales. The first function (BIMF1) separated the variations at the scale <1.67 km, which can be attributed to the terrain/landscape variations at field- to farm-scales. Sub-catchment level variations were separated in BIMF2 (2–5 km) and BIMF3 (8–10 km), and the major topographic trends in the catchment were separated in BIMF4 (12–15 km) and BIMF5 (>15 km). The slope and wetness index calculated from the BIMFs showed more detailed features in topographic variations than those calculated from the DEM. Separating two-dimensional spatial variability of terrain attributes at multiple scales using BEMD might provide opportunities for multi-scale digital soil mapping, data filtering, denoising and dimensionality reduction.

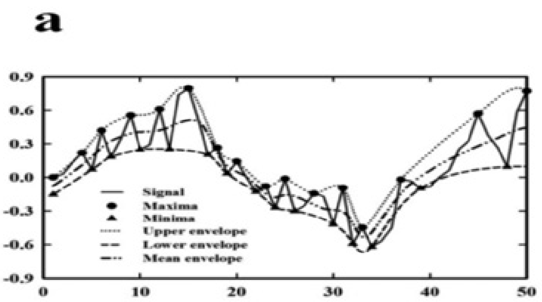
## Introduction

The Hilbert-Huang transform (HHT; Huang et al., 1998) is a method only recently applied to soil science (Biswas et al., 2009; Biswas and Si, 2011) and that can be used to analyse nonstationary and nonlinear data such as that representing the variability of soil properties in space. Empirical mode decomposition (EMD), the first of two steps, decomposes a data series into a number of components adaptively without using a predetermined decomposition structure (methods such as the wavelet or Fourier transform have predetermined structures). The decomposition is data-driven, based on the local frequencies or scales embedded in the data (Huang et al., 1998). The adaptive nature of this method successfully decomposes nonlinear and nonstationary data series in the time and/or spatial domain. The EMD extracts oscillations from the data series into a finite and often small number of mode functions (known as intrinsic mode functions, IMFs) that are generally in good agreement with intuitive and physical signal interpretation. The two-dimensional (2D) extension of the EMD has also been introduced and is known as bi-dimensional EMD (BEMD; Linderhed, 2002). Similar to the EMD, BEMD also extracts the variations in 2D data series into a small number of bi-dimensional IMFs (BIMFs) according to dominant scales. BEMD is increasingly being used in signal and image processing applications and holds promise to separate out scale-specific variations in soil and terrain properties in two dimensional spatial fields. It has the ability to locally separate the spatial frequencies that create the variations in two dimensions. The aim of this research is to demonstrate the use of the BEMD to separate the variability in soil or terrain properties from a two dimensional spatial field.

## Materials and Methods

To demonstrate the BEMD, we used a 20 m × 20 m grid resolution DEM of Simmons Creek catchment in southern New South Wales, Australia. The catchment is located at the eastern edge of the Riverine Plain, near the township of Walbundrie, NSW. The catchment covers an area of 178 km2 and around 98% of the area is committed to agricultural holdings. The terrain of the catchment is relatively level to undulating at 180 to 200 m above sea level, rising steadily to a local maximum of 340 m on the mid-westerly boundary at Mullemblah hill. There is a general trend of increasing elevation towards the north-east boundary of the catchment.

We start with a short overview of the one-dimensional EMD method before describing the BEMD. The main goal of the EMD is to adaptively decompose a dataset in one-dimension into a number of components or IMFs that satisfies two conditions; 1) it has the same number of zero crossings and extrema, and 2) at any point in space, the mean value of the upper and lower envelope (defined by the maxima and minima), tends towards zero. While, the zero crossing is the point where the series changes its sign, the extrema are the largest and smallest values of a series (the maxima and the minima; Fig. 1a). The extreme values are used to prepare the upper (joining maxima) and lower (joining minima) envelopes using a spline function. The mean of the envelopes is calculated at each location (Fig. 1a). An iterative process that separates the finest local mode is used to obtain an IMF. The iterative process results in the elimination of riding waves or fluctuations in the series and in the smoothing of uneven values (amplitudes) of the properties in consideration. The BEMD also decomposes a spatial series into a number of BIMFs based on the local frequency (or scales) or oscillation information. The method is very similar to that of the EMD but with more complicated methods of extrema detection and surface interpolation of envelopes. Assume elevation (or a soil property), *S* measured in a spatial field, (*x, y*) that can be decomposed into a finite number of BIMFs separating the variations at different frequencies or scales using BEMD. The steps of BEMD are shown with a flowchart (Fig. 1b). For a more detailed account of the methodology see Linderhed (2002) and Nunes et al. (2003).



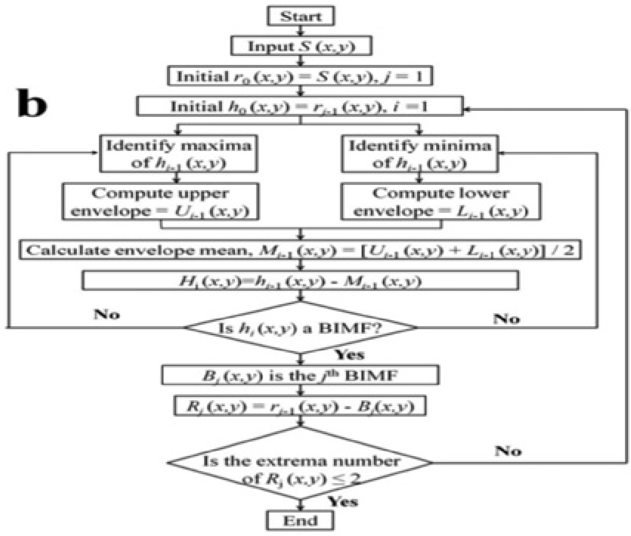




**Figure 1: a) Maxima, minima, and upper, lower and mean envelope of a spatial series and b) steps in bi-dimensional empirical mode composition (BEMD).**

The BEMD was implemented in Matlab software (Mathsoft Inc., Natick, MA). After separating each BIMF, the residuals were calculated. At each step, the sum of variations in BIMFs and the residuals were equal to the variations in the original DEM. The scales of the BIMFs were determined using 2D Fourier transform, which converts the spatial information into frequency information. The frequencies were converted to the scales using scale = sampling interval / frequency. Different geomorphic products including slope and topographic wetness index were calculated from the original DEM and different BIMFs using an open source Geographic Information System (GIS) software (System for Automated Geoscientific Analyses (SAGA); available online at http://www.saga-gis.org/en/index.html).

**Results and Discussion:**

The DEM of the study area was decomposed into 6 different BIMFs (BIMF1 though BIMF6; Fig. 2a - f) - separating the variations at different scales. The residual after separating each BIMF was also calculated and presented in Fig. 2g – l. The 2D Fourier transform of the BIMF1 identified a minimum frequency of 0.6 cycles per km indicating a scale of maximum 1.67 km, the smallest scale of variation. These scales are likely representing variation at the field to farm level - important to many users of topographic data. Separating BIMF1 leaves a residue (Fig. 2g) of larger scale variations which was subsequently decomposed further into higher order BIMFs. The BIMF2 (Fig. 2b) and BIMF3 (Fig. 2c) separated the variation at scales of around 2 – 5 km and 8 – 10 km, respectively. This medium- to large-scale variation may represent the sub-catchment level. The fourth (Fig. 2d) and fifth (Fig. 2e) BIMFs separated the variation at much larger scales of 12 – 15 km and >15 km, respectively showing the elevation trend within the catchment. The gradual change in the elevation from south-west to the north-east corner of the catchment was clearly separated in BIMF4 (Fig. 2d). After separating five BIMFs, there was almost negligible variation to separate in BIMF6 (Fig. 2f).The sequence of BIMFs shows the change of scales (Fig. 2a - f).

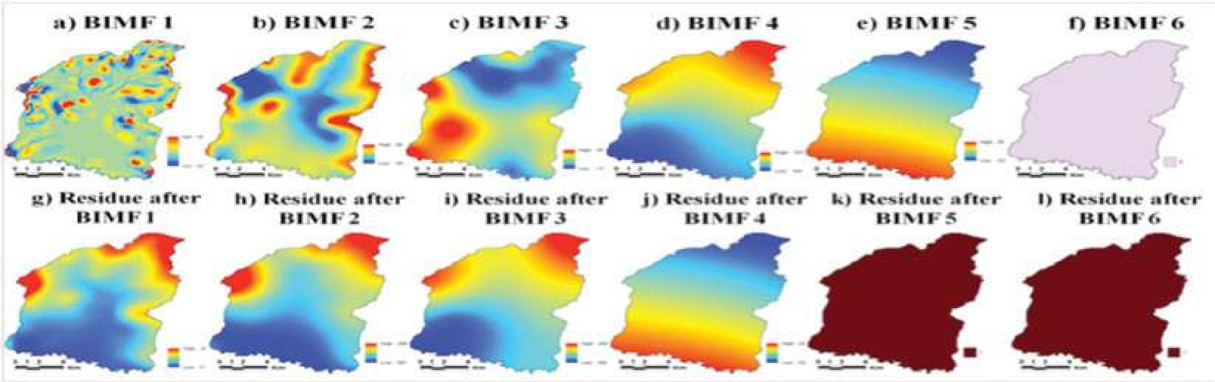



**Figure 2: a – f) Bi-dimensional intrinsic mode functions (BIMFs) of the digital elevation model of the study area and g – l) the residue after separating BIMFs.**

The BEMD separated the overall variations in the DEM into a number of BIMFs according to the dominant scale of variation (these can also be considered as multi-scale DEMs). There is no *a priori* basis for the BEMD, which adaptively separated the variations as they are present in the data (Linderhed, 2002). After separating out small scale variation, the data was left with larger scale trends. The large scale trend creates the nonstationarity in the dataset. Separating the variations present at different scales can help deal with different assumptions associated with data including nonstationarity and nonlinearity (Huang et al., 1998; Linderhed, 2002). Separation of DEMs into multiple scales will help users choose the level of information required - from field or farm scale to large catchment or regional scales.

The BIMFs or multi-scale DEMs can be used for various purposes such as deriving various geomorphic products (e.g. slope, wetness index etc.) for environmental applications. For example, slope was calculated for the original DEM (Fig. 3a). Slopes at different scales were calculated from the corresponding BIMF (Fig. 3b - e). The major variations in slope were present mainly in the northern half of the catchment (Fig. 3a). The slopes calculated from the original DEM (Fig. 3a), without scale separation, masked various small scale features. However, slope calculated from the BIMFs (Fig. 3b – e) retained detailed topographic features. For example, the western and north-eastern corners had greater topographic variation than other parts of the catchment. While, the slopes calculated from the BIMF1 (Fig. 3b) could identify more detailed topographic variation at the scales < 1.67 km, the slopes calculated from the original DEM (Fig. 3a) could not identify those features. This may be because multi-scale topographic variations smooth the variations in slopes by masking individual features. The analysis has produced slope variations at multiple scales, enabling the user to select the scale of interest for their specific application.

As with the slope, the topographic wetness index was also calculated for the original data (Fig. 3f) and the BIMFs (Fig. 3g - j). The BIMF1 derived wetness index (Fig. 3g) clearly shows more detailed features at small scales compared to the wetness index derived from the original DEM (Fig. 3f). The original data, without scale separation, could not identify all the wetness features within the catchment.

After separating the variations at multiple scales, one can match the dominant scales of data with the dominant scales of the covariates in preparing a multi-scale digital soil map and scale-specific prediction of soil properties. Separation of required information at dominant scales can help reduce the dimensionality in very large data sets, for example those collected with remote sensing technologies. Remotely sensed data can be captured quickly and contains detailed information relevant to variation in soil properties. However, these data often carry a finer scale noise. Separating the smallest scale variations using BEMD will help in filtering to denoise the dataset.

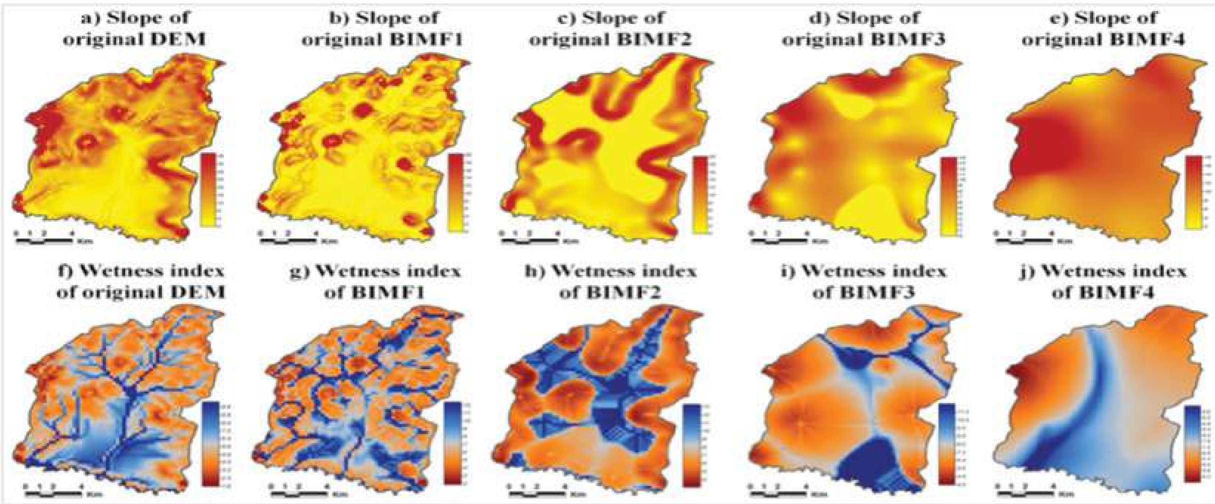



**Figure 3: a – e) Slope and f – j) wetness index of the study area calculated from the DEM and BIMFs.**

In summary, the decomposition of variation in spatial soil data into multiple scales using bi-dimensional empirical mode decomposition holds promise for various applications. Separation of DEMs into multiple scales enable users to choose the level of information wanted for their specific applications - from field or farm to large catchment or regional scales.

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# Spatial representation of errors in the Australian soil resource information system (ASRIS) in Tasmania

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The spatial representation of error was undertaken in two pilot areas in Tasmania at level 4 and at level 5 for ASRIS data. The area weighted mean value for organic carbon (OC) and pH, coefficient of variation, combined error and the largest error term were produced for each ASRIS layer (1, 3, and 5) for OC and pH.

Maps were produced that visually quantify errors and show the sources of error across polygons mapped at different scales. The maps allow for clear differentiation of area weighted mean values at both level 4 and level 5 of ASRIS data. Maps of combined errors allow for visual estimation of the amount of error associated with the mean values which can be standardised in the maps of coefficient of variation. The fourth map of what the largest source of error is for that particular layer/soil attribute shows that the dominant source of error for pH in layer 3 is generated from the range of land facets making up each polygon (i.e. many components with diverse values) with a lesser amount of error due to spatial error. The dominant source of error for OC in layer 1 was due to spatial error with a lesser amount of error due to component error. In no instances was measurement error the dominant source of error. The difference in dominant source of error from component to spatial are likely to be due to the difference in scale of the information used to populate the mapped polygons.

# Piloting ground cover measurements in North West Victoria

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Maintaining adequate ground cover, particularly during peak hazard periods and/or on susceptible land is one of the most effective approaches to reduce the risk of wind and water erosion. Developing tools to accurately detect and monitor changes in ground cover would significantly improve management and management practices. Consequently a remote sensing methodology to estimate fractional ground cover could greatly improve our knowledge of erosion risk and help target and evaluate efforts to reduce the threat of permanent soil loss and subsequent decline in production and ecosystem services provided by the soil.

Measurements of the three fractions of ground cover; bare ground, non-photosynthetic vegetation and photosynthetic vegetation across a range of soils is required to calibrate the satellite imagery and provide a repeatable relationship. For adequate calibration of the satellite imagery it is also important to measure a range of cover fractions from very high to very low cover in all of the fractions.

From January to February in 2012, 57 sites across the north west of Victoria were described, photographs taken and the fractional ground cover measured using the national standards. In May 2012, 30 of these sites were revisited and the measurements repeated. On average the total cover at these 30 sites reduced by 21.3% from an average total cover of 66.8% (with a range from 24.5 to 94.0%) in Feb to 52.6% (with a range from 2.5 to 87.5%) in May.

Three independent methods of remote sensing analysis will now be trialled using all the data collected in Victoria, to produce a fractional ground cover map.

# Assessing soil electrical conductivity for site-specific management with electromagnetic induction and response surface sampling design: A case study of East Nile Delta, Egypt

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Spatial characterization of the variability of soil electrical conductivity (EC) is required for site-specific management. The objective of this study is to incorporate an integrated methodology involving a hand-held electromagnetic sensor (Geonics EM38), and the ESAP (Electrical conductivity Sampling, Assessment, and Prediction) model to assess, and predict soil electrical conductivity at field scale for site-specific management. The salinity of 67.2 ha of a wheat pivot field at East of Nile Delta, Egypt, was analysed by reading the bulk soil electrical conductivity (ECa) with the EM38 sensor at 432 locations. The spatial response surface sampling design (RSSD) was invoked to select 20 optimal calibration soil sampling sites from EM38 sites. At those sites, soil core samples were taken at 0.3 m intervals to a depth of 0.9 m, and electrical conductivity of the saturation extract (ECe), saturation percentage (SP), and water content (WC) were measured. ESAP was employed for calibrating the EM38 sensor, and predicting soil salinity at the sampled depth intervals. Salinity was the dominant factor influencing the EM38 readings. The multiple linear regression (MLR) calibration model was used to predict the depth-specific ECe values at the remaining non-sampled locations. The MLR calibration model predicted ECe from EM38 readings with $R^2$ ranging from 0.41 to 0.73 for the multiple-depth profile. Furthermore, the MLR model provided field range estimates of soil salinity. Ninety-one precent of the field had ECe values below 4 dS $m^{-1}$. The obtained salinity maps were helpful to display the spatial patterns of soil salinity for site-specific management.

**Keywords**
Precision agriculture, Soil salinity, Electromagnetic induction, Spatial response surface sampling, Multiple linear regression, Site-specific management, East Nile Delta.

# The use of biochar for restoration of agricultural productivity on previously mined land in the Hunter Valley

**Andrew Regan**, John Lawrie, Greg Hancock, Shane Curry

Open-cut coal mining is a major industry in the Hunter Valley, NSW and one which impacts on agricultural land. An open-cut mine typically strips the topsoil and overburden, mines the resource underneath, reinstates overburden and topsoil and undertakes rehabilitation processes in an attempt to restore land to its original productive capability. The use of biochar in mine site rehabilitation in the Hunter Valley has the potential to restore soil physical, chemical and biological functions and thereby improve both rehabilitation success and agricultural productivity.

The potential impacts of biochar on mine rehabilitation may be two fold:

- Improved soil physical and chemical properties; and
- Reduction in GHG emissions due to improved N fertiliser efficiency.

The restoration of disturbed land back to agricultural productivity is limited in many instances by inherently poor physical and chemical soil properties including water holding capacity, aggregate stability, cation exchange capacity, pH, and nutrient (carbon, nitrogen, phosphorous, sulphur and potassium) levels. Biochar can potentially act as an ameliorant that can improve soil productive capacity by enhancing the soil properties listed above. The application of biochar also has important implications for long term sequestration of carbon in the soil, but these will not be covered in this study.

The benefits of using biochar in agricultural soils have been widely reported, but relatively few field studies have been conducted in Australiaand, to our knowledge, no previous biochar studies have focused on topsoils and overburden stripped from mines and subsequently rehabilitated in the Hunter Valley region. Furthermore the economic viability and proposed greenhouse gas benefits of biochar production requires investigation. For example, obtaining a sustainable supply of feedstock poses a major limitation to biochar production in the Hunter Valley region and in many instances, world-wide.

This project will consist of:

1. A life-cycle analysis of biochar production in the Hunter Valley to determine the long-term net greenhouse gas emissions and identify potential limitations to production and distribution;
2. Controlled pot experiments at the University of Newcastle; and
3. A three year field trial on rehabilitation works at Bengalla Mining Company outside Muswellbrook, NSW.

The field trial will compare multiple application rates of biochar and the role of green mulch crops to ascertain optimal combinations for restoring soil functionality. Grazing pastures (native or introduced grasses) will be planted and revegetation success will be measured using Landscape Function Analysis (LFA) and/or crop yield. Monitoring of "downstream" effects of biochar application and any improvements in soil water and nutrient holding capacity will be required. Consequently, runoff from the field plots will be collected in underlying flumes and measured for quantity and quality. Seasonal monitoring of soil properties, such as CEC, pH and nutrient levels will also be undertaken to contribute to the limited field data on biochar application and confirm hypothesised improvements in soil properties.

