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### **Valuing the soil's natural capital**

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# Depth distribution of soil organic carbon as a signature of soil quality

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## Abstract

Soil organic matter is a key component of soil quality that sustains many important soil functions by providing the energy, substrates, and biological diversity to support biological activity, which affects aggregation (important for habitat space, oxygen supply, and preventing soil erosion), infiltration (important for leaching, runoff, and crop water uptake), and decomposition (important for nutrient cycling). Lack of residue cover and exposure of soil to high-intensity rainfall results in poor aggregation, reduced plant water availability, erosion, and off-site impacts of sedimentation and loss of soil nutrients to receiving water bodies. From a soil survey dataset in Georgia USA, profile distribution of soil organic carbon (SOC) was closely matched with an exponential function (i.e., highest at the soil surface and exponentially declining with depth). It is suggested that if sufficient ecosystem service data associated with profile distribution of SOC could be collected, a strong relationship would develop between SOC stratification ratio and various ecosystem services.

## Introduction

Soil, water, and air resources are fundamental components of agricultural systems. Achieving a balance between agricultural production and conservation of natural resources is a necessary step to achieve sustainability. Soil quality can be viewed as an indicator of sustainability, since soil quality is indirectly linked to food production, food security, and environmental quality (e.g., water quality, global warming, and energy use in food production) through its influence on key soil functions. Soil quality is a complex subject, encompassing the many valuable services humans derive from soil, as well as the many ways soils impact terrestrial ecosystems (Doran and Parkin 1994).

Achieving high soil quality requires that soil be able to perform several key ecosystem functions to an optimum capacity within the constraints of inherent soil characteristics and climatic conditions. Some key soil functions of interest in agriculture are:

- supplying and cycling nutrients for optimum plant growth;
- receiving rainfall and storing water for root utilization;
- filtering water passing through soil to protect groundwater quality;
- storing SOC for nutrient accumulation and mitigating greenhouse gas emission;
- decomposing organic matter and xenobiotics to avoid detrimental exposures to plants and the environment.

Soil organic matter – as a source of energy, substrate, and biological diversity – is one of the key attributes of soil quality that is vital to many of these soil functions. Stratification of SOC with depth is common in many natural ecosystems, managed grasslands and forests, and conservation-tilled cropland (Franzluebbers *et al.* 2000; Blanco-Canqui *et al.* 2006; Jinbo *et al.* 2007). The soil surface is the vital interface that receives much of the fertilizer and pesticides applied to cropland and pastures, receives the intense impact of rainfall that can lead to surface sealing following disruption of surface aggregates, and partitions the flux of gases into and out of soil. Franzluebbers (2002a) described a soil quality evaluation protocol that related the degree of soil organic matter stratification to soil quality or soil ecosystem functioning through its conceptual relationship to erosion control, water infiltration, and conservation of nutrients.

Stratification of SOC occurs with time when soils remain undisturbed from tillage (e.g., with conservation tillage and pastures) and sufficient organic materials are supplied to the soil surface (e.g., with cover crops, sod rotations, and diversified cropping systems). Stratification of SOC has been calculated with different depth increments, resulting in somewhat different conclusions of studies. For example, no-tillage (NT) cropland had higher stratification ratio of SOC (1.3, 0-10 cm / 10-20 cm) than under conventional-tillage (CT) cropland (1.0) on an Entic Haplustoll in Argentina (Quiroga *et al.* 2009), but values were lower than in similar evaluations using smaller depth increments on a Typic Kanhapludult in Georgia (3.8 under NT and

1.1 under CT, 0-6 cm / 12-20 cm) (Franzluebbers and Stuedemann 2008). On a Xerofluvent in Spain, stratification ratio of SOC was greater under conservation tillage than under traditional tillage, but values were higher when calculated as 0-10 cm / 25-40 cm than as 0-10 cm / 10-25 cm (Moreno *et al.* 2006). On reclaimed minesoils in Ohio, stratification ratio of SOC (0-15 cm / 15-30 cm depth) increased with time under pasture and forest management (Akala and Lal 2001). During pasture development in Georgia, stratification ratio of SOC (0-15 cm / 15-30 cm depth) increased from 2.4 at initiation to  $3.0 \pm 0.7$  at the end of 5 years to  $3.6 \pm 0.6$  at the end of 12 years (Franzluebbers and Stuedemann 2005; 2009).

The objective of this evaluation was to identify the impact of different sampling depths on the calculated value of stratification ratio among different land uses from historical soil cores collected throughout Georgia. Relationships of stratification ratio to water runoff, soil erosion, nutrient loss, and SOC sequestration are implied from a review of literature. Such relationships need to be quantified in the future.

### Materials and methods

SOC data were evaluated from 267 soil-survey profiles collected from 1954 to 1986 throughout Georgia (Perkins, 1987). SOC was determined by wet oxidation (Peech *et al.* 1947). Concentration of SOC was regressed upon depth of sampling (mid-point of sampling interval, which averaged a 10-cm interval) using the following equation:

$$\text{SOC} = a + b \cdot \exp(-c \cdot D)$$

where, SOC is soil organic C (g /kg), a is the minimum concentration of SOC deep in the profile (g/kg), b is the peak SOC concentration at the surface (g /kg), c is a decay coefficient controlling the magnitude of decline in SOC concentration with depth ( $\text{cm}^{-1}$ ), and D is depth (cm). Number of sampling intervals was  $6 \pm 1$  per profile. The mid-point of the upper-most sampling interval was  $9 \pm 3$  cm ( $14.0 \pm 8.9$  g SOC /kg) and the lowest mid-point was  $140 \pm 37$  cm ( $1.2 \pm 1.0$  g SOC /kg). Mean soil-profile distributions were developed from predictions with a unique equation for each soil profile at 0.1, 5, 10, 20, 30, 40, 50, 60, 70, 80, 90, 100, 125, 150, 175, and 200 cm depths. Stock of SOC was determined from concentration and bulk density of 0-5, 5-10, 10-20, 20-30, 30-40, 40-50, 50-60, 60-70, 70-80, 80-90, 90-100, 100-125, 125-150, 150-175, and 175-200 cm intervals. Bulk density was assumed to be negatively related to SOC concentration using the equation (Franzluebbers 2010):

$$\text{BD} = 1.71 \cdot \exp(-0.013 \cdot \text{SOC})$$

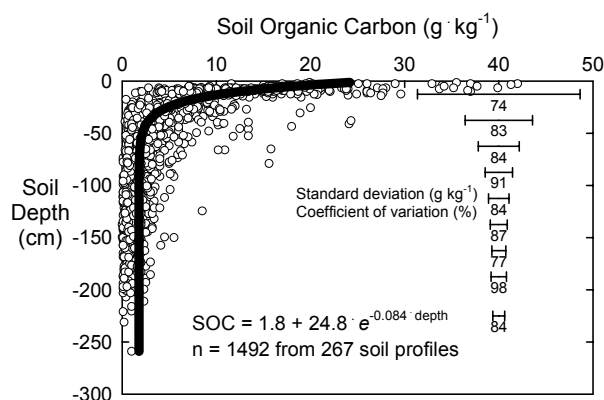
Where, BD is bulk density ( $\text{Mg m}^{-3}$ ) and SOC is soil organic C concentration (g /kg). Stock of SOC was also calculated summed to various cumulative depths from the surface.

Mean soil-profile distributions of SOC and stocks of SOC were compared among soil orders, major land resource areas, and land uses. Significant differences were declared at  $p \leq 0.05$ .