The aims of the pot and field trials are to increase the overall knowledge on biochar and confirm its potential to improve soil properties and agricultural productivity on disturbed mined lands. The life-cycle analysis component aims to elucidate the economic constraints to production and net greenhouse gas emissions. Thus the results of the field trials and life-cycle analysis will confirm the sustainability of biochar production within the Hunter Valley region and its effectiveness in restoring soil productivity on previously disturbed mining land for agriculture.

A further discussion on theimpact of agriculture on mining lands and possible improvements to soil physical and chemical properties following biochar application on such disturbed soils will be presented.

# Use of coal mine waste as a soil ameliorant

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Coal mine tailings are a waste product of the mining process and currently pose a closure liability to mining companies. One way to address this problem is by utilising coal tailings as a feedstock for the production of char, in a process similar to biochar production. The process would not have the same GHG emissions profile as biochar production because the feedstock materials are markedly different. Coal is more inert and carbon dense than typical feedstocks utilised for biochar production as it has been under immense temperature and pressure for millions of years. It was hypothesised that coal tailings could be converted by a pyrolysis process into a higher order reuse product to benefit mine rehabilitation and agricultural production. 'Chailings' (char from tailings) were expected to embody similar properties to that of biochar, including high internal surface area and porosity, high cation exchange capacity (CEC), increased nitrogen sorption and nutrient retention capabilities. The significant preliminary findings of applying chailings at different rates to soils was that with increasing application rate there was a near linear increase in water holding capacity (WHC) for chailings produced at temperatures above 600°C. Increasing the WHC of a soil is typically related to an increase in productivity of the soil. Furthermore, by increasing the productivity more plant mass, above and below ground, can be supported which has been shown to reduce soil degradation and erosion. Hunter Valley mining spoils are renowned for being low in WHC, hence there is potential for the use of chailings to improve rehabilitation outcomes by promoting soil health through improved water holding capacity, improved soil nutrient retention and CEC, improved soil structure and possibly beneficially altering soil pH.

The findings of this research will be presented.

# Analysis of soil samples using the high temperature desorber

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The analysis of complex environmental materials such as soils and sludges for semi-volatile pollutants may be simplified using rapid thermal desorption techniques, in which the sample material is desorbed directly to the gas chromatograph (GC). This process eliminates solvent extraction by volatilizing the organics of interest rapidly from the sample matrix and transferring them immediately to the injection port of the GC without trapping or dilution. The methods, analytical results and instrumental setup with the GC will be shown.

# High throughput measurement of soil extracts using flow injection analysis

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Soil extractants have been developed to extract nutrients from the soil in the form they are available to plants. Different soil types require different extraction solutions. These test procedures attempt to estimate the amount of nutrients available during the growing season. The results are not the total of a given nutrient in the soil, but the "plant available" content within the sample. This allows soil fertility to be measured.

The results of the analysis are used to determine what amount of fertilizer and other soil amendments will be recommended for future growing season. For example, if too much lime is applied to a given soil, the pH may become so high that needed nutrients (like iron) become unavailable to plants. If too much fertilizer (containing N and P) is applied, excess could run off, and adversely affect water quality. Also, applying these materials at levels above what is required is not economical. Applying too much or too little fertilizer can adversely affect plant growth reducing yields.

Flow Injection Analysis is an effective way to automate the analysis of soils with high sample throughput. Where a "typical" FIA method allows the analysis of an average of 60 samples and standards per hour per channel (or 300 samples and standards per hour on a 5 channel system), Lachat's ultra high throughput methods achieves throughputs of up to 120 samples and standards or more per hour per channel.

A selection of data from the suite of methods, along with a comparison of reagent use between the "regular" and "UHT" suite of methods will be shown.

# Analysis of Environmental Samples by Simultaneous CCD-ICP-OES

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Heavy metals are naturally components of the Earth's crust. Although at trace levels some heavy metals (e.g. selenium and zinc) are essential for the human body, most of them are toxic or poisonous even at low concentrations. Heavy metals include elements like arsenic, cadmium, chromium, mercury, lead, and thallium. They often enter the body via the food chain, ambient air, or drinking water. Since the start of the industrial revolution, waterways and coastal waters have been heavily polluted. Likewise, contaminants have been introduced into the soil, with negative consequences not only for the food chain, but also for drinking water. Ultimately human health is at risk.

Because of its multi-element determination capability, high dynamic linear range, and sensitivity, inductively coupled plasma optical emission spectrometry (ICP-OES) is widely used for the analysis of waste water, soil, and sludge. Trace, minor, and major elements can be determined simultaneously ensuring low costs of analysis. The application is described in several standard procedures such as US-EPA Methods 200.7, 3050 B and 6010C, EN 13346:2000, and ISO 11885:1996. Due to economical reasons, environmental labs usually have a larger number of samples per day in order to work highly cost-effective, simultaneous ICP-OES instruments are required. The applicability of the new SPECTRO GENESIS ICP optical emission spectrometer (SPECTRO Analytical Instruments, Kleve, Germany) with axial plasma observation was tested on typical environmental samples.

# Does incubating bagged soil cores prior to oven drying affect extractable nutrient concentrations?

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The 2011 Australian Standards for Soil and Chemical methods state that soil samples should be kept cool or cold between field sampling and laboratory analysis to minimise biological transformations and other chemical reactions. This can be difficult in practice, as soil samples sent to commercial laboratories are often posted from their origin in sealed plastic bags and could spend several days in transit, before being dried for analysis. Depending on the air temperature during transit, it is possible that chemical and biological transformations may influence extractable nutrient concentrations. This replicated study used 2 different dairy pasture soils (red clay loam and sandy loam) to test the effect of storing field moist soil samples in sealed plastic bags prior to drying. Samples were stored at 4˚C, 20˚C and 35˚C for 48 h and the effect on Olsen and Colwell P, Colwell K and KCl S was measured. Results showed that the extractable nutrient concentration of soils were generally not affected by the incubation temperature, with extractable S concentrations marginally ($P = 0.06$) lower when incubated at 20˚C compared to cooled to 4˚C. It is possible that some S was immobilised by soil microbes under the warm, moist conditions, whereas temperatures of 35˚C may have been less conducive to microbial activity. These results suggest that some biological changes may have occurred as a result of temperature treatment, however the effect on extractable nutrient concentrations were minimal and the importance of cooling samples prior to extractable nutrient analysis, requires further investigation.

# A preliminary assessment of the ability of the DGT soil phosphorus test to predict pasture response in Australian pasture soils

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Recent research has found that Diffusive Gradients in Thin-Films (DGT) technology is more effective at predicting wheat response and uptake to added P than currently used tests such as Colwell P or resin P. Since intensive pasture industries are large consumers of P fertiliser and the common agronomic soils tests (Olsen and Colwell P) have limitations in terms of their harsh extraction process and difficulties in predicting pasture P response across a wide range of soil types, climatic conditions and pasture species, an examination of the new DGT-P test was justified. As pasture calibration studies are expensive, this study used historic data from 25 sites from the National Reactive Phosphate Rock project (1992-1994). Stored soil samples were analysed for DGT-P, Olsen P and PBI and Colwell P. When the recent soil test P concentrations were related to the relative pasture yield, the results showed that in general, Colwell and Olsen extractable P methods more accurately predicted relative pasture P response compared to DGT-P. We hypothesise that bicarbonate P tests are extracting some of the less labile and organic P sources in the pasture soils and therefore better reflect the P sources that pasture roots are able to access over an annual cycle. In contrast, DGT-P may better reflect inorganic P sources likely to be important in a short rotation wheat crop. However, it is recommended that further, more comprehensive studies be undertaken with newer pasture species and N fertiliser applications, before a firm conclusion can be drawn.

# Spatial distribution of soil nutrients and organic matter in an arable paddock on flat landscape

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## Abstract

Soil properties can vary spatially due to differences in topography, parent material, vegetation or land management. The objective of this study was to quantify the spatial distribution of soil nutrients and organic matter within a 10.4 ha paddock (predominantly Templeton silt loam; flat topography) in Canterbury, New Zealand that had a long-term history of arable cropping. Samples (0–7.5 cm depth) were collected in a grid pattern (30–35 m sampling intervals; total of 91 samples) for determination of mineral N, anaerobically mineralisable N (AMN), Olsen P, total C and total N. The data were evaluated using geostatistical and classical statistical methods. Geographic information system (GIS) was used to produce nutrient and organic matter maps. Although the paddock had been managed uniformly for many years, nutrient levels presented substantial variability. All measured variables except mineral N showed moderate to strong spatial dependence and the effective spatial autocorrelation distance of AMN, Olsen P, total C and total N ranged from 184 to 700 m. Distinct fertility zones were identified for both P and N suggesting that improvements in nutrient use efficiency could be achieved through variable rate fertilization. The relationship between soil texture and organic matter was examined using samples collected along two perpendicular transects within the paddock. Soil C showed a strong positive correlation ($R^2 = 0.79$, $P<0.001$) with the amount of clay plus fine silt (<5 µm fraction) and a negative correlation ($R^2 = 0.81$, $P<0.001$) with sand content. These results suggest that textural variation was a major factor influencing within-paddock variability in soil organic matter at this study site.

## Introduction

Soil properties can vary spatially due to a variety of factors, including climate, parent material, topography, vegetation and land management (Trangmar 1986; West 1989; Weijun 2010). Within paddocks, soil characteristics and productivity can vary even over small spatial scales (Wells *et al*. 2000; Gaston *et al*. 2001). Soil fertility assessments commonly rely on a random soil sampling protocol to obtain an average fertility value for a paddock. Thus, spatial variability is not fully considered and consequently, some parts of the paddock may receive excessive fertiliser while other parts may suffer nutrient deficiency due to under-application.

Geostatistics coupled with geographic information system (GIS) mapping technologies has substantial potential to discern spatial patterns within paddocks. In this study we used geostatistics and GIS to map and interpret the spatial variability of nutrients and soil organic matter at a paddock scale. A second objective was to identify factors contributing to within-paddock variability in the measured variables, especially soil organic matter.

## Materials and Methods

The experimental site was a flat (average slope gradient < 0.5%), rectangular paddock (400 m long x 260 m wide; 10.4 ha) at Lincoln University, Canterbury, New Zealand that had a long-term history of arable cropping (wheat, peas, and seed crops). The predominant soil type is Templeton silt loam (Orthic Tenosol; Isbell 1997). Samples (0–7.5 cm) were collected in a grid pattern with a sampling distance of 35 m across the paddock and 30 m along the paddock (total of 91 samples). The samples were air-dried and sieved (4 mm) prior to analysis. Mineral nitrogen ($NH_4^+$ and $NO_3^-$), anaerobically mineralisable N (Keeney and Bremner 1966), total soil C and N (Leco C/N analyser) and Olsen P (Olsen *et al*. 1954) were measured. Classical statistical parameters and Geo-statistical analyses were conducted.

## Results and Discussion

*Classical statistics*

Although the paddock had an even topography and had been managed uniformly for many years, substantial variability was observed for Olsen P, Mineral N and AMN (CV ranged from 16 to 33%; Table 1). Soil organic matter also exhibited significant variability, with total C ranging from 19 to 31 g/kg (Table 1). Variability was reduced somewhat by elimination of outliers identified using Q-Q plots. For example, exclusion of two outliers for mineral N reduced the CV from 33 to 26% and, when a single outlier for Olsen P was excluded, CV decreased from 26 to 21%.

**Table1: Descriptive statistics of soil properties at the experimental site.**

| Soil properties | Min. | Max. | Mean | Std.Dev | CV % | Skewness | Kurtosis |
|---|---|---|---|---|---|---|---|
| Mineral N (µg/g) | 15 | 93 | 33 | 11 | 33 | 1.159 | 1.771 |
| AMN (µg/g) | 37 | 83 | 50 | 8 | 16 | 0.659 | 0.754 |
| Total C (g/kg) | 19 | 31 | 27 | 2 | 8 | -0.649 | 0.154 |
| Total N (g/kg) | 1.8 | 2.5 | 2.2 | 0.2 | 8 | -0.850 | 0.657 |
| Olsen P (µg /g) | 14 | 53 | 20 | 5 | 26 | 0.804 | 0.118 |

Because of high skewness (Table 1) and the presence of outliers, it was necessary to normalize the data prior to the geostatistical analysis. Logarithmic transformation reduced skewness and kurtosis for AMN and Olsen P and the transformed data passed the normality test. Box-Cox transformed data for total C and total N passed the normality test, but the transformed mineral N data did not. Transformed data were used in the spatial analysis.

*Geostatistical analysis*

Geostatistical analysis was conducted to determine the spatial structure of all soil properties. The theory of geostatistics has been widely used in spatial studies of soils (Trangmar *et al*. 1985). In this case study, empirical semivariogram values for soil properties were obtained using *Equation* 1:

$$\gamma(h)=\frac{1}{2N(h)}\sum_{i=1}^{N(h)}[Z(Xi)-Z(Xi+h)]^2 \qquad (1)$$

Where $\gamma(h)$ is the sample semivariance between all observations $Z(Xi)$, Z is the measured value at a particular location, *N(h)* represents the number of paired data at the distance *h*, and *(h)* is the separation distance.

**Table 2: Geostatistical parameters for soil nutrients and organic matter.**

| Soil property | Semi var. model | Nugget $C_0$ | Sill $C_0$+ C | Range α (m) | Nugget/Sill (%) | Spatial dependence | $R^2$ |
|---|---|---|---|---|---|---|---|
| AMN | Spherical | 0.0017 | 0.0034 | 184 | 50 | moderate | 0.83 |
| Olsen P | Gaussian | 0.0056 | 0.03 | 700 | 14 | strong | 0.96 |
| Total C | Gaussian | 11.1 | 53.2 | 300 | 21 | strong | 0.98 |
| Total N | Gaussian | 10.4 | 29.0 | 268 | 36 | moderate | 0.97 |

Parameters of the variograms for soil nutrients and organic matter are given in Table 2. The ratio of nugget variance ($C_0$) to sill variance ($C_0$+ C) indicates the degree of spatial correlation, with ratios <25%, 25–75%, and >75%, indicating strong, moderate and weak spatial correlation, respectively. For the measured variables, there was moderate (AMN, total N) to strong (Olsen P, total C) spatial correlation within the experimental site (Table 2). The range value (α), which indicates the distance beyond which sampling points are independent of each other, was as high as 700 m for Olsen P and ranged between 184 and 300 m for the other measured variables.

Spatial distribution maps for soil nutrients and organic matter were obtained using ordinary kriging methods based on the observed semi-variogram parameters. Unlike the other variables, mineral N did not show a clear spatial structure (Figure 1). Olsen P was highest near the paddock gate (possibly due to fertiliser spillage) and along the southern paddock edge. Much of the centre area had Olsen P <20 µg/g (the value below which application of fertilizer P may be beneficial for arable crops). In paddocks such as this, with distinct P fertility zones, it may be possible to achieve significant improvement in P use efficiency through variable rate fertilization.

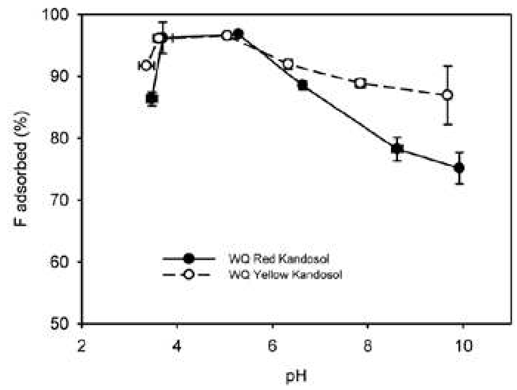


**Figure 1. Spatial distribution maps for AMN, Mineral N and Olsen P.**

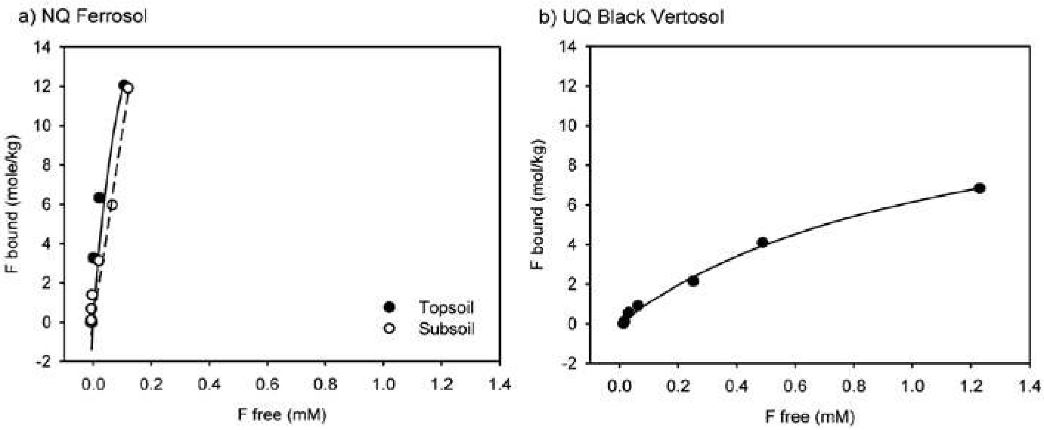


**Figure 2. Spatial distribution maps of soil total N and C.**

Total N and C showed distinct spatial patterns, with low values in the south-west and high values in the northern and eastern sectors (Figure 2). Within-paddock variability in total soil C cannot be attributed to differences in management practices as the entire paddock had been managed in a uniform manner for many years. To examine whether the spatial pattern was related to textural variation, we performed a textural analysis on samples collected along north-south and east-west transects through the paddock. These transects included samples with a wide range of total C and N values. The soils were separated into three size fractions (>50, 5–50, and <5 µm) after sample dispersion using an ultrasonic vibrator. The results revealed that there was considerable textural variation within the paddock. For example, the sand content (>50 µm) ranged from 15% to 40% within the analysed samples. There was a very strong, positive relationship ($R^2 = 0.79$) between total C and the amount of fine (<5 µm, i.e., clay and fine silt) material and a negative relationship ($R^2 = 0.81$) between total C and sand content (Figure 3). Similar relationships with texture were observed for total N (not shown). The positive relationship between fine material and total C (and N) reflects the fact that organic matter tends to be protected by the surfaces of fine mineral soil particles.

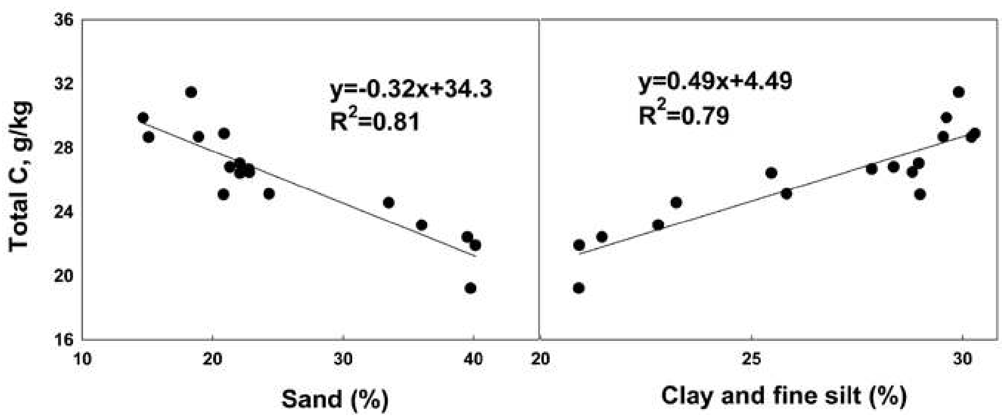


**Figure 3. Relationship between soil texture and total C, based on samples collected along two perpendicular transects in the paddock.**

## Conclusions

There was substantial spatial variation in the measured soil properties within this flat, uniformly-managed paddock. Anaerobically mineralisable N and Olsen P both showed moderate to strong autocorrelation, while soil organic matter (total C and N) had moderate spatial autocorrelation. Our results confirm that the combination of geostatistics and GIS mapping can provide a powerful tool to describe the spatial distribution of soil properties and to develop high resolution maps that may aid variable rate management (e.g. fertilisation) at a paddock scale. Soil texture difference was a major contributor to spatial variability in soil organic matter at this study site.

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# Nitrification inhibitor lowers relative nitrous oxide flux compared to untreated and urease inhibitors applied to urea during winter 2011 cropping in southern Queensland

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**Abstract**

Wheat production dominates cropping in Australia's northern grains region and urea is the most commonly applied nitrogen fertiliser. However, nitrogen fertilisers contribute substantially to our nation's greenhouse gas emissions principally as nitrous oxide. Urease (NBPT trading as Agrotain®) and nitrification (DMPP trading as Entec®) inhibitors can be added to urea to change the nitrogen release pattern but the effects on nitrous oxide loss have not yet been fully evaluated in this environment. A field trial with wheat was established in 2011 measuring relative nitrous oxide losses from untreated urea against Agrotain® and Entec® treatments band applied at-sowing at 60 kg/ha N. Gas samples from static chambers were collected following rainfall-triggered events (15 mm) and analysed for carbon dioxide, methane and nitrous oxide. Nitrous oxide fluxes were generally <5 µg $N_2O$-N/m$^2$/hr for all treatments over ten sampling events from late June (sowing) to mid-October (anthesis). However two rainfall events in late August generated 20 µg $N_2O$-N/m$^2$/hr from untreated urea and 9.5 µg $N_2O$-N/m$^2$/hr from Agrotain®, whilst Entec® treatment was <1.5 µg $N_2O$-N/m$^2$/hr. While there was a spike in emissions with untreated urea (in this trial), growers require further information on the effectiveness of minimising nitrous oxide losses via urea modifiers on crop growth, nitrogen recovery and utilisation, and greenhouse gas emissions as part of their nitrogen management decision making.

**Key Words**

Entec®; DMPP; Agrotain®; NBPT

## Introduction

Fertiliser nitrogen is a major nutritional input for wheat, the principal crop within Australia's northern grains region. As the length of time since land use change into cropping increases, so too does the likely amount of external (fertiliser) nitrogen input required for crops to achieve 90% maximum yield – a surrogate of the economically optimum nitrogen supply. Nitrous oxide emissions associated with nitrogen fertiliser are a substantial contributor to the nation's total greenhouse gas budget (Dalal *et al*. 2003). Urea is the predominant nitrogen fertiliser form used in Australia's northern grains region. However the nitrogen release characteristics may be modified through "efficiency enhancers" such as controlled release coatings, nitrification inhibitors or urease inhibitors.

Of these release modifiers, two – one a nitrification, the other a urease inhibitor - are commercially available for use in Australia. The nitrification inhibitor 3,4-dimethylpyrazole phosphate (DMPP, trade name Entec®) suppresses nitrification by limiting the bacterial action of *Nitrosomonas* species, slowing the conversion of nitrogen from ammonium to nitrate forms. The urease inhibitor N-(n-butyl) thiophosphoric triamide (NBPT, trade name Agrotain®) is a product primarily targeted at surface broadcast application for top-dressing. It slows the conversion of urea to ammonia reducing the potential ammonia loss by volatilization. Whilst it has been evaluated in southern Australian wheat production regions under their Mediterranean climate (with a wetter, colder growing season), these products have received little examination in the more monsoonal climates present in the northern wheat production area with drier and warmer growing seasons. Also, the semi-arid nature of the northern cropping region with unreliable winter growing season rainfall requires fertiliser nitrogen application to be made generally either before or at-sowing for winter cereals. This research compares the relative nitrous oxide loss of two modified urea products versus conventional urea applied at sowing to a rainfed wheat crop in 2011.

Addition of Entec® substantially reduced nitrous oxides fluxes from fertiliser bands during the experimental period compared to untreated urea and Agrotain®. The highest flux measured with Entec® fluxes was 3.3 µg $N_2O$-N/$m^2$/hr however most were generally ≤ 1.4 µg $N_2O$-N/$m^2$/hr. Prior to 48 mm being received in late August, all treatments emitted similar amounts of nitrous oxide, however after that rainfall event both urea and Agrotain® treatments had nitrous oxide fluxes much higher than Entec®. Crop response in biomass and grain yield to the 60 kg N/ha applied nitrogen was relatively low (mean of 980 kg/ha grain or ≈ 20% increase in yield) suggesting the previous failed chickpea crop has carried forward a reasonable nitrogen supply for this crop.

## Methods

The experimental site was on the property "Colonsay" (27°28'S, 151°23'W) in the Formartin district of the Darling Downs, southern Queensland. The soil is a haplic, self-mulching, endohypersodic, black, Vertosol (Isbell 1996) of the Norillee Series (Beckmann and Thompson 1960). Sand, silt and clay contents in the surface 0-0.1 m are 31%, 12% and 57% respectively. Bulk density measures from analogue sites in Dalgliesh and Foale (1998) are 1.01 g/$cm^3$ for 0-0.1 m and 1.06 g/$cm^3$ for 0.1-0.3 m. During the 2010 winter season, chickpea (*Cicer arietinum*) were sown across the site however grain harvest was not possible due to a wet growing season overwhelming the crop with Botrytis grey mould (*Botrytis cinerea*).

On 27 June 2011 wheat (*Triticum aestivum* cv. Pacific Seeds LongReach "Spitfire") was sown at 45 kg/ha on 37.5 cm rows with 10 kg P/ha (as Incitec-Pivot Fertilisers triple superphosphate 20.7% P) and 10 kg S/ha (as Yara Nipro ammonium thiosulfate 16N 34S) placed in the seed row. Plot size was 20 m long and five wheat rows across with three replicates in blocks.

Nitrogen at 60 kg/ha was band applied between the wheat rows as either urea (46% N), Incitec Pivot Fertilisers Green Urea 14™ which is treated with the urease inhibitor N-(n-butyl) thiophosphoric triamide (NBPT, trade name Agrotain®) at 0.1% w/w active ingredient, or Incitec Pivot Fertilisers Entec® urea which has the nitrification inhibitor 3,4-Dimethylpyrazole phosphate (DMPP, trade name Entec®) added at 0.4% w/w active ingredient. An untreated control treatment receiving zero nitrogen was also established.

Two 25 cm PVC static gas chambers per plot were installed the same day as nitrogen application with one positioned over the sown wheat row and the other over the fertiliser band. Conveyor matting was laid as a walkway to facilitate moving between plots and improve access across the site during wet conditions. Marine ply boards (1.5 x 0.6 m) were laid from the matting to access the chambers themselves during sampling.

Initial gas monitoring covered the anticipated urea nitrification period and occurred 1, 3, 7, 11, 21 and 28 days after application. Subsequent sampling events were triggered when the site received at least 15 mm rainfall in the preceding 24-72 hour period. A PVC cap with gas sampling port was placed on the chamber to initiate the sampling period. At 0, 30 and 60 minute intervals approximately 20 mL of gas was extracted via syringe and placed into a evacuated 12 ml Exetainer® (Labco Limited). After collecting the 60 minute sample, caps were removed from the chambers. Sample analysis was conducted using gas chromatography, and flux data for carbon dioxide, methane and nitrous oxide was discarded if the Pearson's correlation coefficient ($r^2$) was <0.80 for carbon dioxide. Flux rates were calculated as in Barton *et al*. (2008) where the emissions are the linear increase in concentration during the measurement period. The flux of the wheat chamber was subtracted from that of the fertilised band to generate a net treatment flux. The flux of the control plot represents background loss.

Soil temperature was measured using a digital thermometer inserted to 0.05 m at the time of sampling at each set of chambers and the mean of all is reported here. Rainfall is as recorded at the Landmark "Colonsay" depot approximately 500 m from the field site. A soil sample (0-0.1 m) was collected in one replicate from each treatment at the first gas sampling time after the rainfall trigger was received. Samples were analysed for nitrate (method 7B1a) and ammonium (method 7C2a) (Rayment and Lyons 2011) and gravimetric moisture at 105°C and treatment means are reported.

Above-ground biomass was measured at physiological maturity by cutting a 1 m transect from the centre three rows of each plot and drying samples at 65°C for 48 hours. Grain yield was determined by machine harvesting all rows. Statistical analysis of dry matter and yield results was done using Genstat (VSN International 2011).

**Results and Discussion**

From establishment (27 June) to late August, rainfall measured at the site was low (Table 1) and so nitrous oxide flux was low despite the reasonably warm soil temperatures. Bureau of Meteorology records for Bowenville (Station 41008) report median rainfall for July and August of 26 mm each. In research on a similar soil type at nearby Warwick, Wang *et al*.(2011) indicated nitrous oxide losses commonly occurred under combinations of temperatures (air) exceeding 10$^{o}$C and water filled pore space (in the 0.07-0.13 m soil layer) exceeding 50%. These conditions appear to have been met with the 48 mm rainfall event prior to sampling in late August, after which nitrous oxide flux rates were highest from untreated urea (≈ 25 µg $N_2O$-N/m$^2$/hr) over the two sampling dates. Agrotain® flux was lower in comparison (≈ 6 µg $N_2O$-N/m$^2$/hr) suggesting some protection from denitrification loss, however Entec® treatments generated the lowest nitrous oxide flux with < 1.7 µg $N_2O$-N/m$^2$/hr release. The lower nitrous oxide flux and soil nitrate overall at the anthesis sampling event, triggered by rain in mid-October, suggest crop growth had recovered much of the applied nitrogen (Table 1).

Laboratory studies with a Wimmera district Vertosol in Victoria demonstrate 90% hydrolysis of urea after 6 days at 15$^{o}$C and 60% water filled pore space. In the same study, after 15 days at 25$^{o}$C only 40% of Agrotain® treated urea had hydrolysed when the experiment was terminated (Suter *et al*. 2011). The question emerging from that data is how long the treatment would continue to delay hydrolysis.

**Table 1: Rainfall, soil gravimetric moisture, soil temperature, soil nitrate and nitrous oxide fluxes from fertiliser bands in four treatments at "Colonsay", Darling Downs Qld during winter 2011**

| Date | Rainfall (mm) | Soil $H_2O$ (%) | Soil Temp (oC) | Soil NO3 (mg/kg) and $N_2O$ flux (µg $N_2O$-N/m$_2$/hr) | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Control | | Urea | | Agrotain | | Entec | |
| | | | | $NO_3$ | $N_2O$ | $NO_3$ | $N_2O$ | $NO_3$ | $N_2O$ | $NO_3$ | $N_2O$ |
| 28-Jun | 0 | 41 | 14.6 | 18 | 0.8 | 25 | 0.0 | 12 | 0.6 | 24 | 0.0 |
| 30-Jun | 2.5 | 39 | 13.0 | 18 | 0.0 | 18 | 0.0 | 26 | 0.0 | 27 | 0.0 |
| 04-Jul | 0 | 37 | 11.9 | 19 | 0.4 | 19 | 0.1 | 15 | 0.2 | 21 | 0.0 |
| 08-Jul | 0 | 30 | 9.7 | 15 | 0.0 | 30 | 0.0 | 18 | 0.8 | 16 | 0.0 |
| 18-Jul | 9 | 39 | 12.4 | 31 | 1.2 | 27 | 1.2 | 46 | 5.3 | 32 | 0.4 |
| 25-Jul | 0 | 30 | 10.2 | 20 | 0.0 | 26 | 2.2 | 20 | 0.1 | 22 | 0.0 |
| 29-Aug | 48.5 | 57 | 15.5 | 15 | 1.0 | 13 | 26.6 | 16 | 5.6 | 23 | 0.6 |
| 31-Aug | 0 | 57 | 15.3 | 29 | 2.3 | 42 | 23.7 | 65 | 6.4 | 26 | 0.9 |
| 17-Oct | 99 | - | 20.4 | 5 | 0.3 | 6 | 2.5 | 6 | 1.1 | 5 | 1.7 |
| 18-Oct | 0 | - | 18.5 | 4 | 2.6 | 10 | 2.5 | 6 | 2.8 | 5 | 1.0 |

While the biomass response to nitrogen treatment appeared minimal, grain yield suggested some fertiliser treatment effect compared to the control with a mean increase of 700 kg/ha (15%) from the three nitrogen treatments (Table 2). However this was not statistically significant. This grain yield response equates to an agronomic efficiency of 16 kg grain/kg N applied which is similar to results from short-fallow wheat grown in 2003 (Lester *et al*. 2008) and 2011 (D. Hall *pers comm*.) at the adjacent long-term nitrogen x phosphorus experimental site. Analysis of biomass for nitrogen uptake is currently underway and should provide further opportunity to compare nitrogen use efficiency.

**Table 2. Wheat biomass and grain yield in control (nil) or 60 kg N/ha applied as standard urea, Agrotain® or Entec® urea treated at "Colonsay", Darling Downs Qld.**

| | | Control | Urea | Agrotain | Entec |
|---|---|---|---|---|---|
| Biomass (kg/ha) | P=0.910 | 10957 | 11113 | 11683 | 11437 |
| Grain Yield (kg/ha) | P=0.177 | 4797 | 5727 | 5240 | 5537 |

## Conclusion

Entec® treatment resulted in the lowest nitrous oxide fluxes following fertiliser nitrogen application during this study. However, both Agrotain® and Entec® reduced nitrous oxide fluxes enough to encourage further agronomic and economic evaluation covering a range of seasonal conditions for use as a nitrogen fertiliser treatment in dryland cereal production across Australia's northern grains region.

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# Implications of soil morphological and chemical characteristics in the highlands of Central Province, Papua New Guinea for sustainable vegetable production

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Soil morphological and chemical characteristics determine the productive capacity and sustainability of land use. In Papua New Guinea (PNG), increasing production of temperate vegetables is needed to supply growing city population of Port Moresby (PoM) due to rural to urban migration, increasing expatriate populations, an emerging middle class and changes to food preferences in response to the recent mining and oil boom in PNG. Preliminary investigations show that soils of the cooler, moderate to high rainfall highland areas of Central Province are suitable for vegetable production. The major soils (andisols and ferrosols), have suitable structure, but have multiple nutrient deficiencies and are at risk of erosion. Initial studies of production of some several varieties of broccoli, cabbage, carrots and capsicum during 2011-12 at Tapini (altitude 1100 -1800 masl, temperature - maximum 19-23°C, minimum 9-12°C) and Sogeri (altitude 650 masl, temperature - maximum 27-30 °C, minimum 16–19 °C) produced acceptable yields of vegetables of high quality, for example broccoli - 2 300-6 000 kg/ha, cabbage 10 000-19 000kg/ha and carrot 900-1 200kg/ha at Tapini, and capsicum 5 000-6 700kg/ha at Sogeri, provided nutrient deficiencies were corrected. This work is being expanded and repeated in 2012-13, supported by detailed soil analyses. Production under low, medium and high input systems is being evaluated to provide information on soil and crop management practices that enhance sustainability. The findings are being used in development and extension activities to support sustainable expansion of vegetable production on contrasting soils in favourable temperature and rainfall environments.

# Influence of porosity and permeate suspended clay concentration on clay entrainment within a Red Ferrosol

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## Abstract

Soil pore size is an essential factor affecting pore blockage and subsequently saturated hydraulic conductivity (Ksat). In this study we examined the effect of soil bulk density on suspended clay movement in a Red Ferrosol. Water with different suspended clay concentrations (0, 5, 10 and 20 g $L^{-1}$) was allowed to infiltrate cylindrical soil cores packed at two bulk densities (1.0 and 1.2 g $cm^{-3}$). The Ksat was measured at both bulk densities and at all suspension concentrations. The outflow suspension concentration was also measured during the drainage of up to 10 pore volumes of suspension. Following application of the suspension, 2-cm long portions of undisturbed, drained soil cores were sampled from the top and bottom of the column and the soil-water characteristic measured. Low bulk density (1.0 g $cm^{-3}$) soil allowed easier passage of suspended clay than the soil at high bulk density (1.2 g $cm^{-3}$). Significant increases in water retention occurred near the soil surface (top of soil core) than subsurface (bottom of soil core). Soil-water retention measurements show that the pore blockage largely occurs in the larger conducting soil pores and takes place within close proximity of the soil surface. This suggests that dispersed clay is likely to become entrained in close proximity to its dispersal/infiltration point.

## Introduction

Soil macropores affect soil infiltration and saturated hydraulic conductivity (Ksat), whereby a minor increase in soil bulk density (decrease in soil macropores) causes a major decrease in Ksat. For example, Kim *et al*. (2010) studied the effect of compaction on soil macropores and related parameters and showed that an 8% increase in soil bulk density decreased saturated hydraulic conductivity by 69%, due to macroporosity decreasing by 70%. The entrainment of dispersed clay would also be expected to increase with compaction.

The main soil colloids are associated in soil aggregates, due to the cohesive nature of colloids, but are released following aggregate degradation (Oades, 1993). The main processes of soil aggregate degradation can be summarised as slaking, swelling and dispersion. The impact of clay swelling and dispersion on macroporosity, pore blockage and Ksat is widely documented; however, there are few studies explaining the impact of changes in porosity on the transport of the clay colloids and subsequent accumulation within soil layers. This study examines the effect of soil bulk density on suspended clay transport, colloid accumulation and the consequent effect on Ksat in soil cores using water containing different amounts of suspended clay.

## Methodology

To obtain contrasting clay suspensions, a soil with high clay content (Black Vertosol) was dried, sieved through a 2 mm sieve and 130 g oven-dried equivalent was transferred to a 500 $cm^3$ beaker with 400 $cm^3$ of tap water (electrical conductivity ~0.4 dS $m^{-1}$). Ultrasonic energy was applied to the suspension for 15 min to disperse soil aggregates. After the suspension had cooled and the large soil particles had settled, it was decanted into a 20 L plastic container. This procedure was repeated several times until 18 L of clay suspension was obtained. Four subsamples (each 100 $cm^3$) of the suspension were dried in an oven at 105 °C for 24 h to determine the clay concentration (90±4 g $L^{-1}$). This bulk suspension was diluted to prepare sufficient quantities of suspensions of three different concentrations (5 ± 1, 10 ± 2.1, and 20 ± 3.7 g $L^{-1}$). The sediment concentration of these suspensions was measured in the same way as for the initial bulk suspension.

Sufficient quantity of the surface 15 cm of a Red Ferrosol (Table 1) was collected from the Agricultural Field Station at the University of Southern Queensland, Toowoomba. This soil was air dried, passed through a 2 mm sieve and equilibrated overnight with tap water equivalent to 1.2 times the plastic limit (Misra and

Sivongxay, 2009). The soil was then packed into PVC tubes (8 cm height and 5 cm internal diameter) at two different bulk densities (1.0 and 1.2 g $cm^{-3}$) using a method similar to that described by Misra and Li (1996). After packing, the lower ends of the cores were supported by cheesecloth and an additional 2 cm PVC tube was attached to the top of the cores for water head application.

**Table 1: Selected physical and chemical properties of the Red Ferrosol used in this study**

| Properties | Mean value ± SE |
|---|---|
| Clay % | 44.1 ± 0.5 |
| Silt % | 25.5 ± 1 |
| Sand % | 30.4 ± 0.7 |
| $pH_{1:5water}$ | 5.8 ± 0.05 |
| $EC_{1:5}$ (electrical conductivity, dS $m^{-1}$) | 0.35 ± 0.001 |
| Exchange sodium percentage (ESP; %) | 3 ± 0.5 |
| Cation exchange capacity (meq 100 $g^{-1}$ soil) | 26 ± 1 |

*Saturated hydraulic conductivity measurement and leaching*

Four concentrations of suspended sediment solutions (tap water control and 5, 10 and 20 g $L^{-1}$) were applied to four replicates of the cores at each bulk density (total of 32 cores). The cores were supported and allowed to freely drain. A constant head of 1.5±0.3 cm was maintained during the application of 10 pore volumes of the suspended sediment. The leachate was collected and weighed to determine changes in Ksat and sediment outflow concentration. Clay concentrations were determined on 20 $cm^3$ aliquots of the outflow water after application of each pore volume.

*Soil water retention and soil bulk density measurement*

Core samples were taken from the upper and lower surface of the columns in the tap water (0 g $L^{-1}$), 5 and 20 g $L^{-1}$ sediment treatments after application of the suspensions to determine the effect of sediment concentration and initial soil bulk density on water retention and final soil bulk density. Aluminium rings (3 cm height, 3 cm diameter) of known weight were inserted from the top and the base of the cores (Fig. 1). The rings were then removed and supported with a piece of cheesecloth from below. The samples were saturated over a porous plate with tap water via capillary rise for 24 h. The weight of the ring before and after saturation was taken with an electronic balance (± 0.001 g). The samples were then equilibrated at -100, -200, -300 and -400 kPa water potential over 72 h using a pressure plate and the water content measured at each potential. After the final water content was measured, the soil was removed from the rings and dried in the oven at 105°C for 24 h. The weight of dry soil and the volume of the metal rings were used to determine the final soil bulk density after treatment.