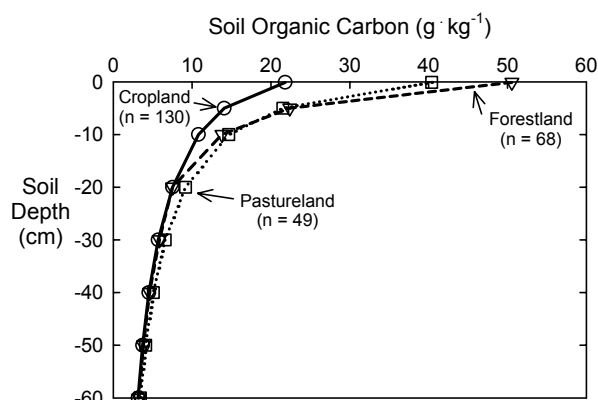
### Results and discussion

From the 1492 samples collected from 267 soil profiles, SOC concentration was highly stratified with depth (Figure 1). A large amount of variation occurred among all sampling depths, as evidenced by the coefficient of variation ranging from 74 to 98%. Soil profiles were, therefore, sorted into categories of soil orders (205 Ultisols, 35 Alfisols, 11 Entisols, 8 Inceptisols, 6 Spodosols, and 2 Mollisols), major land resource area within the large group of Ultisols (102 Coastal Plain, 47 Piedmont, 25 Ridge and Valley, 17 Blue Ridge, and 14 Flatwoods), and land use within soil orders and major land resource areas (130 cropped, 68 forested, 49 pasture, and 20 miscellaneous use). Depth distribution of SOC was significantly affected by land use category (Figure 2). At 5- and 10-cm depths, SOC concentration was greater under pastureland and forestland than under cropland. At 20-, 30-, and 40-cm depths, SOC concentration was greater under pastureland than under forestland and cropland. At 50- to 100-cm depths, SOC concentration was greater under pastureland than under cropland; forestland was not different from either of the extremes.

Stratification ratio of SOC was similarly different between the less-disturbed land uses of forestland and pastureland compared with the more-disturbed land use of cropland. Stratification ratio of SOC was 4.9 under forestland, 4.7 under pastureland, and 3.2 under cropland when calculated as 0-10 / 20-30 cm ( $\text{LSD}_{p=0.05}$  of 1.7), was 3.6 under forestland, 3.5 under pastureland, and 2.7 under cropland when calculated as 0-20 / 20-40 cm ( $\text{LSD}_{p=0.05}$  of 0.9), and was 3.8 under forestland, 3.6 under pastureland, and 3.1 under cropland when calculated as 0-30 / 30-60 cm ( $\text{LSD}_{p=0.05}$  of 0.8). Calculation of stratification ratio of SOC was more discerning among land uses when the numerator was limited to the surface 10 to 20 cm only.



**Figure 1. Soil organic C depth distribution from all 267 soil profiles in Georgia.**



**Figure 2. Soil organic C depth distribution when averaged across data within a land use category.**

This survey approach resulted in unequal distribution of observations among soil orders, major land resource areas, and land use systems. Due to the low number of observations in Alfisols, Entisols, Inceptisols, Mollisols, and Spodosols, no differences in SOC stock at various depths and in stratification ratio of SOC were detected (data not shown). Only in Ultisols with sufficient observations were there differences in SOC stock and in stratification ratio of SOC (Table 1). Pasture land use contained significantly greater SOC stock than cropland in Blue Ridge, Piedmont, and Coastal Plain MLRAs, but not in Ridge / Valley and Flatwoods MLRAs (Table 1). Across all soil orders and MLRAs, cropland contained lower SOC than other land uses at 0-10, 0-30, and 0-100 cm depths. Stratification ratio of SOC was greater under forestland and pastureland than under cropland. Unfortunately, detailed management information from this soil survey approach was not reported. There are a diversity of crop and pasture management strategies (e.g. crop rotation sequence, cover cropping, manure application, tillage type, stocking rate, fertilization regime, etc.) that could influence SOC sequestration, but such differences could not be separated in this analysis.

**Table 1. Stock of soil organic carbon (SOC) at various depths and stratification ratio of SOC among major land resource areas (MLRA; among Ultisols) and general land use category. All is for all soil orders and MLRAs.**

MLRA	Land use	No. obs.	Stock of SOC (Mg/ha)			Stratification ratio		
			0-10 cm	0-30 cm	0-100 cm	0-10/20-30 cm	0-20/20-40 cm	0-30/30-60 cm
Blue Ridge	Crop	7	21.5	46.7	80.7	2.4	2.2	2.5
	Pasture	2	34.8	76.7	120.4	2.9	2.7	3.2
	Alternate	3	37.3	72.1	108.6	5	3.7	3.8
	Forest	5	26.4	48.9	76.7	3.8	3.2	3.7
<i>LSD (p=0.05)</i>			13.4*	24.7*	33.7*	2.9	1.6	1.7
Ridge and Valley	Crop	13	23.6	49	76.7	3.2	2.8	3.3
	Pasture	4	29	48.2	69.2	8.8	6	5.8
	Alternate	4	26.5	59.2	99.1	2.3	2.2	2.7
	Forest	4	27.1	52.9	79.6	3.7	3.2	3.6
<i>LSD (p=0.05)</i>			13.1	24.9	29.7*	4.3*	2.7*	2.6*
Piedmont	Crop	27	19.5	40	64.6	3	2.6	3.1
	Pasture	8	29.6	53.8	78.6	5.3	4.1	4.5
	Forest	12	28	55.9	86.4	3.8	3.2	3.5
	<i>LSD (p=0.05)</i>			5.1*	9.6*	13.9*	1.9*	1.1*
Coastal Plain	Crop	67	16.8	34.7	57.1	3.1	2.7	3.1
	Pasture	13	25	47.2	70.1	6.9	4.5	4.2
	Alternate	10	20.9	46.2	78.1	2.5	2.3	2.7
	Forest	12	25.6	48.9	83.6	7.4	4.6	4.3
<i>LSD (p=0.05)</i>			7.0*	13.6*	23.8*	3.4*	1.6*	1.3*
Flatwoods	Crop	2	15.1	25.8	33.4	4.9	4.4	5.9
	Pasture	2	9.2	20.9	34	1.9	1.9	2.2
	Forest	10	17.7	28.5	42.6	6.3	4.7	4.8
	<i>LSD (p=0.05)</i>			10.4	14.9	18.3	6.5	4.3
All	Crop	130	19.1	39.9	66.1	3.2	2.7	3.1
	Pasture	49	25.9	52.5	86.2	4.7	3.5	3.6
	Alternate	20	25.2	53.2	85.5	3.1	2.7	3.2
	Forest	68	25	48.3	77.7	4.9	3.6	3.8
<i>LSD (p=0.05)</i>			3.7*	7.4*	12.9*	1.7*	0.9*	0.8

Stratification of SOC with depth has been shown to (a) positively impact soil structural integrity and water infiltration (Franzluebbers 2002b), (b) reduce soil loss and nutrient runoff (Franzluebbers 2008), (c) enhance soil biological activity (Franzluebbers 2009), and mitigate greenhouse gas emissions (Franzluebbers 2010). Much more research is needed to bolster these relationships so that this broad measure of soil quality can help promote greater resource efficiency and sustainability in the future.

## Conclusions

Soil organic matter under conservation management (pastureland and forestland) is typically more stratified with depth than under conventional cropping. This stratification should be viewed as an improvement in soil quality, because several key soil functions are enhanced, including soil structure, water infiltration, soil conservation, cycling of nutrients, and sequestration of C from the atmosphere. This analysis of deep-soil profiles throughout Georgia indicates that stratification ratio of SOC could be best calculated as 0-10/20-30 cm or 0-20/20-40 cm, because management induced changes in SOC are generally restricted to the surface 10 to 20 cm in soils of this warm and humid region.

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# Dynamics of soil microbial community structure and activity during the cropping period of cotton

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## Abstract

Healthy soil is a prerequisite for maintaining agricultural productivity and microorganisms are integral part of the soil ecology. We attempted to study the temporal changes in the soil microbial activity as well as community structure by analyzing soil samples collected at different stages of cotton crop growth. Soil physical and chemical properties were analyzed. Enzymes are the direct mediators of soil health; therefore dehydrogenase activity of different soil samples was compared. Application of molecular techniques has revolutionized the soil microbial studies by bypassing the need of culturing the microbes. Community amplified ribosomal DNA restriction analysis (ARDRA) using universal 16S rDNA primers revealed close similarity of samples collected on 60<sup>th</sup> day and onwards. Zero day sample was distinct from other samples. Data obtained from the dehydrogenase assay and ARDRA fingerprinting did not correlate. It is advised that multidimensional/polyphasic approach should be adopted to fully understand the soil biological processes.

## Key Words

Microbial community, soil properties, dehydrogenase, soil DNA, 16SrDNA, ARDRA.

## Introduction

Soil is an integrated system constituting various interdependent physical, chemical as well as biological processes that are markedly influenced by environmental factors. Healthy soil is a prerequisite to a strong agricultural economy. Soil microorganisms providing the biological interface with the soil physical and chemical environment; affect the environment and in turn, get affected by it. The soil microbial community is organized in complex food webs and stabilizes various soil processes including the biogeochemical processes. It is of great practical significance to observe and compare temporal microbial community diversity in an agricultural field throughout the cropping period of a crop in order to identify the factors that influence the temporal microbial community structure and function. The bacterial diversity associated with the agricultural cotton field crop was investigated using culture-independent approaches like ARDRA and DGGE of the soil community DNA. Dehydrogenase activity was measured, as enzymes are also direct mediators of soil mineralization and processes.

## Methods

### *Experimental site, soil sampling and storage*

The study site was the experimental field of cotton crop in Indian Agricultural Research Institute, Pusa, New Delhi, India. Eight random subsamples were collected from the field (0-10 cm depth) using rectangular sampler (5 x 5 x 10 cm), pooled, sieved (2mm mesh size) and stored at -80°C. Soil sampling has been done 7 times corresponding to 0 day, 15<sup>th</sup> day, 30<sup>th</sup> day, 60<sup>th</sup> day, 90<sup>th</sup> day, 120<sup>th</sup> day and 150<sup>th</sup> day after the day of sowing (0 day).

### *Analysis of physical properties of soil*

All the analysis were done in triplicates.

- 1) *Dry matter and water content*: Determined by the weight loss method (Schichting and Blume 1966).
- 2) *Maximum water holding capacity*: 50g moist soil samples were saturated with water. From each cylinder, 25 grams of soil was taken in porcelain dishes, dried to constant weight at 105°C for 3 hours, cooled in desiccator and weighed. It is expressed in terms of gram water x 100. % WHC = (saturated soil-dried soil)/dried soil x 100
- 3) *Particle size distribution*: Percentage was calculated using the following formula.  
Sand fraction % = Weight (g) of fraction on sieve/25 x 100; Silt fraction % = Weight (g) of fraction under sieve/25 x 100; Clay fraction % = Oven dried soil weight (g)/25 x 100

### *Analysis of chemical properties of soil*

All the analyses were done in triplicates.

- 1) *pH*: Saturated soil solution (1:2.5::soil: water) was prepared and pH was determined using a glass electrode pH-meter.
- 2) *Organic carbon*: It was calculated by titration protocol of Walkley and Black (1934).
- 3) *Exchangeable sodium*: It was calculated by flame photometry.
- 4) *Exchangeable potassium*: It was calculated by flame photometer.
- 5) *Available nitrogen*: It is present in the form of  $\text{NH}_4^{4+}$ ,  $\text{NO}_3^{-}$  and  $\text{NO}_2^{-}$  (nitrite doesn't contribute significantly) in the soil and can be extracted and measured spectrophotometrically (Keeney and Nelson 1982).
- 6) *Available phosphorus*: It was calculated colorimetrically (Olsen and Sommers 1982).

### *Dehydrogenase activity of the soil*

Dehydrogenase activity was measured in triplicates by adding 2.5 mL of sterile ddw and 1 mL of 3% aqueous solution of triphenyl-tetrazolium chloride to 6 g of soil sample, followed by incubation at 30°C for 24 hrs in dark and methanol extraction afterwards. After determining the absorbance at 485 nm, the amount of TPF produced was calculated by reference to a calibration graph prepared from TPF standards (100 µg of TPF/mL methanol).

### *Community DNA extraction and quantification*

Soil microbial community DNA was extracted in triplicates using direct lysis based on the method of Zhou et al. 1996 (Williamson, personnel communication) and checked using standard marker by agarose gel electrophoresis (0.8%). The DNA was gel extracted using QiaexII kit (Qiagen, Germany). Nanodrop Spectrophotometer ND-1000 quantified the DNA prior to any further analysis.

### *Polymerase Chain Reaction Amplification of 16S rDNA*

PCR amplification from 50 ng of extracted soil DNA was conducted with a total volume of 50 µl by using universal primers 27f and 1492r in triplicates (Martin-Laurent *et al.* 2001, Suzuki *et al.* 2005) in PTC-200 (MJ Research). Other reagents were 200 mM of each dNTP, 2 U of DNA polymerase and 1× PCR buffer under the following conditions: 5 min at 94°C, 35 cycles of 1 min at 94°C, 1 min at 55°C, and 2 min at 72°C, plus an additional 15-min cycle at 72°C.

### *Amplified Ribosomal DNA Restriction Analysis*

Pooled 16S rDNA amplicons were concentrated using Microcon-PCR centrifugal devices (Millipore Corp., USA) and subjected to *HaeIII* ARDRA. 10 µl amplicons were restricted overnight with 5U of restriction endonuclease. ARDRA profile was checked on 2.5% Metaphor agarose gel electrophoresis, converted into a 2-dimensional binary matrix and analyzed using MVSP 3.1. UPGMA dendrograms were constructed by calculating Jaccard's coefficient of similarity.

## **Results**

Soil characteristics were determined by analysis of physical and chemical properties of the soil sampled on the 15<sup>th</sup> day (Table 1). DNA yield/g soil was calculated as microbial biomass (Table 2) that is showing no significant variation among the samples and pure DNA was extracted in nearly all the samples after gel extraction.

**Table 1. Analysis of soil physical and chemical properties.**

Soil	pH	Organic Carbon (%)	Available Nitrogen (kg N/ha)	Available Phosphorus (kg P/ha)	Available Potassium (kg K/ha)	Exchangeable Sodium (me/100g)	Particle size distribution			Soil Type	Water Holding capacity (%)
							Sand	Silt	Clay		
							-----%-----				
Cotton field soil	7.7	0.58	221	44.8	467	1.1	58	29	13	Loam soil	52.7

Humic acids and other geochemicals were also got co-extracted with the community DNA (Figure 1), so the necessity of gel elution method can't be ruled out. The PCR amplification was quite difficult in the 90<sup>th</sup> day soil. Figure 2 showed the pattern of variation of the dehydrogenase activity corresponding to the different phenological stages of crop plant. Dehydrogenase activity is direct indicator of microbial activity. A plateau of highest microbial activity was observed during 30<sup>th</sup> day (6.56 µg/mL) and 60 day (6.79µg/mL) after