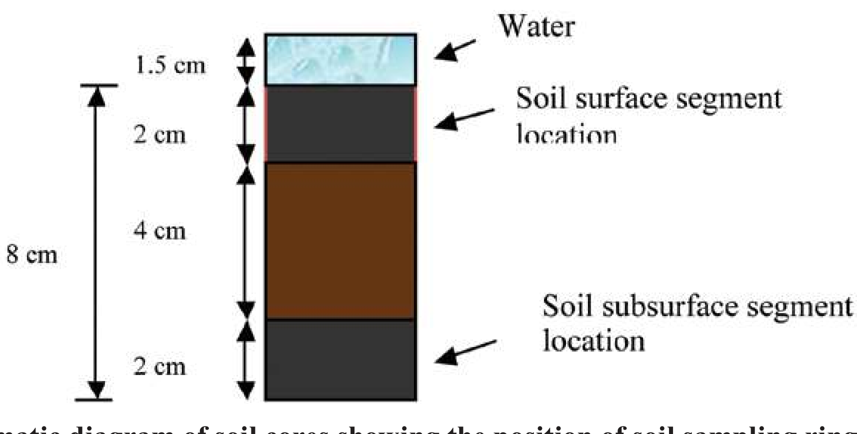


**Fig. 1: A schematic diagram of soil cores showing the position of soil sampling rings used for soil water retention measurements.**

## Result and discussion

*Effect on hydraulic conductivity and sediment in the outflow*

Ksat decreased with increasing suspended sediment concentration and increasing volume of water applied with a steady state Ksat generally observed after 6 to 8 pore volumes had been applied (Figure 2). The application of tap water to soil cores packed at 1 g $cm^{-3}$ (Fig. 2a) resulted in a small decrease in Ksat with volume of water applied and a steady state of ~700 mm $h^{-1}$ after 6 pore volumes had drained. However, applying the same water to a soil with a higher bulk density of 1.2 g $cm^{-3}$ (Fig. 2B) resulted in a Ksat of ~120 mm $h^{-1}$ after a similar volume was applied. The application of the 5 g $L^{-1}$ suspension resulted in only a relatively small decrease in Ksat in the 1 g $cm^{-1}$ core (~600 mm $h^{-1}$) but in the 1.2 g $cm^{-1}$ core there was a substantial relative decrease (~30 mm $h^{-1}$). Similarly, the 20 g $L^{-1}$ suspension produced a steady state Ksat of 46 and <1 mm $h^{-1}$ in the 1 and 1.2 g $cm^{-3}$ cores, respectively.

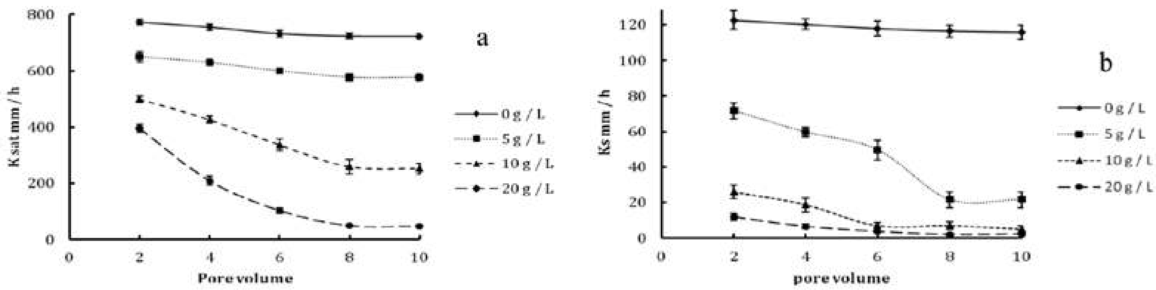


**Fig. 2: The effect of clay suspension concentration and volume of applied water on Ksat of a Ferrosol with bulk density of (a) 1 and (b) 1.2 g cm-3.**

Sediment concentration in the outflow indicated the relative entrapment of applied suspended sediment within the core. Suspensions applied at low concentration (e.g. 5 g $L^{-1}$) to the low density core were found to pass through the core without significant entrapment (Fig. 3a). However, evidence of sediment entrapment was observed at all applied suspension concentrations in the 1.2 g $cm^{-1}$ core (Fig. 3b) and where higher suspended sediment concentrations were applied to the lower density core (Fig. 3a). The effect was greater at high concentration with the outflow sediment reaching a steady state after 6 to 8 pore volumes of suspension had been drained.

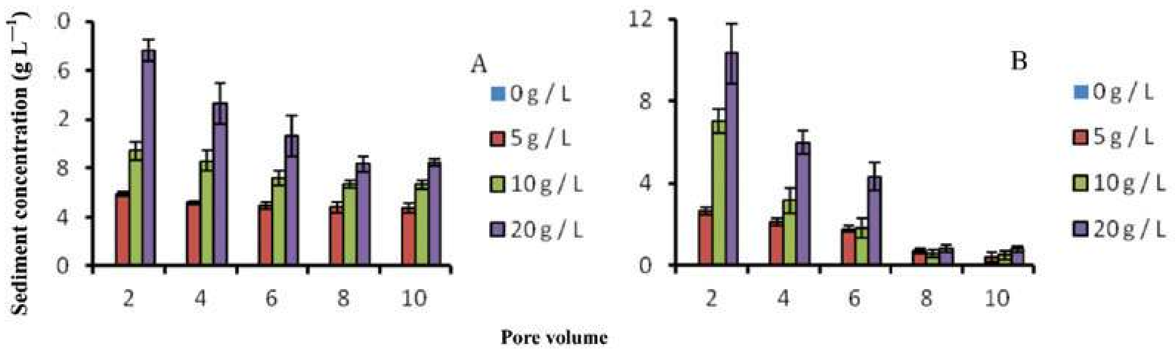


**Fig. 3: Effect of suspended sediment concentration for the applied water and pore volume applied on the outflow sediment concentration (+/- SE) from a Ferrosol core packed at (a) 1 and (b) 1.2 g $cm^{-3}$; no outflow sediment was evident in any of the 0 g/L leachates.**

*Effect on the soil-water characteristic*

The soil-water characteristic was affected by both soil density and the applied suspended sediment concentration (Figure 4). The soil surface of the 1.2 g $cm^{-3}$ soil (Fig. 4b1) retained more water than that of the low density (1 g $cm^{-3}$) soil at all potentials (Fig. 4a1). Increasing the sediment concentration in the applied water resulted in higher soil-water retention with increasing soil-water tension suggesting that there was a reduction in macropores and an increase in pores equivalent to these suctions. The effect of applied sediment concentration on soil-water characteristic was more pronounced in the surface than in the bottom of the cores (e.g. compare Fig4A1 to 4A2 and B1 to B2). This suggests that pore blockage due to entrainment of the suspended sediment in the applied water occurs primarily in the surface layers and that blockage of pores occurs predominately within larger pores. This is consistent with the notion that macropores are primarily responsible for hydraulic conductivity as soil-water potential approaches saturation.

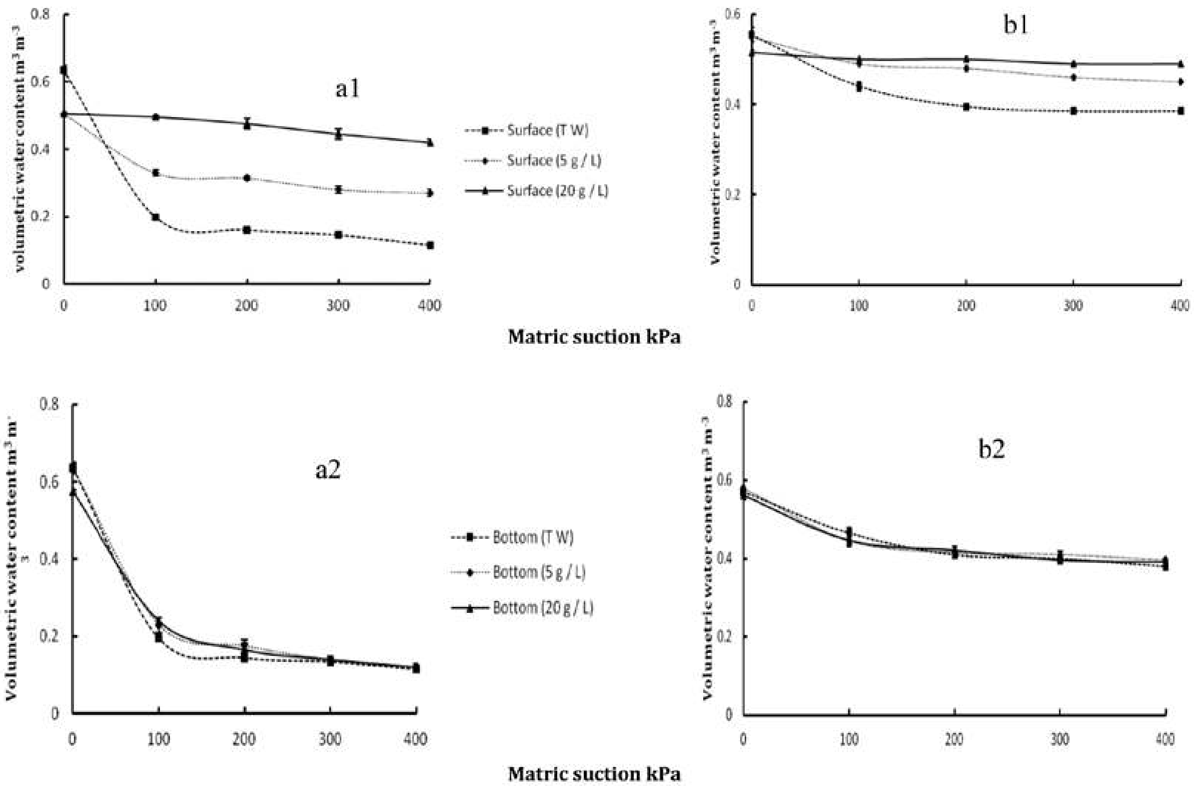


**Fig. 4: Effect of sediment concentration in water applied to a Ferrosol column packed at (a) 1 and (b) 1.2 g cm$^{-3}$ on the soil-water characteristic of samples taken from the (1) surface and (2) subsurface soil layers. Error bars are +/- SE.**

## Conclusion

This experiment confirms the importance of pore size on movement of dispersed clay within soil. The movement of the suspended sediment through the soil cores without entrapment was higher in the less dense cores. Increasing the soil bulk density creates a finer soil pore network causing clay to become more readily entrained within soil layers. Importantly, soil-water retention measurements show that the pore blockage largely occurs in the larger conducting soil pores and takes place within close proximity to the soil surface. This suggests that dispersed clay is likely to become entrained in close proximity to its dispersal/infiltration point.


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# The effects of lime, gypsum and lime/gypsum combinations, after 2.5 years, on two sodic soils under dryland cropping conditions in the Macquarie Valley, NSW

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## Abstract

While gypsum, and to a lesser extent lime, are reported as ameliorants for soil sodicity, there is relatively little information concerning the use of these ameliorants in combination. This study investigates the use of lime and gypsum combinations on selected soil properties relating to sodicity on two different soils used for dryland cropping in the Macquarie Valley of New South Wales (NSW). Lime and gypsum treatments were applied to two soils (lime [L] and/or gypsum [G], applied at t/ha rates: L0G0 [control], L2.5G0, L0G2.5, L2.5G2.5, L2.5G5, L5G2.5, and L5G5) that were sampled after 2.5 years and analysed for soil pH, EC and aggregate stability. Changes in pH and EC, as well as correlations, suggest that lime has dissolved more readily in the presence of gypsum. While aggregate stability was not significantly enhanced, significant relationships between soil EC and aggregate stability were found.

## Introduction

The use of lime and gypsum in the amelioration of sodicity under dryland conditions has been well documented (see Ghafoor *et al.* 2001). Even still, there remains a paucity of information concerning the field-based effects of lime and gypsum combinations on sodic soils, irrigated or otherwise (Bennett 2011). The higher dissolution rate of gypsum causes hydrogen ions ($H^+$) to become displaced from the negatively charged clay faces during exchange with $Ca^{2+}$, which slightly lowers the pH and increases the solubility of lime (Valzano *et al.* 2001). In the study of Valzano *et al.* (2001), improved soil stability, due to this effect, was still observed after three years and 650 mm of rainfall. Additionally, a slight, but significant, increase in EC was observed after the same timeframe and rainfall in soils with lime applied at 1 $t.ha^{-1}$ and a lime/gypsum combination applied at 2.5 $t.ha^{-1}$/1 $t.ha^{-1}$, respectively, as compared to the control soil. Because the combination of lime and gypsum has been observed to provide a synergistic ameliorative effect that is apparently soil specific, it is important to present further field-based research on a broader range of soils.

This study investigates the use of lime and gypsum combinations on selected soil properties relating to sodicity on two different soils used for dryland cropping in the Macquarie Valley of NSW.

## Methods

The experimental sites were situated in NSW near Warren ("Bellevue"; 31°32'00.70"S 147°47'46.13"E) and near Trangie ("Agriland"; GR 31°59'20.27"S 14°807'05.06"E) where full-field replicated experimental treatments of sodic ameliorants were applied. These experimental treatments consisted of lime (L) and/or gypsum (G), applied at t/ha rates: L0G0 (control), L2.5G0, L0G2.5, L2.5G2.5, L2.5G5, L5G2.5, and L5G5. Treatments were applied in August and September of 2007. The soil sampled from Bellevue is from an Episodic-Endocalcareous Brown Vertosol. Clay mineralogy suggests that the soil is illite and kaolinite dominant. While some smectite is present, the majority of this appears to be interstratified with illite. Soil sampled from Agriland is from a Calcic Sodic Brown Dermosol. The mineral suite of the Agriland soil is dominated by illite and kaolinite in the A horizon, while smectite is more prominent at depth.

Soil samples we taken 2.5 years post application of lime and/or gypsum using a single stroke, tractor mounted, hydraulic corer with a 45 mm internal diameter soil core. These were taken at approximately 16 m intervals from the furrow and parallel to the rows at 0–100, 100–200 and 200–400 mm depths. A total of 28 samples were taken per treatment, which equated to 392 samples sites, and 1,176 samples by depth, per property. These were air-dried and crushed to pass a 2 mm sieve, with a proportion of intact soil aggregates from each sample set aside. The analysis of the Agriland and Bellevue soil post-lime and gypsum application, included $pH_{1:5}$, $EC_{1:5}$ soil:water (Rayment and Higginson 1992) and aggregate stability in water (ASWAT, Field *et al.* 1997). These measurements were used as proximal indicators of soil stability and ameliorant dissolution in lieu of exchangeable/soluble cation data, which was beyond the scope of the work. Statistical analysis of these measurements was undertaken using analysis of variance, utilising Tukey's pairwise comparison and honest significant difference (h.s.d.), and correlation analysis.

## Results and discussion

*Treatment effects on, and relationship between, pH and electrolyte concentration*

Lime is known to increase the pH of soils through $Ca^{2+}$ exchange and the formation of bicarbonate ($HCO_3^-$), the rate of which depends on the rate of lime dissolution (Cregan *et al.* 1989), viz. the lime must dissolve in order for a change in pH to occur. The increases observed in pH for the Bellevue topsoil indicate that lime has dissolved throughout the 2.5 yr period since application and subsequently raised the pH (Fig.1). While the pH results for Agriland are not statistically significant, the slight rise in pH where lime has been applied could be attributed to lime dissolution (Fig.1). The general effect of gypsum on soil pH is a slight acidifying effect, due to displacement of $H^+$ (Valzano *et al.* 2001). While not a significant result in this study, the consistently lower mean pH value where gypsum alone was applied is most likely due to this process. This was more pronounced in the Bellvue soil than the Agriland soil, therefore it is likely that gypsum has had a longer lasting pH effect in the Bellevue soil (control initial pH of 7.1) as compared to the Agriland soil (control initial pH of 6.4). The presence of lime where gypsum is applied has also been shown to decrease the dissolution of gypsum (Naidu *et al.* 1993). Combining the effects of pH and the presence of lime with gypsum on dissolution longevity, this would explain the propensity of the L2.5G5 treated soil to have a significantly higher pH than the soil treated with L0G2.5 at Bellevue.

As lime dissolves it can also augment the EC of the soil (Naidu *et al.* 1993). Valzano *et al.* (2001) observed the application of lime (1 $t.ha^{-1}$) to have increased topsoil EC after three years and 650 mm of rain in a Brown Sodosol (pH approximately 6.0–6.3 in the topsoil). Similarly, increases in soil EC for the current study were observed where lime had been applied for both Agriland and Bellevue (Fig.2). For the Bellevue soil, the L2.5G2.5 and L2.5G5 treatments caused a significant increase in EC as compared to the L0G2.5 treatment ($0.15>p>0.05$). Whilst gypsum is more commonly known to increase EC than lime, it is likely that the EC effect due to gypsum has subsided due to leaching and the time afforded to dissolution (Sumner 1993). Interestingly, where gypsum was applied alone, the general effect was a slight decrease in EC. This can be attributed to a probable short-term improvement in soil structure, due to gypsum application, that has resulted in enhanced leaching of electrolyte over a longer period of time than lime applied alone could achieve (Valzano *et al.* 2001). This reinforces the short-term nature of the EC effect of gypsum and the need to reapply gypsum to maintain such an effect.

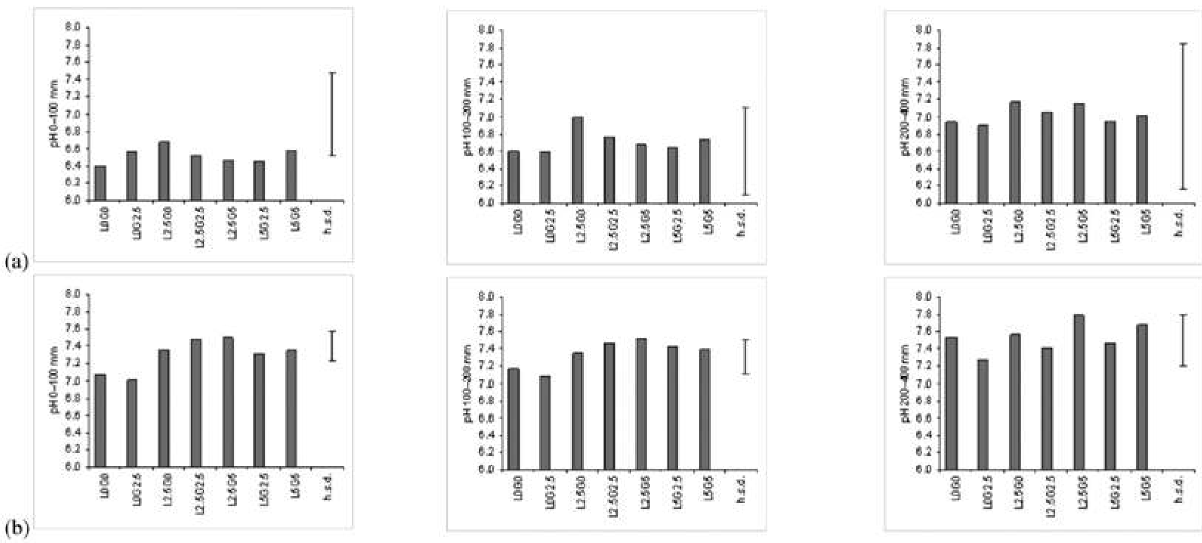


**Figure1: Changes in soil pH as a result of treatment effect after 2.5 years. Graphs in row (a) pertain to Agriland, and graphs in column (b) pertain to Bellev**ue

As lime is required to dissolve in order to provide a potential rise in pH, it stands that if lime is also responsible for increases in EC, pH should strongly and positively correlate with EC. This was the case for both the Agriland and Bellevue soils (Fig.3). Irrespective of statistical significance within each measurement (i.e. within pH or EC), a significant correlation is meaningful in that it shows the effect is a result of treatment application, and not just random 'noise'. Of the two soils, the relationship was most consistent in the Bellevue soil, which is probably because the pH is approximately half a unit higher in this soil, meaning that the rate of dissolution would be slower. During the lifetime of the experiment 1222 and 1322 mm of rain fell on Bellvue and Agriland, respectively, that would have also affected the dissolution rate of gypsum and the subsequent effects on lime. Thus, the longevity of the EC effect is more likely to be greater for Bellevue. This is not to say that an EC effect due to lime has not occurred at Agriland. Indeed, the 200–400 mm layer shows that a highly significant relationship between pH and EC exists. The fact that the upper layers do not show such a relationship is most likely a result of improved aggregate stability and increased leaching, as well as a higher concentration of electrolyte in the leachate, due to enhanced dissolution from a lower pH than the Bellevue soil.

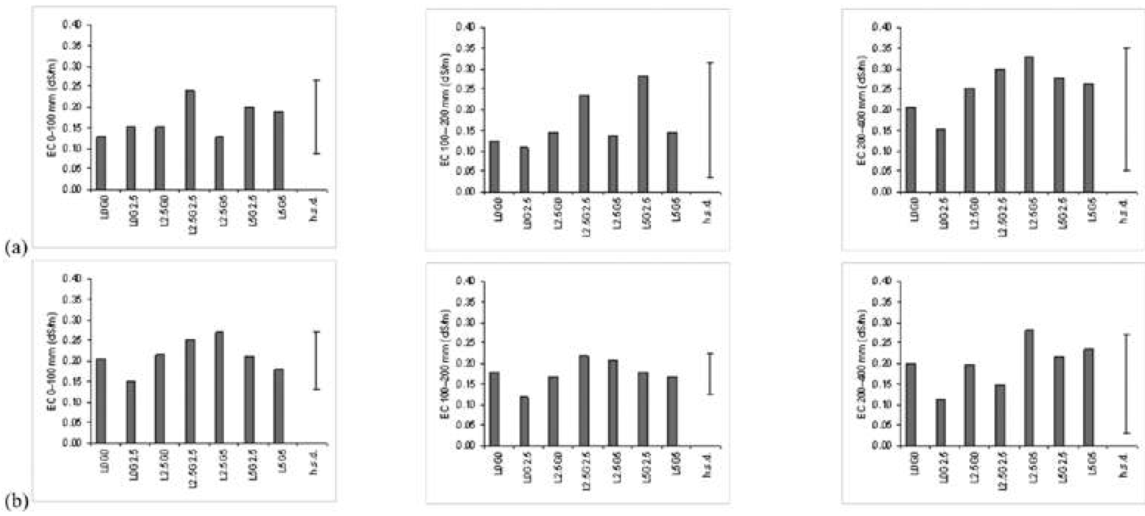

**Figure 2. Changes in electrolyte concentration as a result of treatment application after 2.5 years. Graphs in row (a) pertain to Agriland, and graphs in column (b) pertain to Bellevue**

*Treatment effect on aggregate stability*

While there were no significant effects on aggregate stability resulting from treatment application, there was an evident trend for aggregate stability to be enhanced where lime and gypsum combinations had been applied in comparison to the control soil (Fig.4). The variability in ASWAT data was high, which could have occurred as a result of a dilution effect, viz. A difference in ameliorant effect throughout the soil layer, especially as the boundary of the adjacent soil layer is approached (exacerbated by cut-and-fill land-forming techniques). For the Agriland soil, the L2.5G2.5 and L5G2.5 treated soils were observed to persist as the soils with the lowest DI, or equally the lowest DI, throughout the layers investigated. For the Bellevue soil, the treatment providing the most consistent decrease in DI was the L2.5G5 treatment; this is possibly due to a greater amount of gypsum required to augment lime dissolution at higher pH than that of the Agriland soil. For both soils, the treatments having the greatest effect on DI throughout the investigated profile were the higher EC treatments, as compared to the remaining treatments (Fig.4). Hence, aggregate stability is likely augmented by lime dissolution through the addition of greater amounts of electrolyte.

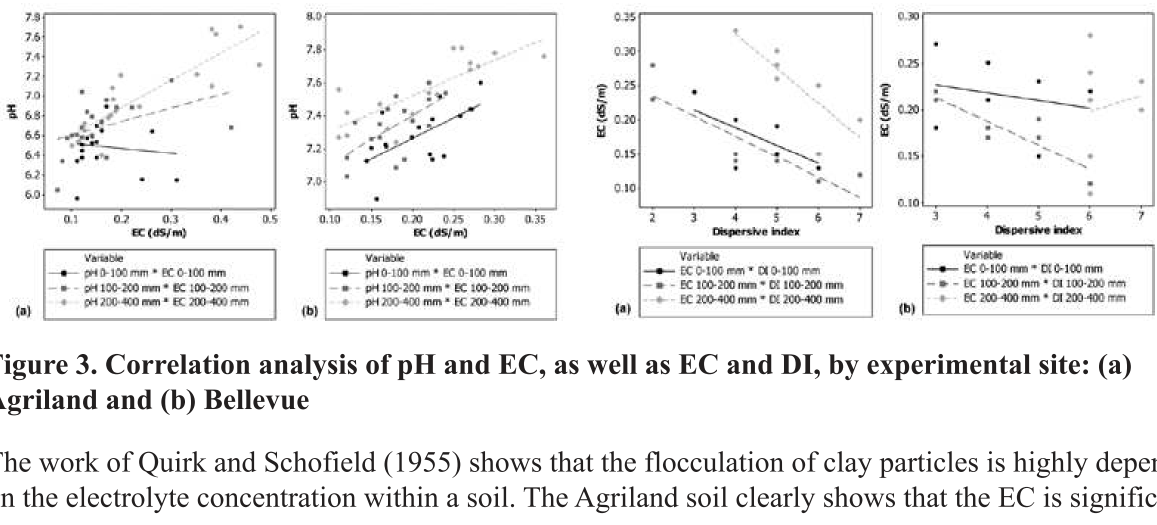

**Figure 3. Correlation analysis of pH and EC, as well as EC and DI, by experimental site: (a) Agriland and (b) Bellevue**

The work of Quirk and Schofield (1955) shows that the flocculation of clay particles is highly dependent on the electrolyte concentration within a soil. The Agriland soil clearly shows that the EC is significantly and negatively related to the dispersive potential of the soil for all layers, while only the 100–200 mm layer for the Bellevue soil shows a significant relationship (Fig.3). For the Bellevue soil, it is suggested that the recent cut-and-fill land-forming (in the year 2005, as compared to 1992 at Agriland) is responsible for the variation. It is probable that such cut/fill variability would mask the effects of treatment application, unless pronounced. Furthermore, fill areas would be more likely to benefit from a small rise in EC than would the exposed subsoil of cut areas with a higher ESP. This reinforces the fact that sodic soils, and their level of stability, are dependent on more than just ESP (Sumner 1992).

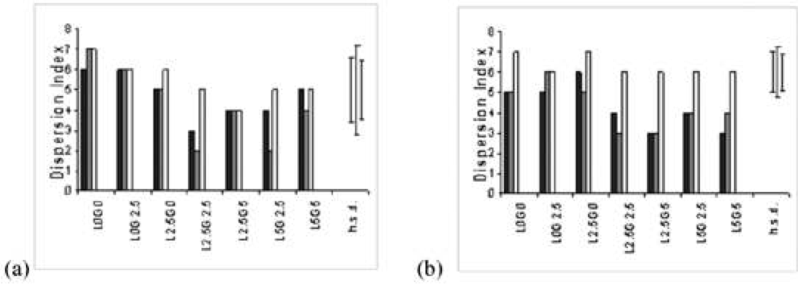


**Figure 4. Changes in aggregate stability in water as a result of treatment application after 2.5 years; (a) represents Agriland data; (b) represents Bellevue data; columns within treatments represent the 0–100 mm (black), 100–200 mm (grey) and 200–400 mm (white) layers; h.s.d. intervals are ordered relative to the layer order**.

*Possibility of a synergistic effect between lime and gypsum on soil sodicity*

As soil exchangeable cations were not determined, it is only possible to speculate on the effects of lime and gypsum combinations on soil sodicity levels. However, it is feasible that synergy between lime and gypsum has occurred. Where EC levels were significantly different, application of lime and gypsum had occurred in combination (Fig.2). Valzano *et al.* (2001) found a similar result for EC after three years on a Brown Sodosol where a L2.5G1 treatment had been applied. Thus, the observed EC levels in the current study may be indicative of a lower ESP. Furthermore, if lime is responsible for the maintenance of EC, and greater EC where applied in conjunction with gypsum, then it is quite probable that the soil solution is $Ca^{2+}$ enhanced where combinations are applied (Valzano *et al.* 2001). The Bellevue soil was the only soil with significant differences in EC after 2.5 years (Fig.2), and of the two soils, is the most likely to have benefitted from a synergistic effect, although this does not preclude the Agriland soil from such an interaction. As previously discussed, the lower pH in the Agriland soil (Fig.1) may have caused the amendments to have completely dissolved in before soil sampling occurred after 2.5 years. The EC/pH relationship in the 200–400 mm layer for this soil suggests that lime has had an effect on soil EC, albeit possibly dwindling (Fig.3). Therefore, the possibility of a lime/gypsum synergy after 2.5 years, while unable to be proven, should not be ruled out. Cation analysis of these soil samples should be undertaken in the future.

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## Salinity transects under an artesian bore drain

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More than xx km of bore drains exist in the Queensland Murray-Darling Basin. Bore drains are shallow (~0.5 m) V-shaped constructed channels used to route and supply water from artesian bores. Water losses in bore drains can exceed 90% and drains have been progressively closed down as part of a capping and piping program (GABSI). In the life of a bore and its drain network, tens of thousands of tonnes of salt have been deposited. Transects of soil cores were dug across two drains on contrasting soil types (saline Grey Vertosol, non-saline Red Kandosol) to evaluate the distribution of salts under a drain. Morphological attributes such as mottling were indicative of long-term saturation in and close to the drain. Salinity (conductivity and chloride) data suggests a horizontal wetting front with a slight rise in salinity within 4 m at the Red Kandosol, but the reverse (leaching) at the Grey Vertosol. pH and exchangeable sodium were substantially increased within 4 m of the drain at the Red Kandosol site, achieving values similar to the bore water chemistry. Despite the bore water having a high SAR, the ESP of the soils was not uniformly increased. Data suggests continual leaching of salts in both soil types but to a greater extent in the Red Kandosol – a not unexpected result given its lighter texture.

# Sheet erosion risk propagation for natural resource management decisions

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Efficacy of the Revised Universal Soil Loss Equation (RUSLE) has improved greatly in NSW with advances in the quality and resolution of static components: slope and slope-length (LS) factor from high resolution digital elevation models (25m pixels) and soil erodibility (K) factor from great soil group mapping. Improvements in the spatial and temporal resolution of cover (C) factor and rainfall erosivity (R) factor allow useful risk propagation products to be provided to land managers.

Rainfall erosivity has been derived from the SILO daily rainfall 5km grid (See Yang et al 2012) from 1890 to 2011. 1950-2010 has been chosen as the baseline due to better data quality. From this baseline cumulative frequency of rainfall erosivity can be calculated for each raster.

Monthly time series assessments of cover factor from February 2000 until February 2012 have been calculated from MODIS fractional cover (500m pixels) using method of Guerschwin etal (2009). Cumulative frequency of bare soil and hence cover factor have been calculated for each raster.

RUSLE-based sheet erosion has been calculated for each month from February 2000 as well as using rainfall erosivity probabilities. Results can be compared with sheet erosion thresholds for soil formation, maintenance of water quality or agricultural soil loss tolerance

Brief examples illustrate the how the application of resulting frequency and soil flux outputs directly relate to commonly used risk management matrixes are used.

# Interaction of pH and biochar determines the fate of $NH_4^+$ applied as di-ammonium phosphate in bauxite processing residue sand

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## Abstract

This incubation study has demonstrated that the interaction of the changes in base-acid reaction and adsorption capacity induced by greenwaste biochar addition and the original pH in bauxite residue sand determined the availability and dynamics of $NH_4^+$-N/ $NH_3$-N in bauxite residue sand amended with di-ammonium phosphate.

## Key Words

Biochar; ammonia volatilization; pH; bauxite residue sand; adsorption

## Introduction

A large quantity of highly alkaline bauxite residue is produced as a by-product during the Bayer process which is used to refine bauxite and produce alumina. The bauxite residue sand fraction (> 150 µm, BRS) represents the primary plant growth material for rehabilitating the residue storage areas. However, the inherent hostile characteristics (high alkalinity, high salinity, poor nutrient availability, high hydraulic conductivity and low nutrient retention capacity) of BRS provide significant challenges to successfully establish and maintain vegetation cover (Phillips and Chen 2010). Previous incubation studies have shown that the majority of nitrogen (N) applied as di-ammonium phosphate (DAP) is lost via $NH_3$ volatilization in BRS within 7 days following N application (Chen *et al*. 2010). Biochar is a stable carbon (C)-rich material derived from biomass heating processes in an oxygen-limited environment and is considered to have large surface area and a range of different functional groups on the surface and thus a high nutrient retention capacity (DeLuca *et al*; 2009; Taghizadeh-Toosi *et al*. 2012a). The objective of this study was to assess the interactive effects of biochar on pH and subsequent N availability in BRS.

## Materials and Methods

The BRS material was collected from Alcoa's Pinjarra Refinery in south-west Western Australia. The sample was oven-dried (25°C for 48 h) and passed through a 2-mm sieve. Materials < 2 mm were used in subsequent incubation experiments. The BRS had a high pH (11.8; 1:5 BRS:water ratio), a high EC (3.4 dS/m; 1:5 BRS:water ratio) and elevated exchangeable Na (12.8 cmol $kg^{-1}$), but had low C and nutrient contents (0.07 g $kg^{-1}$ C, 1 mg $kg^{-1}$ $NH_4^+$-N, 2 mg $kg^{-1}$ Colwell P). The greenwaste biochar (produced from greenwaste at 450°C in a low temperature pyrolysis plant by BEST Energies Australia) exhibited a high total C (685 g $kg^{-1}$), Colwell P (400 mg $kg^{-1}$) and pH (9.9), but low total N (3 g $kg^{-1}$), $NH_4^+$-N (< 0.3 mg $kg^{-1}$), $NO_3^-$-N (< 0.2 mg $kg^{-1}$), CEC (24 cmol $kg^{-1}$) and exchangeable Na (2.4 cmol $kg^{-1}$).

The pH of the BRS was then adjusted with 2.453 M hydrochloric acid to achieve a pH of 5.2, 7.1, 8.1 or 9.1 according to Chen *et al*. (2010). These pH adjusted BRS materials were used for the incubation experiment and the pH treatments designated as pH 5, 7, 8, and 9. Distilled water was added to adjust the moisture content of the BRS to 55% water holding capacity (WHC) after taking into account the volume of DAP added. This incubation experiment included 4 pH treatments (5, 7, 8 and 9), 5 rates of biochar (0, 1, 5, 10 and 20% of BRS, w/w) and 1 rate of DAP (equivalent to 5.4 tons $ha^{-1}$). Ammonia volatilisation was measured using the sponge-trapping and KCl extraction method described by Chen *et al*. (2010). This experimental design was completely randomised with three replications. The jars were randomly arranged and incubated at a constant 25°C for the duration of the experiment. The sponges were sampled after each incubation period of 4 and 24 h and 3, 7, 14, 21, 28, 35, 42, 49, 56 and 63 days, respectively. The modified sponge-trapping and KCl-extraction method described by Chen *et al*. (2010) was used for measuring $NH_3$ volatilisation. Residual $NH_4^+$–N ($NO_3^-$-N undetected) in BRS was determined at the end of the experiment using double 2M KCl extraction. This N fraction is considered to be weakly adsorbed $NH_4^+$/$NH_3$. Nitrogen not extracted by 2M KCl was estimated as the difference between the total amount of N added and the sum of $NH_3$ volatilisation and KCl extractable N. This unextracted N pool was considered as strongly adsorbed $NH_4^+$–N /$NH_3$-N by biochar/BRS.

**Results**

Two-way ANOVA results showed that there were significant interactions ($P < 0.01$) between pH and biochar addition for cumulative $NH_3$ volatilization, extractable $NH_4^+$–N, N recovery% and unextracted N (data not shown). The pH and biochar treatments had significant effects on these parameters as well ($P < 0.01$) (data not shown). Biochar addition increased BRS pH, particularly in the low pH treatment (e.g. pH 5) and at high rates of biochar (i.e. 10-20%) (Table 1). However, in the pH 9 treatment BRS pH was not related to the rate of biochar (Table 1).

**Table 1 Bauxite residue sand pH at the end of incubation experiment as affected by addition of biochar.**

| Initial pH | 5 | | 7 | | 8 | | 9 | |
|---|---|---|---|---|---|---|---|---|
| Biochar % (w/w) | Mean | SD | Mean | SD | Mean | SD | Mean | SD |
| 0 | 4.99 | 0.74 | 6.96 | 0.36 | 7.28 | 0.16 | 7.02 | 0.56 |
| 1 | 5.28 | 0.23 | 6.89 | 0.39 | 7.11 | 0.26 | 6.82 | 0.40 |
| 5 | 5.15 | 0.86 | 7.43 | 0.25 | 7.27 | 0.26 | 7.79 | 0.47 |
| 10 | 6.29 | 0.09 | 7.57 | 0.31 | 7.72 | 0.19 | 7.31 | 0.16 |
| 20 | 6.88 | 0.28 | 7.84 | 0.27 | 7.73 | 0.17 | 7.38 | 0.36 |

At low initial pH (5), cumulative $NH_3$ volatilization increased significantly but weakly-adsorbed $NH_4^+$–N /$NH_3$-N decreased with the rate of biochar addition over the whole experimental period (63 day incubation). This coincided with the increase in strongly-adsorbed $NH_4^+$–N /$NH_3$-N (Figures 1, 2). At pH 7 and 8 treatments cumulative $NH_3$ volatilization decreased with the rate of biochar addition, while both weakly-adsorbed and strongly-adsorbed $NH_4^+$–N /$NH_3$-N increased (Figure 2). In contrast, at high pH (9) treatment, it appears that biochar addition did not have significant impacts on cumulative $NH_3$ volatilization and $NH_4^+$–N /$NH_3$-N adsorption.

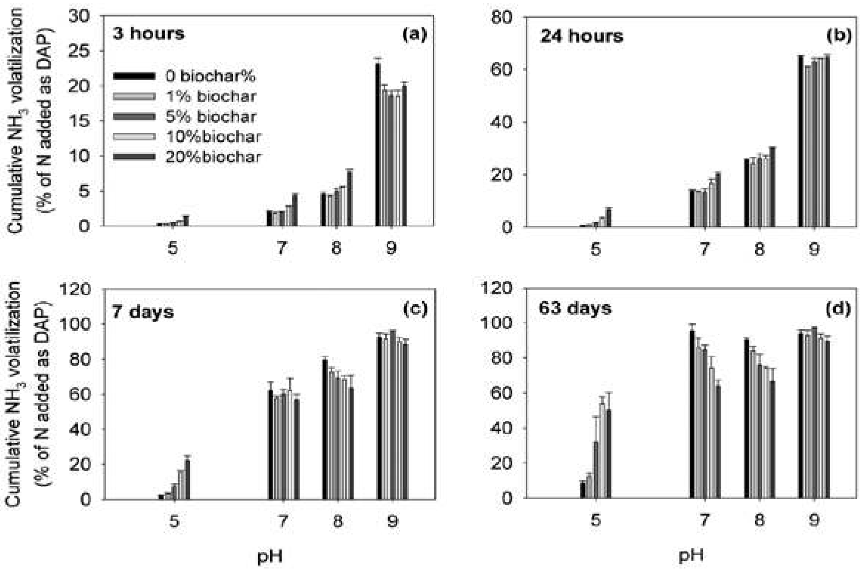


**Figure 1: Cumulative NH3 volatilised at 3 hours, 24 hours, 7 days and 63 days following the incubation in the BRS as affected by addition of biochar and pH treatment. The vertical bars indicate the standard errors of the mean (n=3).**

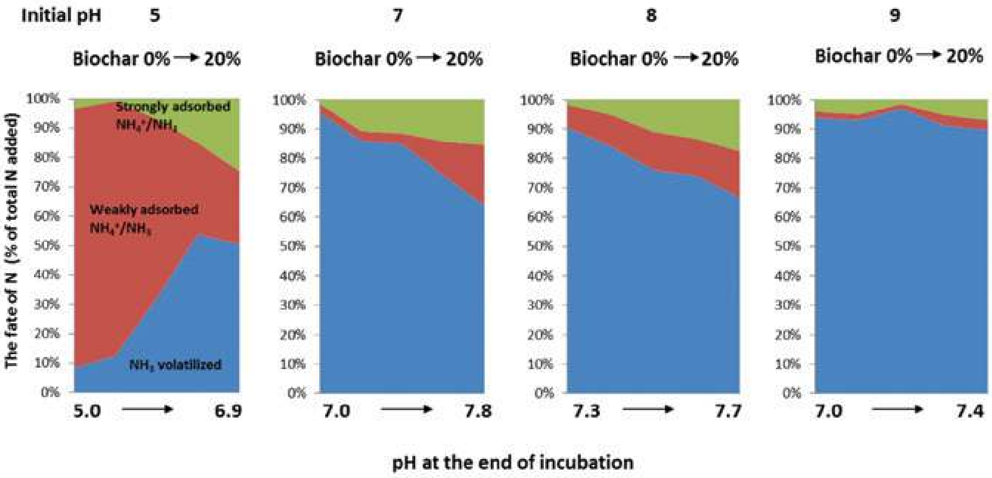


**Figure 2: Interactive effects of biochar addition and initial pH treatments on the fate of $NH_4$+-N added in bauxite residue sand.**

In the pH 5 treatments, increasing pH (rather than adsorption capacity) resulting from biochar addition drove greater $NH_3$ volatilization ($NH_4^+ + OH^- > NH_3 + H_2O$) and lower adsorption (weakly) of $NH_4^+$-N / $NH_3$-N (lower availability of $NH_4^+$-N /$NH_3$-N) with the rate of biochar addition (Figures 1, 2). In the pH 7 and 8 treatments, increasing adsorption capacity played a dominant role over the pH in driving greater adsorption (weakly) of $NH_4^+$-N /$NH_3$-N and lower $NH_3$ volatilization (lower availability of $NH_4^+$-N / $NH_3$-N). In the pH 9 treatments, the high initial pH (9) drove the loss of the majority of $NH_4^+$-N /$NH_3$-N pools via $NH_3$ volatilization. In all pH treatments, biochar addition generally increased the strongly adsorbed $NH_4^+$-N /$NH_3$-N possibly due to the trapping of $NH_4^+$-N /$NH_3$-N in micro pores (nanometre-size) formed in the internal layers of biochar during the pyrolysis (e.g. Chan and Xu 2009). Moreover, specific complexation of $NH_4^+$-N /$NH_3$-N by organic ligands on biochar surface may also be responsible.