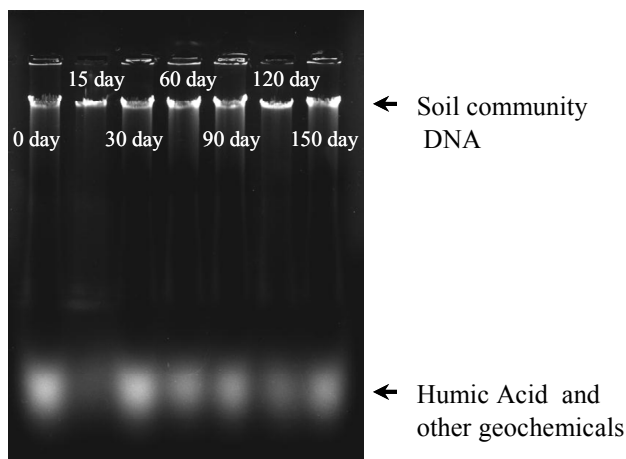


sowing. The 0 day activity was lowest as 4.84 µg/mL which rose to 6.79 µg/mL measured on 60<sup>th</sup> day and again fall down to a value of 5.47 µg/mL before reaching to 6.02 µg/mL on the 150<sup>th</sup> day of sampling. ARDRA profile of 16SrDNA amplicons revealed marked similarity of 30<sup>th</sup>, 60<sup>th</sup>, 90<sup>th</sup>, 120<sup>th</sup> and 150<sup>th</sup> day sample. 15<sup>th</sup> day sample was also almost 95% similar to the rest. Only 0 day sample was showing deviation of around 75% from the rest of the group revealing the microbial community profile of 0 day was very much different from the rest of the samples indicative of shifting in the bacterial diversity across the cropping period (Figure 3). Figure 4 depicted the presence of various bands of different sizes in all the samples.

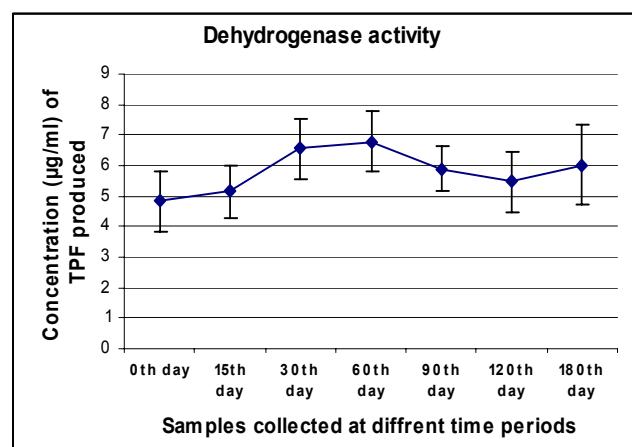
**Table 2. Estimates of microbial biomass in terms of DNA yield (ng DNA /g soil), absorbance ratio and PCR results associated with the samples.**

Soil sample collected	Microbial Biomass (DNA extracted, ng DNA g/soil) <sup>A</sup>	Absorbance at260 nm/ Absorbance at280 nmc	Absorbance at260 nm/ Absorbance at230 nmd	16S rDNA amplification results b
0 day (16th May 2006)	297.8±16.6	1.38	0.29	40272
15 day	284.1±28.4	1.75	0.13	40272
30 day	255.4±20.1	2.18	2.44	40272
60 day	314.6±16.4	2.03	0.57	40272
90 day	279.4±31.7	2.27	-1.34	40271
120 day	312.6±19.7	1.58	0.26	40272
150 day	300.8±27.0	1.89	0.78	40272

<sup>A</sup>DNA yields represent the averages ± standard deviations of DNA extracted in triplicates. <sup>b</sup>Amplification results are presented as the number of DNA samples (soil samples) yielding the desired size amplified product divided by the number of (DNA samples) soil samples analyzed. <sup>c</sup>High 260/280 ratio (>1.7) indicates pure DNA. <sup>d</sup>Low 260/280 ratio indicates protein contamination. <sup>e</sup>Low 260/230 ratio indicates humic acid contamination.



**Figure 1. Soil Community DNA extracted from the samples collected at different stages of plant growth.**

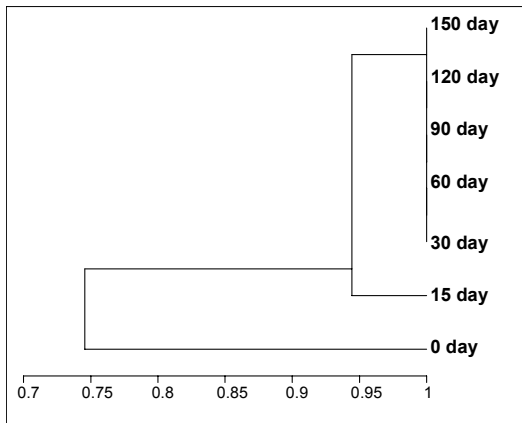


**Figure 2. Soil dehydrogenase activity assayed at different stages of plant growth.**

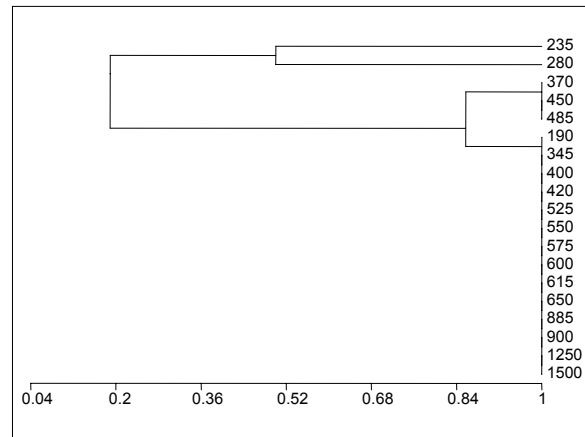
370bp, 450bp and 485bp bands were noticed in all the samples except for the first sample which only as showing the presence of 280bp band. Overall comparison of the first and the last sample corresponding to the day of sowing and day of harvesting respectively revealed marked variation the microbial community structure.

### Conclusion

Noticeable impact of plant growth stages as well as routine agricultural practices (chemical inputs, irrigation, seasonal variations, etc.) was observable on microbial activity and community structure which are constantly changing throughout the cropping period corresponding to the different phenological stages of the plant. Changes in chemical composition of the plant root exudates with the growing age definitely decide the microbial community structure and diversity around the rhizosphere. No correlation was found between the microbial activity and community structure. Only 16S rDNA approach is not sufficient in describing the dynamics of microbial ecology in an agricultural field. So, multidimensional/ polyphasic approach should be adopted in understanding soil biology aspects.



**Figure 3. Cluster analysis of community ARDRA band patterns using Jaccard's coefficient of similarity and the UPGMA method of tree construction. The labels represent the time period of the sample collection considering 0 day as the day of sowing and 150 days as the day of harvesting.**



**Figure 4. Cluster analysis of community ARDRA band patterns using Jaccard's coefficient of similarity and the UPGMA method of tree construction. The labels represent the approx. band sizes of the restricted ARDRA fragments.**

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# Economic value of improved soil natural capital assessment: a case study on nitrogen leaching

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## Abstract

Soil survey is fundamental to assessing soil natural capital. However, over recent years there has been minimal investment in improving the quality of NZ soil survey, possibly due to poor articulation of the economic value. This paper demonstrates a positive benefit to both the farming and general community from the combination of a new soil survey, nitrogen leaching measurements, and a new mitigation technology to reduce N-leaching from dairy grazed pasture. In our study area the annual N-leaching is estimated to be approximately 25% greater than if estimated using data from the old soil survey. We argue that if nitrification inhibitors are applied to only 25% of our study area, the overall reduction in N-leaching can be improved by 10 t N/yr if the new soil map is used to target inhibitor application to the hotspot soils with the greatest N-leaching. We estimate the retained N is worth \$42.49 per kg N for the farmers and the community. From this benefit alone, the cost–benefit ratio for the new soil survey is 1:6 in the first year. This study demonstrates the value of soil survey in soil natural capital assessment and its ability to provide a quick return on investment.

## Key Words

Soil survey, soil natural capital, cost–benefit, nitrogen leaching.

## Introduction

New Zealand (NZ) has had a patchy history of soil survey. The need for improved soil survey is well recognised within the land management industry (Manderson and Palmer 2006), but has yet to materialise as substantial investment. This may be due to the economic value of NZ soil survey not being clearly articulated to potential investors. Worldwide there are few studies that demonstrate the economic value of soil survey (Craemer and Barber 2007; Giasson *et al.* 2006). Craemer and Barber (2007) argue that if clear prospects exist for improved yields or farm returns, the private sector should have sufficient incentive to invest. They argue that the business case for public investment in soil information needs to be strongly linked to market failure and public good arguments. In Australia strong arguments for public investment exist in terms of basic research and development (e.g. natural capital assessment), externalities (e.g. groundwater pollution), and information failure (e.g. getting research findings to potential adopters). Investment may also be justified if data underpin an information value chain (i.e. basic research → applied research and innovation → end-user products and processes).

The concept of an information value chain is illustrated by the success of focus farms in NZ. Mackay *et al.* (1998) demonstrated a substantial potential economic return from using soil survey information to identify land management units on individual farms, for which the most suitable management practices could be matched. Cost–benefit analysis identified that if only 10% of NZ farmers adopted this approach and lifted profitability by 8%, then the return would be \$20m per year over a 20-year period. A similar study analysed the cost–benefit ratio (CBR) of the monitor farm programme (MFP), where focus farms relevant to different geographical areas are used to demonstrate the value of new management techniques to the local farming community (Garland and Baker 1998). Local farmers reported a net benefit of \$6,500 per year, resulting in a CBR of 1:20 in the first year.

In both these studies it is not possible to evaluate the net benefit arising directly from improved soil knowledge alone, as these projects integrate a range of management techniques. However, both of these studies support the argument that the value of improved soil survey extends beyond an assessment of soil natural capital, to underpinning an information value chain that identifies research findings relevant to a particular farm. The objective of this paper is to demonstrate the economic value that could arise from an improved assessment of soil natural capital in relation to nitrogen (N) leaching and the targeted application of a mitigation technology.

## Materials and methods

### *Study area*

The study area is located on the floodplains of the Mataura and Oreti rivers, Southland, New Zealand. The ecological health of these rivers is of high importance to the Southland community, with most of the population living in towns located adjacent to the rivers. The Oreti River is the water supply for Invercargill City, and the rivers have high recreation and ecological importance, both with renowned fisheries. Environment Southland (2008) report high ecological health in the upper reaches, but the middle and lower reaches are in poor health, exceeding the nitrate-nitrite nitrogen, total phosphorus, faecal coliform, and visual quality national standards set for lowland rivers (ANZECC 2000).

Down-river trends in river health follow land use. In the upper catchment land use is mostly low intensity sheep and beef grazing or conservation land. In the middle to lower reaches intensive pastoral agriculture is the dominant land use, with an even distribution of sheep (32%), mixed sheep and beef (21%), and dairy (30%). In this study we focus on the 17 706 ha of land used for dairy farms, which are recognised as “hotspots” responsible for most non-point-source N pollution from agricultural land in NZ (Monaghan *et al.* 2008). Dairy farming is a growth industry in Southland, doubling in land area over the last decade to effectively 130 000 ha in 2008 (LIC 2008).

### *Soil survey*

Until recently our study area was reliant on soil information from a soil survey at 1: 250 000 scale. Map units and soil types were carried through in later land resource inventory maps at 1:63 360 scale, which are still used today for national planning. In 1998–2001 there was a major community initiative to remap 800 000 ha of the Southland lowlands at a scale of 1: 50 000. In 2001 the total cost of the new soil survey was c. \$2.5m, or c. \$3.13 /ha (S Carrick, unpublished data). This cost includes field survey, laboratory analysis, and map production.

### *The ecosystem service of nitrogen retention*

Nitrogen leaching is calculated for the study area based on the old and new soil surveys. Greenwood (1999) measured nitrate leaching from dairy grazed pasture on different Southland soil types over a one-year period (1998–1999) and under the same stocking intensity (2.4 cows/ha). This study showed marked differences in N-leaching between soil types under similar management (Table 1). These results correlate with later research where N-leaching from cow urine patches was approximately double on the stony compared with deep soils (Di and Cameron 2005, 2007).

Monaghan *et al.* (2008) evaluated the economic return to dairy farmers from N-leaching mitigation techniques across four NZ catchments. Nitrification inhibitors were the most promising mitigation technique, with a net benefit in 2005 of \$16 per kg N retained for the case study farm in the Waikakahi catchment, which has similar soils to our study area. Monaghan *et al.* (2008) assumes the inhibitors achieve a 30% reduction in N-leaching, which is much lower than experimental results (c. 50–70%) but is a conservative reduction generally accepted by industry to recognise that more research is required before specific reductions can be quantified for different environments and management.

An estimate of the economic value for the general community of retaining N can be transferred from the N-trading scheme established for Lake Taupo, NZ. Within the catchment a nitrogen leaching cap has been implemented, allowing farmers a maximum Nitrogen Discharge Allowance (NDA). The only official trader at present is the publicly funded Lake Taupo Protection Trust; set up to achieve a 20% reduction in nitrogen loading by 2021. The trust will pay farmers to permanently reduce nitrogen leaching through land-use change. In 2004 the budgeted average cost for compensation was \$425 per kg N (Environment Waikato 2007), which over an infinite lifetime and 5% discount rate equates to \$21.80/kg N/yr.

The improved estimation of N-leaching in our study area allows targeting mitigation to high-N-leaching soils. The value added by targeted mitigation is compared with the cost of the new soil survey, with both standardised as 2009 NZ\$ by adjusting for inflation. In our calculations the 2009 soil survey cost is \$3.99 per hectare, and the value of the retained N is \$42.49 per kg (\$17.30 per kg for farmers, \$25.19 per kg for the community). We assume that inhibitors achieve a 30% reduction in N-leaching.

## Results

### *Natural capital assessment*

Management of non-point-source pollution is dependent on a reliable inventory of the soil natural capital. In our study area the old soil map identified a single soil type, characterised as a well-drained Recent Soil formed into deep fine alluvium (Table 1). The new soil map shows that less than 17% of the area is the original soil type. Most of the area was mapped as either stony or poorly drained soils.

**Table 1. Comparison of soil attributes and annual nitrate leaching from our study area, when using data from either the old or new soil survey.**

Map	Soil type	NZSC order <sup>A</sup>	Depth of fines	Drainage	Area (ha)	Area (%)	N-leaching (kg N/ha/yr)	Estimated study area total N-leaching (t N/yr)
Old	Recent	Recent	Deep (>0.45 m)	Well	17706	100	50	885
New	Recent	Recent	Deep	Well	2946	17	50	147
	Stony	Recent + Brown	Stony (<0.45 m)	Well	5020	28	70	351
	Brown	Brown + Pallic	Deep	Well	3702	21	50	185
	Gley	Gley	Deep	Poor	4940	28	70	346
	Pallic	Pallic	Deep	Poor	534	3	70	37
	Gley	Gley	Stony	Poor	512	3	70	36
	Peat	Organic	Deep	Very Poor	53	0.3	Not studied	

<sup>A</sup>New Zealand Soil Classification (Hewitt 1998)

### *Ecosystem service of nitrogen retention*

Based on the results of Greenwood (1999) the new survey estimates N-leaching in the study area of 1103 t N/yr, which is 24.6% greater than if predicted from the old soil map. The estimate of N-leaching is likely to be conservative, as the average 2008 stocking intensity was 2.8 cows/ha (LIC 2008), higher than the 2.4 cows/ha in Greenwood (1999). Table 1 does not also take into account N-leaching from paddocks that receive applications of dairy shed effluent, where Greenwood (1999) measured N-leaching to be 28–57% greater than grazed paddocks.