## Conclusions

Results from the present study have clearly demonstrated that the interaction of changes in pH and adsorption capacity induced by greenwaste biochar addition affect the availability and dynamics of $NH_4^+$-N/ $NH_3$-N in BRS amended with DAP. In the BRS with low pH (5), increasing pH rather than adsorption capacity resulting from biochar addition accelerated greater $NH_3$ volatilization, leading to lower availability of $NH_4^+$-N /$NH_3$-N for adsorption. In the BRS with medium pH (7, 8), increasing adsorption capacity induced by biochar addition played a dominant role over the pH in enhancing adsorption of $NH_4^+$-N /$NH_3$-N and lowering $NH_3$ volatilization. In the BRS with high pH (9), the majority of $NH_4^+$-N / $NH_3$-N pools was lost via $NH_3$ volatilization due to the strong acid-base reaction at this pH. Further study is needed to develop and evaluate biochar materials to optimise N retention in DAP amended BRS.

## Acknowledgements

We would like to thank Ms L Hosseinzadeh and M Heenan for their assistance in the experiment and chemical analysis. This research was supported under Australian Research Council's *Linkage Projects* funding scheme (project number LP0989670, FT0990547) and by Alcoa World Alumina, Australia. Associate Professor CR Chen is the recipient of an *Australian Research Council Future Fellowship* (project number FT0990547).

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# Environmental impact assessment of climate change in Nigeria: Effects on natural environment

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The explosion in human population result in excessive land use and increased human impact on the environment, these increases are noticed by the scientist, the technician and even the layman. In an era when human economic activities affect all areas of the environment, it behooves on the government and organizations to plan on the efficient land use techniques and methods of management. Climate change is reported as the greatest and most serious environmental challenge facing the world in the 21st century. Floods hurricanes and drought account for 75% of the world's natural disasters and manifestations of global warming and climate change range from extreme weather events in various parts of the world, these changes include higher frequency of formation of cyclonic storm surges, high intensity of rainfall along with changing precipitation patterns, prolonged drought to hurricanes landslides and wild fires. These unpredictable patterns of weather and climate issues are of significance and projected to have adverse consequences on the Nigerian environment with specific impact on precipitation and water resources, agricultural productivity, and natural marine and terrestrial systems. Also, there are indications that developing countries like Nigeria are particularly vulnerable to climate change impacts because they have little adaptive capacity. Most cities in the Western part of Nigeria have experience the greatest impact of climate change on the natural environment.

**Keywords:** climate change, cyclonic storms, drought and flooding, natural environment, Nigeria

# Rapid evaluation of acid sulfate soils in the floodplain wetlands of the Murray-Darling Basin using a simplified incubation method

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This abstract presents a summary of a recently developed methodology for the rapid assessment of Acid Sulfate Soils (ASS) (Creeper et al. 2102a) and the application of this method to investigate the extent of Inland ASS (IASS) in the floodplain wetlands of the Murray-Darling Basin (MDB) (Creeper *et al.* 2012b). The simplified incubation method adequately classifies ASS materials in a timely manner and functions as a suitable alternative in projects where incubating until a stable pH is obtained (Sullivan et al. 2009) is not possible due to logistical or time constraints. Hence, the simplified incubation method is ideally suited for projects where large numbers of samples require rapid analysis.

The proposed incubation method was successfully used in the MDB ASS risk assessment project (MDBA 2011), to obtain pH measurements from ca 8,000 samples, representing approximately 2,500 profiles from 1,055 geo-referenced wetlands located in the floodplains of the MDB. These results were then used to investigate the occurrence of IASS in the floodplain wetlands of the MDB. Basin-wide, a total of 238 floodplain wetlands of the 1,055 wetlands assessed (23%) were found to present an IASS related acidification hazard. However, the proportion of wetlands that presented an IASS related acidification hazard varied with location in the MDB (ca. 0% to 52% of total). Regions located in the southern MDB, downstream of the Murray-Darling confluence, and in catchments located on the southern side of the Murray River channel in Victoria were found to present the greatest IASS related acidification hazard.

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## Six years of results from a potato rotation and green manure trial in Tasmania

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Irrigated cropping of vegetables and poppies is an important farming system in north-west Tasmania. Green manure crops are often grown between cash crops to provide ground cover and organic matter and to retrieve subsoil nitrogen. A field trial was started in 2006 to assess the long-term effects of three autumn-winter land uses: (a) fallow, and (b) annual ryegrass and (c) brassica (BioQure BQ Mulch™) green manures on cash crop production, soil-borne disease inoculum, and other soil properties. The three green manure treatments were established each autumn on plots of about 3000 m$^2$ in a randomized block design. A cash crop was grown over all plots in each spring-summer period except in 2009/10 when plots were split to accommodate two cash crops, potatoes and onions. Because potatoes were also grown over the whole trial in 2006/07 and 2012/13, the split in 2009/10 allowed 3- and 6-year potato rotations to be compared. These represent rotation lengths which are short and average for potatoes in the region, and may influence the incidence and severity of soil-borne potato diseases powdery scab, common scab, black scurf and stem canker. Concentrations of DNA of the pathogens causing these diseases have been measured in the soil each year, while soil physical and chemical properties have been measured in 2006, 2009 and 2012. These data are presented and discussed.

# Hydrogeological landscapes to describe salinity and sodicity hazard in Central West NSW

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Government Agencies including Catchment Management Authorities are challenged to include a consideration of salinity processes and implications as part of a whole of catchment natural resource management. NRM activities including soil management, farming systems, vegetation management, management of environmental flows within river flow regime management and community education and support are impacted by salinity considerations.

Salinity assessment techniques which have been successfully applied in landscapes with high relief and high rainfall are challenged by landscapes seen in western NSW. Soil salt stores are extremely variable. The mobility of salt stores and the impact of salinity issues is also poorly understood or described. Data sources traditionally used for salinity assessment are not available or not as relevant in the western part of the Central West CMA.

Catchment Management Authority staff can benefit from information products which demonstrate landscape processes at an implementation scale and describe comparative risk and hazard at a strategic scale. CMAs can also benefit from products which demonstrate important knowledge gaps

The Central West CMA has commissioned the development of HGL products for western parts of their area.

HGL products have been developed for a range of landscapes in NSW – although up til now they have been focussed on areas of high relief and high rainfall

The HGL product describes salinity processes, hazard, salt store, salt mobility, impacts felt within the landscape, catchment scale impacts. An important feature of this product is descriptions of the interplay between salinity and sodicity as a landscape/soil characteristic.

# Maximum axial growth pressures of the primary and lateral roots of woody perennials and annual crop

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The ability of woody perennials (WPs) to penetrate hard, dry soils in nature may depend on their ability to overcome high soil resistance by exerting high axial root growth pressures ($\sigma_{max}$). This ability may depend on the characteristics of their seeds (e.g. seed size, natural habitat, etc), seedling age, and soil physical properties. To evaluate this idea we measured the diameter (d) and the maximum axial root growth forces ($F_{max}$) exerted by a single lateral root axis of 3-4 months old seedlings of several WPs originating from different habitats. We also measured (d) and $F_{max}$ on the primary root axes of a large-seeded WP and an annual plant (AP) seedlings of 2 to 10 days old. Values of d and $F_{max}$ diameter for the same root were used to estimate $\sigma_{max}$.

We found that four eucalypt species originating from dry habitats (e.g. *Eucalyptus leucoxylon, E. platypus, E. loxophleba and E. kochii*) exerted significantly ($P \leq 0.05$) greater $\sigma_{max}$ than two species originating from wetter habitats (e.g. *E. robusta* and *E. camaldulensis*). We also found that species growing on fine-textured soils (e.g. *E. platypus* and *E. loxophleba*) exerted somewhat greater $\sigma_{max}$ than species growing on coarse-textured soils (e.g. *E. kochii*). There was no significant difference in $\sigma_{max}$ exerted by AN (e.g. *Pisum. sativum*) and four WPs (e.g. *Acacia salicina, E. leucoxylon, E. platypus* and *E. loxophleba*) and older seedlings exerted greater $\sigma_{max}$. WPs originating from dryer climates on fine-textured soils exerted higher $\sigma_{max}$ than WPs originating from wetter regions, which may explain their wider distribution on hard soils.

# Hydropedology of Tasmanian texture-contrast soils.

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## Abstract

Texture-contrast soils are extensively used for agricultural production in Australia; however, the processes governing infiltration and soil water distribution in these soils are poorly understood. This manuscript reports surprisingly complex interactions between soil morphology, antecedent soil moisture and preferential flow processes in a series of texture contrast soils in southern Tasmania. The occurrence of preferential flow was strongly influenced by antecedent soil moisture, not rainfall intensity as is often reported in the literature. In dry soil conditions, infiltration resulted from up to five different forms of preferential flow including finger flow, rivulet flow, macropore flow and saturated backfilling of voids at the base of the soil profile. At moisture contents near field capacity, infiltration was mostly uniform; however, wetting front instability and lateral flow in the A1 horizon developed due to difficulty displacing existing soil moisture further down the soil profile.

## Key Words

Pedology, duplex, macropore flow, finger flow, preferential flow, subsurface lateral flow

## Introduction

Hydropedology is an emerging field of science in which traditional disciplines of pedology, hydrology and geomorphology are integrated to provide better understand the two-way relationship between soil morphology and the processes governing soil water movement (Lin 2011). This paper details results from intensive hydropedological investigation of a series of texture-contrast soils at the University of Tasmania farm, Australia. Texture-contrast, or duplex, soils typically consist of sandy or sandy loam surface soil overlying a light to medium clay subsoil. They are estimated to occupy 17.0 % of the Australian landmass, and around 80 % of agricultural regions in southern Australia (Chittleborough *et al.* 1994). Unlike other soil types, texture-contrast soils contain an abrupt increase in clay content between the topsoil and the subsoil, which results in the formation of seasonal perched watertables and subsurface lateral flow (Cox and McFarlane 1995; Eastham *et al.* 2000).

## Material and Methods

The hydropedology of four texture-contrast soil profiles was investigated at the University of Tasmania farm, Cambridge, Tasmania. Soil profiles generally consisted of an aeolian sandy loam A1 horizon which overlayed a bleached A2e horizon with clear to sharp boundary to a dispersive, vertic, mottled, light-medium clay B2 horizons. Sites A, B and C, were located on soils developed from Pleistocene alluvial fan deposits, whilst site D was located on soils derived from Tertiary sediments (Holz 1993). Site A was classified as a Bleached Sodic, Natric, Brown Sodosol (Isbell 2002), Mollic Natrustalf (Soil Survey Staff 2006), while sites B, C and D were classified as bleached Sodic, Natric, Brown Kurosols (Isbell 2002), Mollic Natrustalfs (Soil Survey Staff 2006).

Soil chemical attributes, including pH, CEC, organic carbon, ESP (excluding $Al^{3+}$), chloride, EC, and macro nutrients, were determined every 10 cm to 120 cm depth. Soil physical attributes, including particle size, linear shrinkage, Atterberg limits, and dispersion, were also determined for each horizon. Soil hydrological properties, including the soil water characteristic, soil shrinkage characteristic curve and saturated hydraulic conductivity, were determined on intact cores. *In situ* infiltration and hydraulic conductivity was determined using tension infiltrometers at six supply potentials for each soil horizon in both 'dry' and 'wet' soil conditions. The effect of water repellence on infiltration was investigated by comparing infiltration of water to that of a 7M ethanol solution. Development of flow instability was investigated in repacked water repellent, and wettable soils within a Hele-Shaw Chamber. Potential water repellence was monitored over a 12 month period as water drop penetration time (WDPT) and water entry potential (WEP) on air-dried aggregates. Dye tracer studies were conducted to determine the effect of soil morphology on infiltration and preferential flow processes. Brilliant Blue dye tracer (4 g $L^{-1}$) was applied to a 25 mm simulated rainfall event at each of the four sites, in both wet and dry soil conditions.

**Results and discussion**

Differences in soil morphology, structure, horizons, chemistry and infiltration patterns existed between the four sites located within 400 metres of each other (Table 1). The aeolian derived A1 and A2e horizons had similar chemical properties across all sites. However, the depth and continuity of the A2e horizon differed markedly between sites. The A2 horizon was thickest at site B in which sand infills extended from the lower A2e horizon to 70–80 cm depth. Whilst at sites A and C the A2e horizon was discontinuous, ranging from 0–2 cm thickness. The A2e and sand infills contained up to 87.4 % sand and were highly leached. Differences between sites were most pronounced in the B horizons. At site B, the B2 horizons consisted of strong coarse, prismatic structure, separated by deep sand infills. At sites A and C the columnar or prismatic structure was weak, parting to 20-50 mm angular blocky structure, whilst at site D there was no evidence of large columnar or prismatic structure. Subsoils were vertic, linear shrinkage varied from 6.4 % in the B21 horizon at site D, to 17.4 % in the B22 horizon at site C, and slickensides were noted at all sites. Subsoils at all sites were dispersive (Emerson classes 1, 2 or 3) and sodic. The presence of exchangeable aluminium did not appear to prevent dispersion, as site B had both the highest levels of exchangeable aluminium and lowest (most dispersive) Emerson class. Trend in EC, chloride, ESP, pH and exchangeable acidity indicated that site A (lower slope position) had been subjected to considerably less leaching than the other three sites (Table 1).

Dye tracer studies demonstrated that differences in soil structure, chemical attributes or horizon thickness had little effect on the maximum depth of infiltration, or the proportion of soil which participated in flow (Figure 1). Antecedent soil moisture profoundly influenced the depth, rate and occurrence of preferential flow at all four sites. At low antecedent soil moisture, a combination of preferential flow processes resulted in infiltration of the dye tracer to an average depth of 103 cm in which as much as 99.8 % of the soil matrix was bypassed in the lower A1 horizon (site C) and 99.4 % in the B21 horizon (site A). However, at high antecedent soil moisture content, the same volume of dye tracer infiltrated to an average depth of only 35 cm, by mostly uniform processes (Figure 1).

**Table 1. Selected chemical and physical attributes.**

| Site | Depth (cm) | Clay (%) | Organic Carbon (%) | pH ($H_20$) | CEC (meq/100 g) | ESP (%) | Kunsat □ -0.1 kPa (mm $hr^{-1}$) Dry | Kunsat □ -0.1 kPa (mm $hr^{-1}$) Wet | Subsoil structure | A2e depth (cm) |
|---|---|---|---|---|---|---|---|---|---|---|
| A | 0-10 | 8.0 | 1.83 | 6.5 | 1.9 | 4.3 | 45.57 | 14.57 | Weak 20- 50 mm angular blocky | 0-2 |
| | 60-70 | 46.4 | 0.21 | 7.8 | 6.6 | 19.4 | 5.65 | 5.36 | | |
| B | 0-10 | 8.2 | 2.39 | 5.4 | 2.7 | 1.8 | 48.39 | 23.51 | Columnar, parting to moderate 50 – 100 mm subangular blocky | 7-12 |
| | 60-70 | 35.0 | 0.34 | 5.5 | 6.9 | 7.63 | 5.74 | 1.62 | | |
| C | 0-10 | 10.3 | 1.29 | 5.4 | 2.7 | 4.85 | 30.78 | 23.11 | Medium 10-20 mm coarse angular blocky | 0-2 |
| | 60-70 | 39.3 | 0.34 | 5.2 | 7.6 | 9.4 | 2.46 | 3.81 | | |
| D | 0-10 | 17.1 | 2.25 | 6.8 | 7.1 | 0.8 | 20.71 | 6.20 | Weak 10-20 mm coarse angular blocky | 0 (<5 pockets) |
| | 60-70 | 49.8 | 0.32 | 5.4 | 10.0 | 8.9 | 12.57 | 4.39 | | |

*Infiltration into the A1 horizon*

Infiltration into the A1 horizon was strongly influenced by water repellence and antecedent soil moisture. Unlike most water repellent soils, both the severity and persistence of potential water repellence (air-dried) vary seasonally (WDPT 0.19 min in July 2009 to 54 min in May 2010) in which water repellence was not re-established upon drying. Laboratory leaching experiments indicated that the 200-fold seasonal reduction in WDPT resulted from winter rainfall leaching the water repellent compounds (presumably polar waxes) from the soil. In dry soil conditions, water repellence restricted infiltration through large macropores (>500 μm), which decreased intrinsic permeability by one to two orders of magnitude. Water repellence induced wetting front instability, or fingering, which reduced the portion of soil through which infiltration occurred to as little as 2.4 % of the soil at 10 cm depth (site C). When the A1 horizon was at soil moisture contents near field capacity, water repellence was no longer apparent, however, the hydraulic conductivity was significantly lower than when the soil was dry and water repellent. Furthermore wetting front instability developed during infiltration into both soil at high and low antecedent soil moisture content. Reduced hydraulic conductivity and development of wetting front instability during infiltration into wettable soils at moisture contents near field capacity was attributed to difficulty displacing existing soil water further down the soil profile during infiltration of new water.

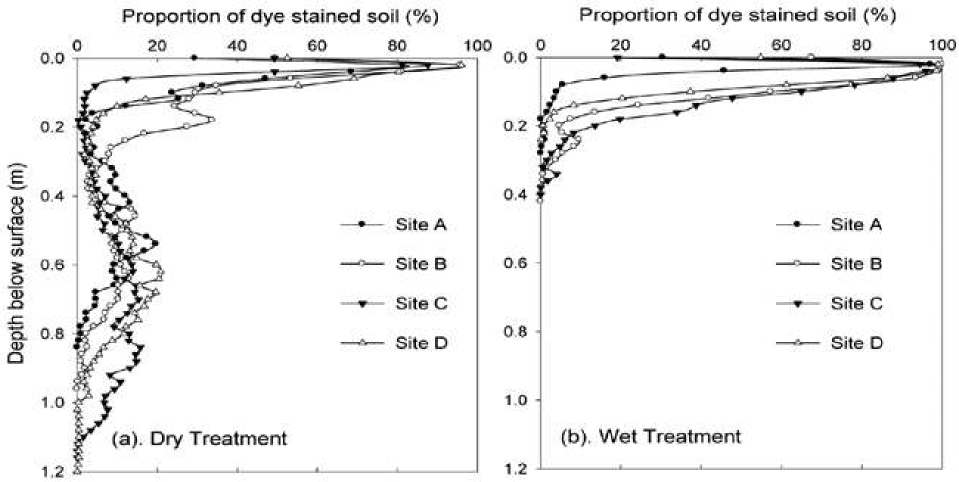


**Figure 1: Effect of antecedent soil moisture on the proportion of soil which participated in infiltration (a) dry treatment, (b) wet treatment. Analysis conducted at 2 cm intervals.**

*Infiltration into the A2e horizon*

Infiltration into the A2e horizon and sand infills resulted from a combination of funnel flow and sorptive flow. Despite similar particle size the near-saturated hydraulic conductivity of the A2e horizon was significantly lower (4.08 mm $hr^{-1}$) than the A1 horizon (48.4 mm $hr^{-1}$) (Table 3). Unlike the other soil horizons, the near-saturated hydraulic conductivity of the A2 horizon was significantly lower in the dry treatment than the wet treatment, which was attributed to precipitation of soluble amorphous silica in the menisci between sand grains (Norton 1994). In moist conditions the silica cementation dissolved resulting in higher micropore and mesopore flow rates.

*Infiltration into the B2 horizons*

Volumetric shrinkage experiments demonstrated that drying subsoil clods from near saturation increased inter-clod porosity (shrinkage cracks) by up to 0.35 $cm^3$ $cm^{-3}$, whilst increasing intra-clod density from 1.35 g $cm^{-3}$ to 1.95 g $cm^{-3}$. Infiltration through shrinkage cracks resulted in infiltration bypassing around 99 % of the soil matrix in the B21 horizon and 94 % of the soil matrix in the B22 horizon. Infiltration was influenced by differences in soil structure between the four sites. At site B, with strongly developed columnar structure, rivulet flow down the side of the shrinkage cracks and ped faces occurred at wetting front velocities between 2000 and 3000 mm $hr^{-1}$, up to three orders of magnitude higher than the measured saturated hydraulic conductivity. As rivulets reached the bottom of the shrinkage cracks, infiltration accumulated and spread laterally through the voids between the clay columns, a process described as 'filling from below'. At sites A, C and D, in which columnar development was weak or absent, the angular to subangular blocky structure resulted in less rivulet flow and greater accumulation of the dye tracer within horizontal and vertical cleavage plains at depths between 30 cm and 100 cm depth. At all sites, the presence of unstained shrinkage cracks demonstrated that only preferential pathways which were hydrologically connected to finger flow or ponding on the upper surface of the B columns, actually participated in flow.

At high antecedent soil moisture content, clay swelling and closure of shrinkage cracks in the B2 horizons decreased the number, size and contribution to flow from macropores. Clay swelling at high antecedent soil moisture content was expected to facilitate ponding on the surface of the B horizon and initiate subsurface lateral flow. However, the upper surface of the B21 horizon had limited dye staining whilst subsurface lateral flow was noted within the A1 horizon rather than the $A2_e$ horizon or the A/B horizon boundary as commonly reported in the literature. The Hele-Shaw experiments indicated the development of flow instability and development of lateral flow within the A1 horizon (rather than the A/B horizon boundary) resulted from difficulty displacing existing soil moisture further down the soil profile following swelling of the clay subsoil.

## Conclusion

Infiltration into texture-contrast soils proved considerably more complex than expected or had previously been reported in the literature. Dye tracer studies demonstrated that differences in soil morphology, chemistry and physical attributes had little effect on the proportion of soil which participated in flow, or the maximum depth of infiltration. Occurrence of preferential flow was largely controlled by antecedent soil moisture content which influenced water repellence and subsoil shrinkage crack development. In dry soil conditions, preferential flow resulted in deeper, faster infiltration than would otherwise be expected. Consequently the risk of agrochemical mobilisation to shallow groundwater via preferential flow is considerably greater than had previously been considered.

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# Factors related to the erosion of a constructed soil cover on mineral tailings

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A recently constructed soil and vegetation cover on mineral tailings was showing gully development and areas with poor vegetation establishment. Their primary causes were investigated while quantifying sheet and gully erosion to determine if erosion of the cover may be a future concern. The cover system consisted of a liner above the residue, a thin layer of sand that acted as a drainage layer, subsoil made up of silty clay pallid zone material, and topsoil that contained a large percentage of gravel. The vegetation consisted of shrubs and bushes with little ground cover. The rate of sheet erosion, determined over a 7-month period (spring–summer, experiencing 311 mm of the 1160 mm mean annual rainfall), was 2.2 mm with 95% confidence interval [1.0, 3.4] in vegetated areas and 2.9 mm [1.8, 4.0] in bare areas. Gully deepening was 2.6 mm [−1.5, 6.7] over 7 months with minor head-cut retreat and lateral bank collapse evident, indicating that gully development had slowed considerably after an initial period of major erosion. Gully erosion was triggered by old tracks or bare areas acting as a catchment, resulting in gully initiation down-slope. Neither vegetation (canopy) cover nor the variation in gravel content played a significant role in reducing erosion. The soils of bare areas were sodic and showed pedological evidence and vegetation indicators of waterlogging. When the CAESAR landscape evolution model was used to simulate erosion over short time scales with local weather data, it predicted a degree of both gully development and total erosion substantially less than that measured.

# Bridging soil physics and soil assessment with SINFERS

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The assessment of a soil is its characterization in terms of the functions it performs, the processes it undergoes, and the threats it faces, among other considerations. Knowledge of a soil at this level serves as the input to decision making at the policy and management level below it, and so it is a vital link in the soil management chain. Arguably, one of the most important inputs for such an assessment is a soil's set of physical properties. Without an accurate characterization of a soil's physical properties, it is impossible to assess a soil's relative utility, to map its functions and processes, or to fully understand the threats it faces. Therefore, it is important to have a class of tools whose sole purpose is to produce accurate, inexpensive information about a soil's physical properties. The Soil Inferencing System (SINFERS) is a rule-based expert system to predict unavailable or unfeasibly obtainable soil property values from known soil property data, and its existence supports the creation of such soil assessment tools. This presentation provides an introduction to the technical aspects of SINFERS, its relation to soil physics, and how the project is currently progressing.

# Land use performance for major land types and soils in central Queensland

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### Abstract

FAO's latest projections indicate that as the world's population grows to around 9 billion by 2050, global agricultural production must grow by 70 percent. To help meet this global demand for food and fibre, we will need to assess our existing soils used for agriculture as well as expand the area of arable land. In central Queensland, there was a need to quantify the production and economic performance of various land types and soils used for crop and forage pasture production. This comparison was needed to be able to better match land use to land capability. We modelled pasture growth, crop yield and gross margins for various locations, land uses and land types in central Queensland to generate a matrix of estimates for each of these combinations. We found that the land types with the highest soil water holding capacity and soil fertility such as Brigalow Softwood Scrub, Alluvial Brigalow and Coolibah Floodplain had consistently higher crop and pasture production and gross margins. These land types were all deep Black Vertosols with soil water holding capacities greater than 200 mm. Land types with the lowest soil water holding capacity and fertility such as Mountain Coolibah Woodland, Brigalow Blackbutt (Brown Sodosol) and Silver leafed Ironbark on duplex (Brown Sodosol) had the lowest production and gross margins. Because these marginal land types had a higher environmental cost for a lower potential profit, we recommend that they are therefore better suited to lower risk land uses such as perennial grass pastures.

### Key Words

Land capability, land type, land use, economics, modelling, soils.

### Introduction

Food and fibre production will need to increase by 70% globally to meet an increasing world population, which is estimated to have increased by 2.3 million by 2050 (FAO 2009). New land will be required to help meet this demand, as well as looking at the performance of our existing soils used for agriculture. This has the potential of putting agricultural production systems and the environment under ever increasing pressure.

Proper matching of land use to land capability has important production implications for sustainable and profitable crop and forage production in central Queensland. A comparison of land use performance on major land types in central Queensland was needed to be able to better match land use to land capability.

The purpose of this study was to quantify the production and economic performance of various land use options on major land types in central Queensland used for crop and forage pasture production. To achieve this end, land types in central Queensland were mapped using Geographical Information Systems (GIS). Major land types used for crop and forage production were then identified and evaluated for production and economic performance for different land use options for various sites in central Queensland. The crop and pasture model, APSIM, and economic models were used to provide an indication of the profitability of different land uses across different land types and locations. In this paper, we provide estimates of crop and pasture production and gross margins for a range of land uses, land types and locations in central Queensland.

### Method

A land type map of central Queensland was prepared using ArcGIS and an existing map of the Fitzroy Basin. This map was further extended to include land types that fell in the Belyando Suttor Catchment to come up with a single electronic land type map of central Queensland (Figure 1). We have used land types in this study rather than soil types to be consistent with other grazing land management work done in Queensland and also because it takes into account other factors that affect land suitability and limitations to production. A land type is defined as an area of land that has characteristic patterns of soil, vegetation and landform that are easily recognised by landholders in a region (Whish 2010). Out of 35 land types described in the region, only 11 were suitable for growing grain and forages, including some of the more marginal land types. These 11 land types were used for modelling and economic analysis (Figure 1).

Crop and pasture modelling was done with Agricultural Production Systems Simulator (APSIM), a modular modelling framework that was developed to simulate biophysical process in farming systems (Keating et al. 2003). The land uses modelled were opportunity cropping with sorghum, wheat and chickpea (based on soil water availability); annual summer cropping (sorghum); annual winter cropping (wheat); annual forage sorghum; annual forage oats; annual lablab and perennial buffel grass. A spreadsheet based economics model was developed to analyse APSIM outputs for each of the land uses over 51 years (1959 – 2009) for 11 land types/soils for 10 locations/climates.

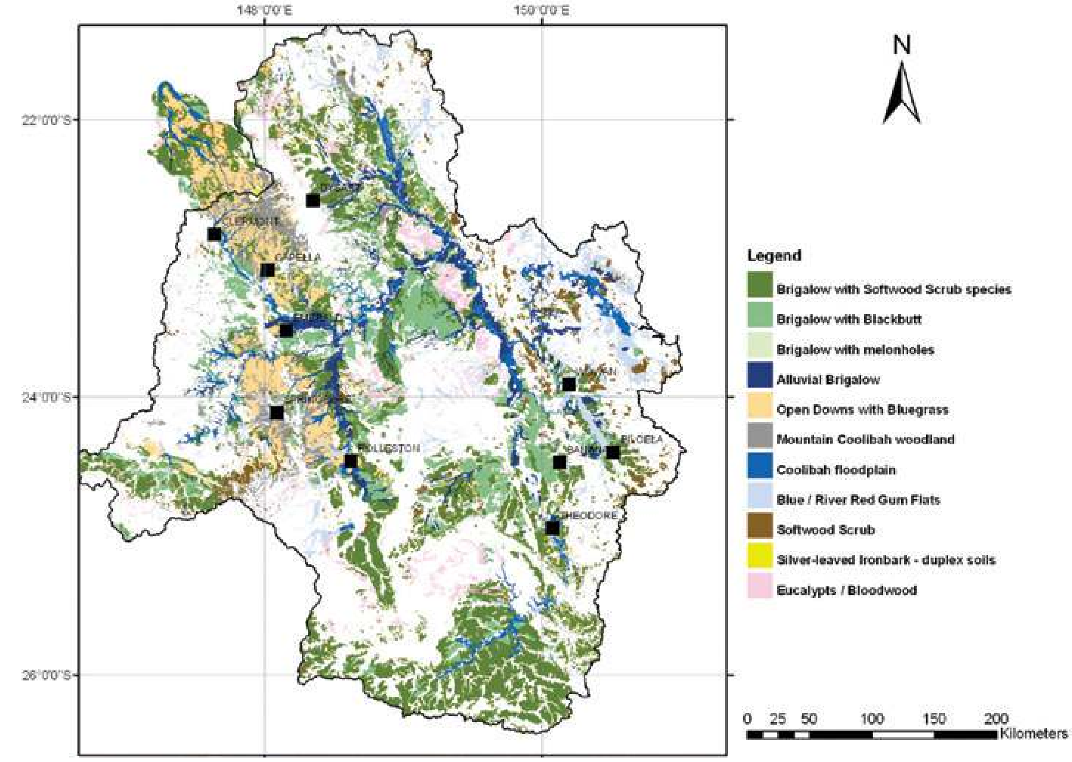


**Figure 1: Land types of central Queensland suitable for cropping and forage pastures**

The central Queensland region has a subtropical climate with summer-dominant rainfall. Rainfall is highly variable and a slight decrease in summer rainfall dominance occurs from north to south. Potential evaporation generally exceeds rainfall. In crop and forage growing areas of central Queensland, rainfall ranges from 500 to 700 mm/yr.

Cracking clay soils or Vertosols are widespread through the basin and are important economically as they have high soil water holding capacity and therefore are the main cropping soils. The predominant land types with cracking clay soils are soils developed on basalt (open downs, mountain coolibah, brigalow or gidgee scrub); soils developed on other parent materials (brigalow and/or softwood scrub); soils developed on river and creek flats (coolibah or blue gums). Open downs soils are the most extensive in central Queensland and their features include a very high water holding capacity and moderate soil fertility. Brigalow and/or Softwood Scrub Cracking Clay Soils occur throughout all areas of central Queensland and are the most extensive cropping soils. They have a high water holding capacity and were originally fertile soils with medium concentrations of nitrogen and medium to high phosphorous.

Soils developed on river and creek flats such as the coolibah or blue gum black cracking clay soils are found on the floodplains of the major river systems and their tributaries in the region. Usually these soils have high clay contents (50– 65%) and high water holding capacities. Texture contrast or duplex soils are widespread in central Queensland. These texture contrast soils generally have higher runoff (Owens et al. 2007), due to their hardsetting surface, low to moderate PAWC and subsoil permeability. Only a small proportion of these Sodosols (mostly black and brown Sodosols) are suitable for cropping. Dermosols include the dark and brown non-cracking clays found in association with the cracking clays in the brigalow and gidgee lands and also on recent alluvium along floodplains. Dermosols have variable levels of soluble salt and exchangeable sodium through the profile, and also have moderate to high PAWC, depending on their depth.

**Results and Discussion**

Crop and pasture production was always greatest for these land types for most land uses and locations in central Queensland – Brigalow Softwood Scrub, Alluvial Brigalow and Coolibah Floodplain (Figure 2). These land types have consistently higher production and economic performance with more opportunity crops because of the higher fertility and higher PAWC. However, some land types such as Alluvial Brigalow and Coolibah Floodplain have limitations such as occasional flooding in certain areas which need to be taken into consideration.

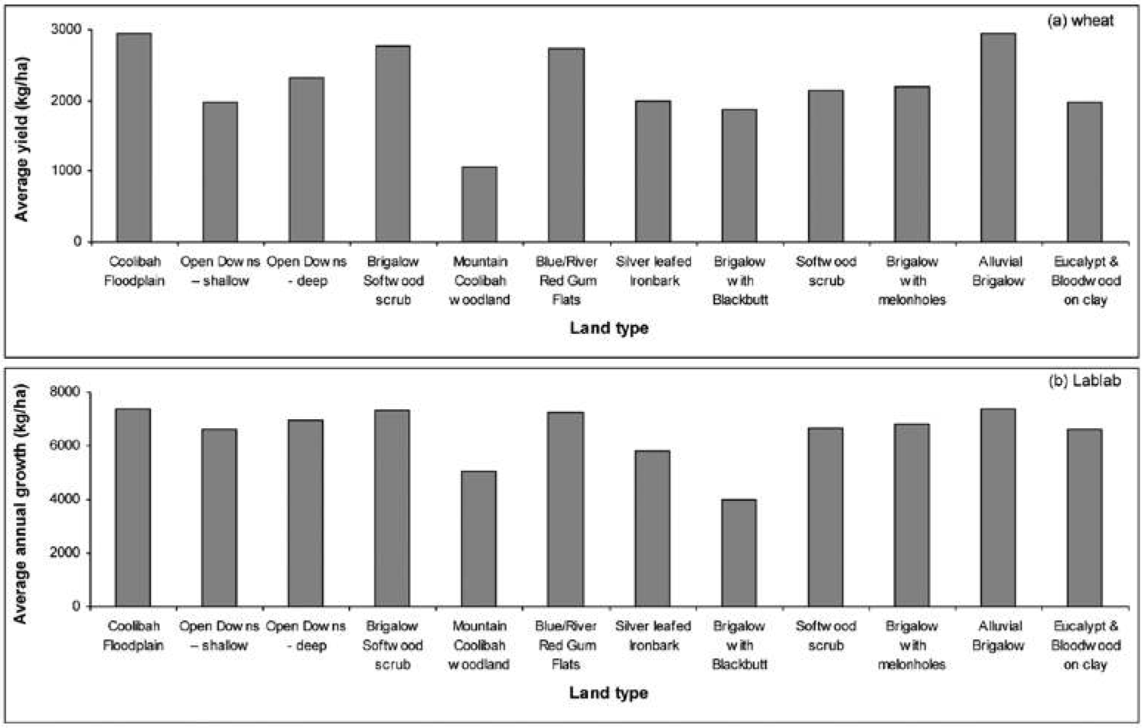


**Figure 2 Comparing crop and pasture production for all land types at Theodore for wheat and lablab.**

The lowest performing land types were Mountain Coolibah Woodland, Brigalow Blackbutt and Silver leafed Ironbark on duplex (Figure 2). These land types have the lowest soil water holding capacity and lower soil fertility. The Mountain Coolibah Woodland land type is a shallow soil with 60cm of rooting depth and low water holding capacity. Brigalow Blackbutt and Silver leafed Ironbark on duplex are Sodosols which have a hard setting soil surface with greater runoff and soil erosion. Marginal land types such as these are better suited to permanent pastures. Owens et al. 2007 modelled runoff and drainage for all soils in the Fitzroy Basin for salinity assessment and found higher rates of runoff and drainage in the soils found in these marginal land types; i.e. soils with shallow rooting depths, low PAWC and hard-setting surfaces. These marginal soils have a higher 'environmental cost' for a lower potential profit.

Opportunity cropping returns the highest gross margins/ha/yr for all land types except the marginal land types (Figure 3), followed by continuous sorghum. All scenarios showed larger variation in income from cropping and a more stable income from grazing permanent pastures over the study period 1959 to 2009. Sequences of wet years, for example in the early 1980's, provided larger returns from both cropping and grazing. Sequences of dry years, for example in the early 1990's, gave reduced returns from cropping, whereas cash flow from the buffel grazing enterprise was able to keep the business viable. Buffel grass pasture returns consistently low but positive gross margins due to the absence of planting costs and exhibits low levels of variation from year to year. This illustrates why mixed grain and cattle producers with a long term perspective keep some cattle on their properties on permanent pastures such as buffel grass.

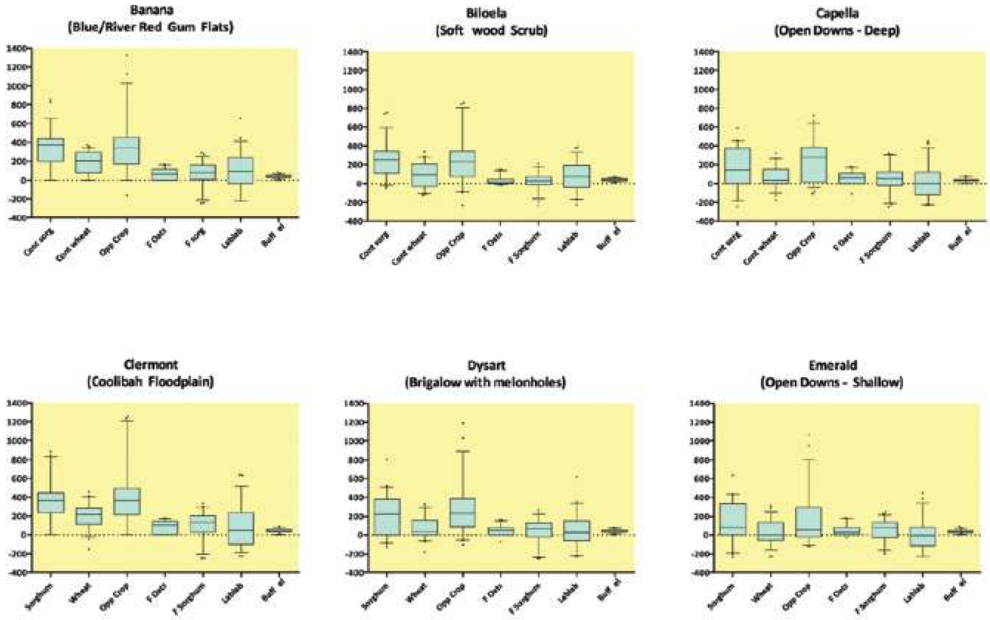


**Figure 3: Comparison of land use performance in terms of mean annual gross margins ($/ha/yr) for selected land types and locations. Grain cropping offers the potential for higher gross margins with an associated increase in risk, and grazing offers lower returns while reducing risk.**

## Conclusion

Land type maps and economic performance data presented here for central Queensland can be used by producers to better match land use performance to land capability. Production and economic performance of each of these land types can be compared for the different land uses and locations in central Queensland to help improve overall farm profitability. Production and gross margin figures from the more marginal land types can also be used as a basis for decisions to take out the poorer county from crop and high output forage production and into lower risk land uses such as perennial grass pasture which provides a consistent income and reduces risk of salinity or erosion. Comparison of the production and economic performance for these land types, land uses and locations could also be used to make decisions about procuring land. This study can help design better adapted farming systems with increased flexibility and more sustainable economic growth.

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# A wide span controlled traffic farming concept for efficient and environmental friendly vegetable production

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A new farming concept for vegetables is being developed where the growing beds are three to four times the width of what is possible with traditional farm mechanization. By using less than 10% of the land for wheel tracks, 20 to 30% more plants can be grown in fields with 9 m wide beds compared to traditional growing systems with wheel tracks for every 1.5 or 2.0 m.

A prototype of a Wide Span carrier has been developed. It will be tested on a commercial farm where all operations for producing onions will be performed with implements mounted within the span of the carrier. For many vegetable crops there are so far no harvest solutions for controlled traffic farmers. This will be solved as well, as a bunker harvester is being developed for the wide span carrier. Onions will be unloaded at field edges only thereby avoiding tractors and trailers in the field.

As part of my PhD study I will assess the environmental impacts of the new farming concept by a targeted Life Cycle Assessment (LCA) for growing of onions. Also the impact on the soil conditions is evaluated in a trial where non compacted plots on the farm are compared with plots that have been compacted. The trial includes measurement of draft requirements during tillage, analysis of the water retention, measurement of root development and measurement of yield and quality of the harvested crop.

# Variability of soil physical properties of dairy farms in south eastern Australia

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Soil provides physical support, and supplies water and nutrients for pasture growth. Dairy farm management practices associated with animal treading can have detrimental impacts on physical properties of soils. This paper aims to: (i) quantify variability of soil physical properties; and (ii) develop a sampling strategy for future monitoring of Victorian dairy soils. Analysis of historical data showed that macroporosity (APS) to be the most variable soil property (CV = 158%) in the three dairy regions of Victoria, followed by clay (CV = 60%), sand (CV = 43%) while bulk density (BD) being the least variable (CV = 9%). Analysis of data from the southwest region showed that significant ($P < 0.01$) differences between the 24 farms were observed for all soil properties. Differences in soil properties between soil types were also found to be significant ($P<0.01$) except for APS and field capacity (FC). The interaction between farms and soil types was observed to be non-significant ($P>0.1$) for all soil properties. Mean values of BD, and APS for the 24 farms varied from 0.76 – 1.4 $Mg/m^3$ and 5.5 – 21.5 vol% respectively. Variance component analysis showed, that for BD, APS and FC, variances between sampling points within a site were the highest, followed by between farms, between paddocks within a farm, between sites within a paddock and between soil types. Practical implications of these findings for developing a sampling strategy for monitoring dairy soils are discussed.