### *Economic value of the new soil map*

The middle to lower Mataura and Oreti rivers have poor ecological health, and it is arguable that farmers and the community are legally obliged to improve water quality in order to meet national standards. Nitrification inhibitors provide one option to reduce N-leaching and give a positive economic return to farmers (Monaghan *et al.* 2008). The new soil map shows that the hotspots of N-leaching are the poorly drained and stony soils (Table 1), and therefore these should be targeted for inhibitor use.

If the farmers and community decide an achievable target is to apply inhibitors to 25% of the study area, and the new soil map was used to target the hotspot soils, then the expected reduction in N-leaching would be 93 t N/yr. Without the new soil map the expected N-leaching reduction would be less, as hotspot targeting would not be possible, and we would expect at least 38% of the inhibitor to be applied to soils with lower N-leaching (Table 1). As such the expected reduction in N-leaching would fall to 83 t N/yr. Under this scenario, use of the new soil map is able to improve the reduction in N-leaching by 10.25 t N/yr. The cost of surveying the study area is \$70,605, meaning a 10.25 t saving of N would require \$6.42 per kg N in added benefit to recover the survey costs in the first year. We have estimated the added benefit of the retained N is \$42.49/kg N/yr, which is well above that needed to recover the survey costs in the first year. We recognise the value of retained N may vary between catchments, but it is unlikely to be substantially lower in our study area given the concerns over water quality.

Using our estimate of the value of the retained N, the CBR of the new soil map would be 1:6 in the first year of targeting inhibitor application to hotspot soils. However, application of inhibitors to 25% of the study area only achieves an 8.4% reduction in the total N-leach. If research shows that improved water quality requires the N-leach reduction to be at least 15%, as is the case in the Lake Taupo catchment, then it would be necessary to apply inhibitors to about 50% of the area. If all of the application was targeted at the hotspot soils identified in the new soil map, the CBR for the first year would increase to 1:13.

## Conclusions

This paper demonstrates a net positive benefit to both farmers and the general community from the combination of a new soil survey and a new mitigation technology to reduce N-leaching. It would appear that there is a sound business case for joint investment by the private and public sectors to improve the quality of NZ soil survey. The study also underlines the need for the development of value chains that will enable economic benefits of new knowledge on soil natural capital to be realised. While this paper demonstrates the economic benefit of the new soil map, it accounts for only one ecosystem service. Accounting for other soil services might add further benefits.

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# Enhancing the ecological infrastructure of soils

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## Abstract

The recent financial crisis has led to massively increased investments in built infrastructure as a means of rapidly stabilising and reinvigorating economies around the globe. However, just as built infrastructure delivers a range of socio-economic services that underpin modern societies, there is also an ecological infrastructure that maintains the provision of the ecosystem services that support a wide range of ecological as well as socio-economic benefits. Given the worsening water and food crises and increasing population pressures, one wonders why larger investments are not being made to ensure that our ecological infrastructure has the capacity to continue to produce sufficient flows of ecosystem services to satisfy the world's future needs. A large part of the answer to this question is, despite its importance, that the concept of ecological infrastructure is not yet widely recognised and understood. This paper highlights the importance of investing in the ecological infrastructure of soils. We begin by developing the concept of ecological infrastructure through a comparison of the key elements, systems and services that constitute built infrastructure and ecological infrastructure. We then highlight the role of soils as a fundamental element of ecological infrastructure. We highlight the importance of pore connectivity and soil water flow and transport as essential features of a robust and resilient soil ecological infrastructure that can be invested in, and enhanced, through carbon investment strategies.

## Key Words

Ecological infrastructure, ecosystem services, soil, macropores, water and food crises.

## Comparing built and ecological infrastructure

Water scarcity, projected climate change impacts, the worsening global food crisis and the global financial crisis are powerful drivers for major investments in water and other built infrastructure. Many people now have direct and regular access to a variety of socio-economic services that this type of infrastructure provides. Water, energy, transport and communications infrastructure (Table 1) is used by so many of us so often that we consider them to be essential (Australian Government Treasury 2004)

**Table 1. Built infrastructure, associated systems, and the services and benefits they provide.**

Infrastructure*	Systems	Services	Benefits
Water	Dams, channels, treatment plants	Water for urban, agricultural, industrial use	Sufficient quality water; Flood mitigation
Energy	Power stations, power lines	Generation, storage, transmission of energy	Energy for construction, maintenance & equipment operation
Transport	Road, rail, terminals ports	Despatch, delivery, receipt of goods & services	Access to goods, services and travel
Communication	Transmitters, cables, receivers, satellites	Information storage, transport and delivery	Connecting individuals, organizations across space and time

\* Also includes health, education, industry, defence and other built infrastructure

Investing in built infrastructure provides increased capacity for the delivery of various services required by growing populations. In addition, built infrastructure investments are used to stimulate rapid economic growth, and billions of dollars are now being invested by a number of countries in a wide range of public and private infrastructure developments as part of their response to the global financial crisis.

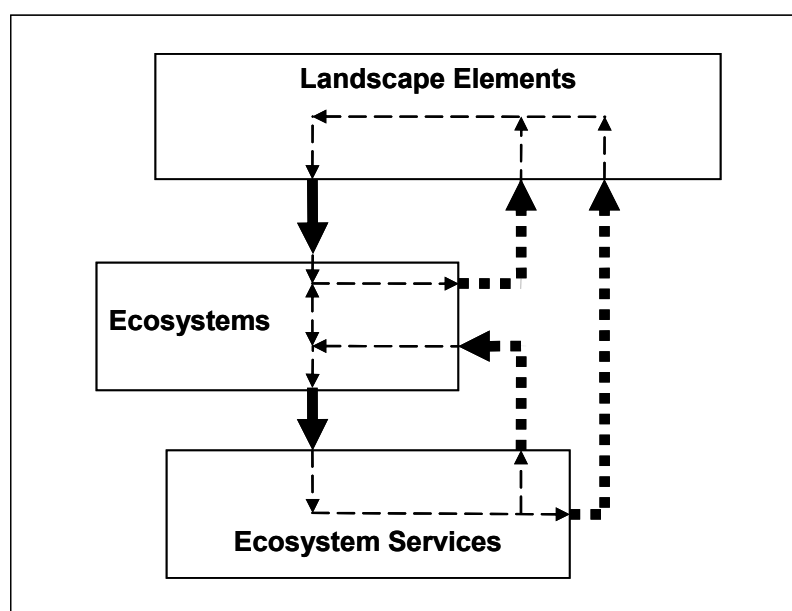
As with built infrastructure, we note that rivers, soils, aquifers, wetlands and other landscape elements are key components of an 'ecological infrastructure' that supports the continuing delivery of ecosystem services required by natural systems for their survival, and mankind for human well-being (Table 2).

**Table 2. Ecological infrastructure, ecosystems, and the services and benefits they provide.**

<b>Ecological Infrastructure*</b>	<b>Ecosystems</b>	<b>Ecosystem Services</b>	<b>Ecosystem Benefits</b>
Rivers	River ecosystems	Water delivery within and between elements (and ecosystems)	Provides water, sediment, nutrients to floodplains, wetlands, aquifers, estuaries; and habitat
Aquifers	Aquifer ecosystems	Water capture, storage, purification, dilution (underground)	Soil moisture, stream flow, wetlands (base flows) in dry seasons
Wetlands	Wetland ecosystems	Water storage, filtration and purification	Inception & dilution of non beneficial organic & inorganic materials; habitat
<i>Soils</i>	<i>Soil ecosystems</i>	<i>Support medium, storage and supply of water &amp; nutrient for plants; waste treatment/removal</i>	<i>Maintain (&amp; increase) soil biological and vegetation productivity &amp; biodiversity</i>

\* Also includes catchments, forests, rangelands, vegetation, floodplains, estuaries etc

Ecological infrastructure consists of landscape elements, ecosystems, ecosystem services and the interconnections within and between them (Figure 1).