# Plantations in streamside management zones of grazed landscapes: Stream flow and water quality

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The use of streamside management zones (SMZs) as buffers to protect streams in the agricultural landscape is a priority internationally for investing in water quality protection, but the effects on stream flow are uncertain. The importance of water use by pasture versus plantations is not known for narrow bands of trees in the lowest and wettest part of a landscape. We are addressing this knowledge gap on pastured farmland in southern Tasmania. The paired-catchment experiment includes an adjacent headwater, intermittent stream. One catchment contains a high-growth-rate SMZ plantation (planted in 2008); the other remains as a reference catchment with pasture throughout the catchment. During the third year of growth, soil water content in the SMZ began to be noticeably lower under the plantation. The plantation effect on total annual stream flow was not detectable by three years of age, but the plantation appeared to have reduced stream flow on some occasions during periods of low flow. Measurements are continuing and will be up-dated for the conference. Establishment of the plantation improved water quality for turbidity, phosphate and bacteria. Research at another site showed that a 20-year-old SMZ plantation could be harvested in accordance with the forest practices code without causing sedimentation of stream water. In this case, drainage from a road and cattle access to the stream remained major water quality concerns. A visit to the paired-catchment experiment is included in one of the conference field trips.

# A review of soil policy in NSW

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A popular opinion amongst many soil professionals is that soil conservation and management appears to have been “lost”. However, investigation of policy in NSW as a case study, demonstrates that soil management appears to be both obvious and deliberate. Nonetheless, there have continued to be losses to soil function, carbon decline is evident and issues such as soil structure, acidity, salinity and sheet erosion remain a significant concern in much of the state. This raises uncertainty as to the effectiveness of current approaches.

Our challenge is to reflect on the appropriateness of policy, systems and processes which, assume that impacts can be mitigated, disturbed ecosystems can be restored, reconstructed or repaired, and that this can be done within timeframes of decades or less.

# Effect of clay delving on water infiltration in texture contrast soils

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Clay delving is a growing practice on texture-contrast soils to increase agricultural productivity. Delving brings up sub-soil clay into the surface sand and (unlike the practice of clay-spreading to overcome water repellence) it disrupts the boundary between the surface sand and subsoil clay. Such disruption is expected to significantly alter the hydraulic properties of the soil profile and potentially increase root distribution and water use efficiency. To evaluate the effects of clay delving on soil hydraulic properties, we impregnated some delved and undelved texture-contrast soils with a blue dye at two sites in the southeast of South Australia under dry conditions. After applying the blue dye to the soil surface using a rainfall simulator, we exposed the soil profiles and took digital images of the stained soil to quantitatively evaluate the extent and distribution of water.

Significantly more water infiltrated the top 10 cm of the delved soil and its distribution was significantly more uniform than in the undelved soil. As expected, preferential and patchy flow characterized water movement in the undelved soil such that significant parts of the profile below 5cm remained completely dry, including the subsoil clay horizon. However, results varied considerably between sites and the effectiveness of delving on infiltration may depend on the initial soil characteristics and the nature of different delving practices at each site. Furthermore, the extent to which delving enhances water distribution once the soil has become wet will be evaluated in the next stage of this project.

# Herbicide wash off from sugar cane trash

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Rainfall runoff of herbicides routinely used in sugar cane production has the potential to cause harm to rivers, lagoons, and the Great Barrier Reef in Northern Australia. The fate of these herbicides can be modeled within the landscape to assist in identifying efficient strategies to reduce the herbicide runoff and developing better land management practices. Cane trash is retained on the soil surface in most of the Australian sugar industry. Herbicides are generally sprayed on trash where they may contribute to herbicide runoff. However, few data are available on the mobility and concentrations of herbicides leaving surface trash cover during rainfall events. The purpose of this laboratory study was to quantify the amount of herbicides washing off sugar cane trash during simulated rainfall, to provide insight into likely herbicide behavior in the field.

Simulated rainfall was used to apply 100 mm of rain at a constant intensity of 50 mm/h to 1$m^2$ plots covered in cane trash. As an initial benchmark study, trash was sprayed with a conservative tracer, potassium bromide (KBr). KBr concentrations in wash off (rain falling through the trash) collected through time were inferred by changes in the electrical conductivity of the runoff samples, and selected samples were then analyzed for bromide concentrations. The effect of ‘time after spraying’ on concentration in wash off was also investigated through a series of experiments where plots were sprayed and left for varying time durations before being rained on.

The preliminary results using KBr show concentrations of bromide were initially high, declining exponentially as a function of time and applied rain. Herbicide tests are expected to show similar results but with changing coefficients due to different levels of adherence and different decay rates. Wash off parameters used in herbicide runoff models will be fitted to the data and compared.

This study will provide insight into the exact nature of the wash off from cane trash and provide wash off parameters for herbicide modeling. This would provide information on the safe application of herbicides and efficient strategies that can be employed to reduce the herbicides wash off to the water bodies.

# Error and uncertainty with the determination of the soil water retention function and saturated hydraulic conductivity for soil water modelling

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## Abstract

Simulation of soil water movement by soil-water models based on Richards' equation requires knowledge of the soil water retention function and saturated hydraulic conductivity. Traditionally these parameters have been determined by desorption which is costly and time consuming. In this study we compared VGM parameter values determined by desorption with two rapid approaches, laboratory evaporative flux and inverse solution of *in situ* cumulative infiltration from tension infiltrometers. Methodology had a profound effect on all VGM parameters except *n*. Methodology influenced saturated hydraulic conductivity values which varied by up to four orders of magnitude. Both rapid assessment procedures were improved by knowledge of the residual water content θr or moisture content at -1500 kPa. In water repellent or vertic soils differences in VGM values resulted in a lack of confidence in all procedures.

**Key Words**

HYDRUS, disk permeameter, soil water retention function, soil-water model, inverse solution

## Introduction

Increased use of soil water models for simulating crop growth, irrigation performance and fate of agrochemicals has resulted in increased demand for rapid means of determining the van Genuchten (1980) - Mualem (1976) parameters (VGM) that describe the soil water and conductivity retention functions. Traditionally the soil water retention function has been determined by desorption using suction tables, tension plates, and pressure chambers. However the desorption approach is costly and time consuming requiring in the order of weeks to months for equilibration of intact cores at each equilibration step. In recent years a number of approaches have been developed to enable rapid determination of the soil water retention function and VGM parameters from either evaporative flux experiments (Wendroth and Wypler 2008) or inverse simulation of *in situ* infiltration from tension infiltrometers (Simunek and Van Genuchten 2000; Simunek *et al.* 1998). While these rapid assessment procedures have been tested on hydrologically simple soils, their use on complex vertic and water repellent soils has not been conducted.

## Methods

Analysis was conducted on a texture contrast soil (Mesotropic Mottled Subnatric, Brown, Sodosol) at the University of Tasmania Farm, Australia. The soil profile consisted of a seasonally water repellent, sandy loam A1 horizon, which overlaid, a bleached, loamy sand, $A2_e$ horizon, with sharp boundary to a neutral to acid ($pH_{CaCl2}$ 4.1–6.1), dispersive (ESP 5.3–10.8), vertic (linear shrinkage 9.86 %), mottled, ligh- medium clay subsoil, with extremely coarse columnar structure.

The soil water retention function θ(ψ) was determined by (a) desorption, (b) desorption with a correction for clay shrinkage, (c) evaporative flux, and (d) inverse parameterisation of *in situ* cumulative infiltration from tension infiltrometers. Desorption was conducted using suction tables and pressure chambers according to Cresswell (2002) using 100 x 75 mm cores and disturbed samples. As subsoil density was influenced by moisture content, the retention function was corrected to account for the reduction in soil volume during desorption using the soil shrinkage characteristic curve (SSCC) which was determined by the 'balloon' approach (Cornelis *et al.* 2006). The evaporative flux procedure determined the θ(ψ) relationships from the matric potential gradient between two tensiometers and the change in soil mass during evaporation from a six cm high intact soil core (Wendroth and Wypler 2008). van Genuchten parameter values were determined using the curve fitting software RETC with the m=1-1/n assumption (van Genuchten *et al.* 1991). Inverse solution of cumulative tension infiltration data was conducted in HYDRUS-2D using the inverse solution developed by Simunek *et al*. (1998). The objective function was defined from the average cumulative infiltration from 4-6 tension infiltrometers operated at six sequential supply potentials in soils at both high and low antecedent soil moisture content. Saturated hydraulic conductivity ($K_{sat}$) was determined by (a) constant head ($\psi$ = +10 mm) approach on saturated cores, (b) matric potential gradient during evaporation from soil core (Wendroth and Wypler 2008), and (c) *in situ* infiltration from tension infiltrometers using the Reynolds and Elrick (1991) approach for solving hydraulic conductivity. For both the tension infiltration and evaporative flux approaches the saturated hydraulic conductivity was determined by extrapolation of the $K(\psi)$ relationship to zero using RETC (van Genuchten *et al.* 1991).

## Results

The soil water retention functions are presented for the A1 and B22 horizons (Figure 1). Methodology had considerable effect on saturated water content ($\theta$s), the residual water content ($\theta$r), the soil structure parameter ($\alpha$) and to a lesser extent the shape parameter (n). Accounting for soil shrinkage during desorption (desorption-shrink) had little effect on the soil water retention function near saturation, however, at -1500 kPa the moisture content was 0.033 $m^3$ $m^{-3}$ higher in the B21 horizon, and 0.021 $m^3$ $m^{-3}$ higher in the B22 horizon.

The desorption and evaporative flux approaches produced lower $\theta$s values than the two *in situ* infiltration approaches (disk – dry / wet). The *in situ* infiltration approaches (disk – wet / dry) also had higher values for the soil structure parameter $\alpha$ (demonstrated by the steeper reduction in the retention function between 0.1 and 10 kPa) than the laboratory procedures. This was particularly evident for the *in situ* infiltration at low antecedent soil moisture content in which infiltration was dominated by macropore process. The higher $\theta$s and $\alpha$ values for the *in situ* approaches (especially the dry treatment) is thought to have resulted from (i) the *in situ* infiltration approaches providing better estimates of macropore function at matric potentials close to saturation (0 to -10 kPa), (ii) reduced likelihood of sampling macropores with the smaller laboratory cores, (iii) swelling of vertic clay subsoils following prolonged saturation for laboratory analysis, and / or (iv) biological gumming of pores due to prolonged saturation of core samples. The residual water content $\theta_r$ was higher for the evaporative approach than the desorption approach or measured moisture content at -1500 kPa. Poor $\theta$r estimation by the evaporative flux approach was attributed to difficulty extrapolating the $\theta(\psi)$ relationship beyond the maximum range of the tensiometers (-60 to -80 kPa). The inverse solution of cumulative infiltration approach experienced considerable difficulty solving to local minima when the $\theta_r$ parameter was included in the object function. However by removing $\theta_r$ from the objective function, and using either the $\theta_r$ determined by desorption or moisture content at -1500 kPa, the inverse solution was able to be solved and the occurrence of non-uniqueness was adequately reduced. Difficulty obtaining local minima for $\theta_r$ was attributed to the tension infiltrometers being highly sensitive to flow in macropores, whilst being rather insensitive to flow in micropores (Simunek and van Genuchten 1997).

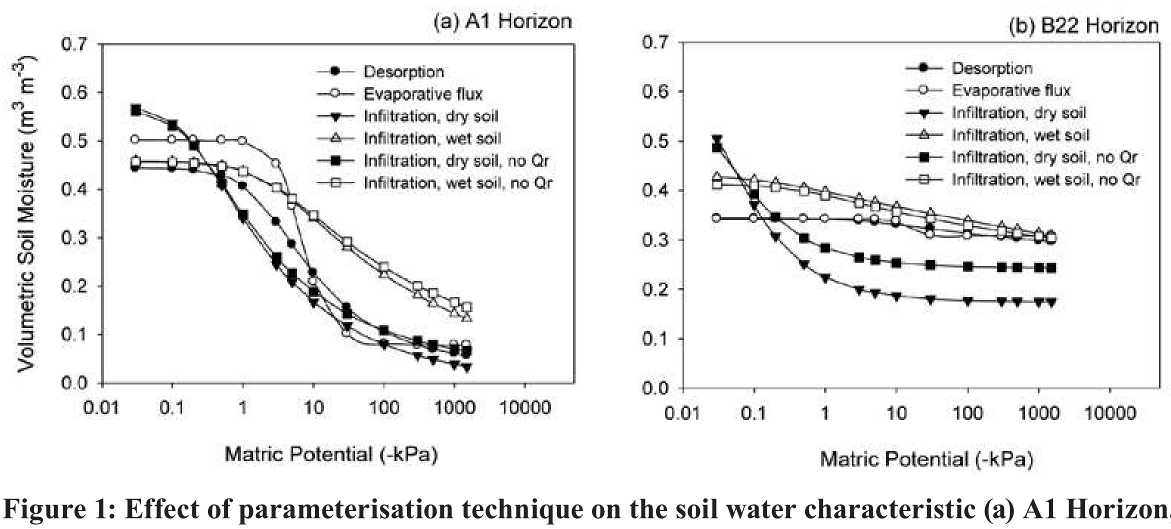


**Figure 1: Effect of parameterisation technique on the soil water characteristic (a) A1 Horizon, and (b) B22 Horizon. No Qr refers to removing the $\theta$r parameter from the objective function.**

Parameterisation approach and antecedent soil moisture significantly ($p < 0.05$) influenced values of $K_{sat}$ by up to four orders of magnitude (Figure 2). Measured $K_{sat}$ from the constant head approach were generally lower than the other approaches, which was attributed to prolonged saturation causing clays to swell and possibly biological gumming of soil pores. Error with the evaporative flux approach was attributed to a lack of data near saturation which resulted from inability to calculate unsaturated hydraulic conductivity at low matric potential gradients during the early stages of evaporation. Large variance in error with the tension infiltration approach resulted from the exponential nature of the K(ψ) relationship near zero, in which small changes in pressure head resulted in large changes in the hydraulic conductivity as matric potential approached zero.

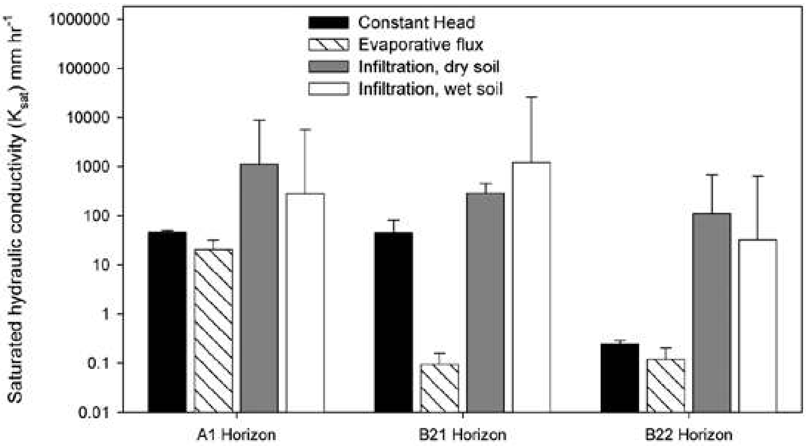


**Figure 2: Effect of parameterisation approach on saturated hydraulic conductivity. Error bars represent + 1 standard error.**

## Conclusion

Determination of VGM parameters by desorption and evaporative flux in water repellent or vertic soils was influenced by the requirement to saturate the soil prior to assessment. For soils in which hydraulic properties vary with antecedent soil water content (water repellent, vertic clays) determination of VGM parameters is required at representative field moisture contents, which in most instances excludes the use of desorption and evaporative flux approaches. Inverse solution of cumulative infiltration from tension infiltrometers was able to discern the effects of antecedent soil moisture on soil structure and soil hydraulic properties. However the approach had difficulty obtaining unique local minima when $\theta_r$ was included in the objective function. Consequently the procedure required $\theta_r$ to be determined from additional pressure plate analysis of disturbed samples at highly negative potentials. This study found that the rapid approaches were best applied at specific pore size ranges, the inverse solution of cumulative infiltration approach operated best between 0 and -10 kPa, while the evaporative flux approach operated best between -5 and -80 kPa, consequently the two rapid approaches were greatly improved by prior knowledge of θr or the moisture content at -1500 kPa.

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# Practical estimation of the water demand for landscape plants

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Increasing urbanization and growing population places greater demands on dwindling water supplies, particularly in Australia which is known as the driest inhabited continent on earth. To help cope with increased urban water demands and a diminishing availability of a secure fresh water supply, reclaimed wastewater is an attractive alternative water resource that also provides a suitable nutrient supply. Sustainable irrigation and fertigation management of reclaimed wastewater results in water savings and environmental protection. Irrigation management necessitates better understanding of various plants' water requirements in order to decrease environmental risks and increase water use efficiency. The difficulty in measuring evapotranspiration from urban landscape plants complicates the development of suitable irrigation management strategies for urban landscapes, particularly when water scarcity is considered and wastewater is applied as the alternative water resource. Complex challenges in improving water use efficiency, irrigation best practice, and management of nutrient supplementation through applied wastewater signify the importance of a robust research program aimed at developing sustainable irrigation and nutrient management practices for landscape plants.

The National Water Commission (2010) and REM reports (2008 & 2007) all declared that the limited understanding of water requirements of natural ecosystems is a key obstacle for regional water management. Although the water demands of agricultural crops are well established in field and laboratory studies, in urban landscapes, most attention has been paid to turf grass rather than the landscape species. This short paper will employ an estimation method of water demand of urban vegetation namely Water Use Classification of Landscape Plants. Veale Garden within the Adelaide Park Lands was used as the study area. This was selected because of the significance of the Adelaide Park Lands as a valuable social, environmental and recreational urban landscape around the city of Adelaide and also because it is potentially an important consumer of reclaimed wastewater, through the new GAP water recycling scheme. The outcome of this research contributes to the practical estimation of the water demand of landscape plants.

# Mapping estimated deep drainage in the lower Namoi Valley using a chloride mass balance model and EM34 data

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The Murray Darling Basin (MDB) accounts for half of all water used for irrigation in Australia. However, improvements in water use efficiency (WUE) are required owing to increasing demands on water (e.g. environmental flows). This requires data on the spatial distribution of soil-hydrological properties, such as deep drainage (DD). Measuring DD using lysimeters, whilst accurate, is site specific. Alternatively, estimates are commonly made using chloride mass balance (CMB) models. Gaining this information across a large area is still problematic due to the prohibitive cost of drilling, sampling and laboratory analysis. Ancillary data, obtained from electromagnetic (EM) instruments, have been used to add value to a limited number of DD estimates. Herein we evaluate the use of a Hierarchical Spatial Regression (HSR) technique to map the estimated DD using a steady state CMB model coupled to EM34 measurements. We compare a standard least squares and a stepwise multiple linear regression (MLR) model. The former includes the use of EM34 signal data in the horizontal (EM34-10H, EM34-20H and EM34-40H) and vertical (EM34-10V, EM34-20V and EM34-40V) dipoles as well as two trend surface variables (scaled-easting and -northing). The latter model only includes a statistically significant ancillary variable (EM34-10H) and a trend surface parameter (scaled Northing) and we use this to estimate DD across the lower Namoi Valley. We conclude that EM34 data available on a 1 km grid is useful for mapping DD on a reconnaissance level as the results are closely related to the physiography. In particular, large DD estimates are associated with the prior stream channels. Conversely, smaller DD estimates characterise the agriculturally significant clay plain which are used extensively for irrigated cotton production. The map of estimated DD will allow improved siting of dams and irrigation fields as well as indicate where more efficient cropping or irrigation systems can be implemented to increase WUE.

# Effect of drip irrigation and plastic cover on soil salt movement in cotton field in arid region

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Water is the limiting factor of agriculture in arid region. To save water, drip irrigation covered by plastic film has been widely applied in cotton field in China. While the effect on water-saving is significant, such practice also increased soil salinity in the soil, groundwater, and downstream rivers. Field experiment is being carried out to identify such changes. Results show that the salt quantity has an increase in different layers of the soil profile. Taking anion as the example, the overwhelming majority are Cl- and SO42-. The Cl- content in vertical direction from 0 cm to 150 cm increases gradually, while the SO42- content appears a peak value at 90 cm depth, which means SO42- transfer towards 90 cm soil layer. The accumulation of Cl- and SO42- is apparent at soil depth of 0-60 cm. Salt accumulation negatively relates to irrigation water use. Though EC of the leaching water from drip irrigation is higher than traditional irrigation, only small amount of salt is leached out, which led to an increase of soil salinity.

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