**Figure 1. Ecological infrastructure consists of landscape elements, ecosystems, ecosystem services and the interconnections within and between them.**

We argue that growing populations will require an increase in the capacity of existing ecological infrastructure if it is to continue to produce the range of ecosystem services necessary for our present living standards to be maintained and improved. Even though the importance of maintaining the flow of ecosystem goods and services is now well-established in the literature (Costanza *et al.* 1997), the role and importance of the ecological infrastructure that sustains the ecosystems that provides the goods and services is barely recognised (Postel 2008).

While our investments in built infrastructure have been ever-increasing, we have not been investing sufficiently in our ecological infrastructure. Inadequate investment in ecological infrastructure has led to a worsening environmental crisis, in which critical ecosystem services have been and are being lost in many regions across the globe. For example 60% of ecosystem services examined by the Millennium Ecosystem Assessment in 2005 were found to be degraded (<http://www.millenniumassessment.org/en/Index.aspx>). Some world famous examples of the kind of environmental degradation resulting from a failure to understand and invest in ecological infrastructure include Lake Chad, the Aral Sea and Easter Island.



In practical terms, investing in ecological infrastructure should include objectives, actions and outcomes aimed at identifying those areas that are most suitable for development, with a primary focus on the regenerative capacity of natural systems to continue to support human socio-economic requirements. Sufficient investment in ecological infrastructure will therefore involve strategic and targeted investments aimed at:

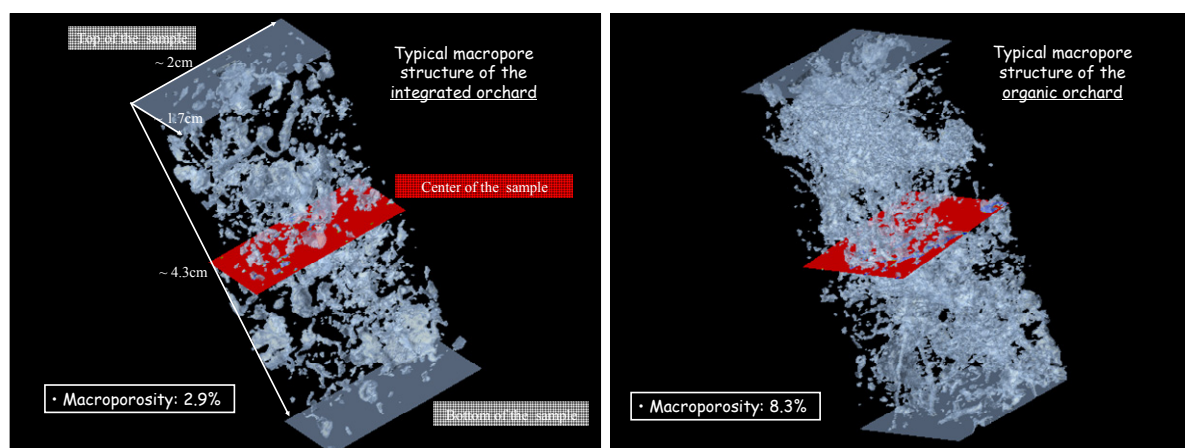
- gaining a better knowledge of the structure, function and processes of ecological infrastructure
- the restoration of degraded or degrading ecological infrastructure, and
- maintaining the resilience and regenerative capacity of ‘undisturbed’ ecological infrastructure<sup>1</sup> in the case of future developments.

We may also need to find a way of enhancing the capacity of ecological infrastructure if we continue to place ever-increasing demands upon it.

### Enhancing ecological infrastructure: Understanding and investing in soils

Soil is a primary ‘filter’ of the world’s water and through this plays a critical and valuable role in determining the quality and quantity of groundwater and surface water (Clothier *et al.* 2008). It is the size and shape of soil pores and their connectivity that either enhances or curtails the capacity of soil to buffer and filter, so understanding and investing in soils to ensure beneficial soil structure is critical. Clothier *et al.* (2008) demonstrated the importance of this and suggested that macropores which support preferential flow and transport underpin 12 of 17 ecosystem services provided by soils. Jarvis (2007) provided an extensive review of the principles and controls on preferential flow and transport in soil. Clothier *et al.* (2008) then estimated the global value of the ecosystem services provided by the soil’s macroporous infrastructure to be some US\$304 billion per year.

Given the significant value of these services provided by macropores, it is critical that we increase both our understanding of, and the investment needed for initiating and sustaining soil macropores. A prime way of achieving this is through carbon investment in soil. Potentially this is a win/win situation: improved ecological infrastructure and carbon capture and storage. Robinson *et al.* (2009) have highlighted the need to improve understanding and definition of soils’ ‘natural capital’. Their definition recognises the quantity, quality and dynamic behaviour of the various components making up the soils’ natural capital, but appears to differ somewhat from our definition of ecological infrastructure in that it does not highlight issues of ‘connectivity’ as a key component of natural capital. Nevertheless, it is a major step forward in highlighting the need for more work on understanding those aspects of soils – their ecological infrastructure – that supports the delivery of ecosystem services. Figure 2 shows X-ray tomographs of two identical soils from neighbouring orchards (genoforms). Different carbon-investment strategies used in these different orchards have changed the ecological infrastructure (phenoforms) of the soils. Consequently, the two formerly identical soils now perform quite different ecosystem services due to the altered macroporous infrastructures and connectivity within them.



**Figure 2.** The X-ray tomographic images of two identical soils that have undergone different carbon-investment strategies, resulting in a different ecological infrastructure with relation to macroporosity

<sup>1</sup> All ecological infrastructure has now been disturbed, at least to some degree, by human activities, for example global climate change

The results of these different carbon investment strategies show that investing in ecological infrastructure can increase the ecosystem-services and hence the benefits provided by soils. This research is focused only on a particular component of soil infrastructure, but the findings highlight the potential benefits of improving our understanding of the structure and function of ecological infrastructure and investing in it.

## Conclusions

Ecological infrastructure underpins the delivery of ecosystems services required by natural systems for their survival, and mankind for human well-being. We have argued that growing populations will require an increase in the capacity of existing ecological infrastructure if present living standards are to be maintained and improved. But while investments in built infrastructure have been ever-increasing, we have not been investing sufficiently in our ecological infrastructure.

Soil is a critical component of ecological infrastructure and this paper reveals how soils' production of ecosystem services can be enhanced through carbon investment strategies. The next challenge is to

- further develop our understanding of the soil's infrastructure and particularly connectivity, and
- use that understanding to implement appropriate investment strategies in ecological infrastructure to ensure delivery of the range of ecosystem goods and services humans depend upon, many of which are currently taken for granted.

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# Evaluation of soil natural capital in two soilscapes

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## Abstract

A stock adequacy method is presented for evaluating the soil natural capital in two contrasting soilscapes. The approach is to estimate the adequacy of soil natural capital stocks to support the soil services required by a specified land use. A stock adequacy index is defined to identify whether soil services are limited by soil natural capital stocks or have a stock surplus. Limiting values are derived from a stock quality–quantity curve determined from land evaluation or soil quality literature, or by modelling. The method is applied to eroded soils in a hill country soilscape, and a coastal sands soilscape. The derived stock adequacy index results expressed the principal variations in quality of the soil resources that serve land management in the two locations. The index is capable of being integrated into land resource assessments and provides a basis for economic valuation of soil natural capital.

## Key Words

Soil natural capital, soil stocks, land evaluation, soil services.

## Introduction

Soil natural capital (SNC) will need to be quantified and mapped at different scales if we are to use the concept to assess and value soil assets and soil services. In this paper we outline a method to quantify SNC in two contrasting soilscapes. We explore the spatial and temporal variation of SNC as a necessary precursor to mapping SNC. We define soil natural capital (adapted from Dominati *et al.* 2009) as the capacity of soil to provide the soil stocks needed to underpin the soil services required by a specified land use. The natural capital of a soil is quantified by a set of morphological, chemical, physical and biological properties that quantify the status of the relevant soil stocks. In this study we have chosen a minimal set of soil services, stocks and associated properties to demonstrate a method to quantify and use that quantification to characterise the variability of SNC in two study areas. The ultimate goal is to quantify the economic value of SNC. The method presented here provides an index of SNC value to which an economic value may be assigned. The index must be capable of being efficiently derived from soil and land resource evaluations.

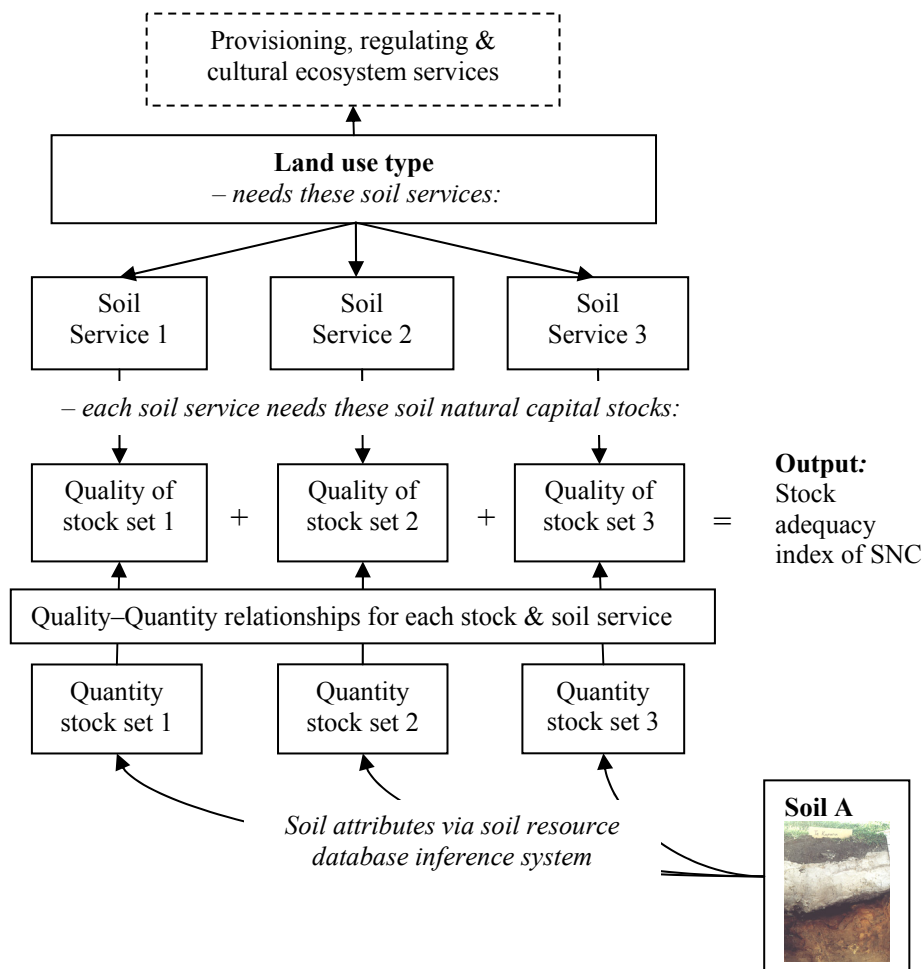
## Methods and materials

The proposed method for evaluating SNC combines the principles used in land evaluation (Rossiter 1996) and soil quality (Sparling *et al.* 2004). It includes the following steps:

- Define the land use type (LUT).
- Define soil services required to support and manage that LUT.
- Define the SNC needed to sustain each soil service in terms of a set of soil stocks.
- Quantify these stocks for each soil type.
- Estimate the quality of each stock to adequately support a specified soil service. The measure of quality is characterised as a stock adequacy index.
- Aggregate stock quality levels across the soil services to derive an aggregated estimate of SNC for the land use. In this study the aggregate is the mean index across all services.

The procedure is outlined in Figure 1.

The method is based on the premise that soil services may be limited by one or more inadequate soil stocks. The stock adequacy index quantifies the quality of a stock to support a soil service. Estimation of the index involves two steps. In step one, a level must be determined for each stock that is adequate for unlimited operation of the chosen soil service. This amount of stock is assigned a quality value of 100%. Values greater than 100% signify a stock surplus and non-limitation of the dependent soil service. The 100% index level is estimated from either (a) land evaluation and soil quality literature recommendations for high class or high quality soils, (b) a site potential value as in the case of soil organic matter where the highest likely level for the site and specified land use is chosen, or (c) a soil process model that represents the soil service that relates stock input to process output. In step two less than adequate stock quality levels are assigned a



**Figure 1. Outline of procedure for evaluating soil natural capital (SNC) for a specified soil and land use type. The stock adequacy indices can be summed over all services to derive an overall SNC index for the soil – land use type combination.**

percentage based on a stock quality – stock quantity curve. In this study the 100% stock quality levels were derived from land evaluation “soil qualities” (Webb and Wilson 1995), and soil quality evaluation curves (Sparling *et al.* 2003). Establishing these levels requires further research. SNC stocks are assumed to include soil capacities, such as available water capacity that is dependent on porosity, as well as soil materials such as carbon. There was no distinction made in this study between soil stocks built up by managed SNC and inherent SNC.

### Soilscapes

Soil natural capital was estimated in two contrasting soilscapes: soft rock hills and coastal sands. Soils of the soft rock hills are described by Vincent and Milne (1990). The soils are developed in weakly indurated siltstone on step hills with average slopes 28 degrees. The land use is rain-fed pastoral sheep grazing, and the soil mantle is subject to soil slip erosion. The major driver of soil variability is the presence or absence of soil erosion. In uneroded sites the soils are Argillic Pallic Soils (Hewitt 1998) with soil spatial variability related to slope position. Soils on crests are well drained and soils on slopes with redox mottled subsurface horizons. In eroded sites the soils are either Recent Soils or Raw Soils with paralithic contacts at shallow depth. Soil temporal variability is related to time since erosion disturbance. Sites were studied that had been eroded at four periods (Lambert *et al.* 1984). We only included hill slopes in this study and did not consider valley floors or erosion accumulation areas.

Soils of the coastal sands soilcape are described by Cowie *et al.* (1967) on dunes and sand plains along the Manawatu coast. The major driver of soil variability in the undisturbed Sandy Brown Soils and Sandy Gley Soils is the depth to ground-water. The well-drained Foxton soils, imperfectly drained Himatangi soils, and poorly drained Pukepuke soils form a drainage catena. Intensive pastoral and cropping during summer is irrigated, which is supplemented by water via capillary rise from water tables within 1 m of the base of the

root zone. Our calculations of water added via capillary rise assumed the water-table height corresponded to the depth to dominant grey soil redox colours. We did not take into account seasonal fluctuations or root depth extension through the growing season. The land use considered in this study was irrigated maize feed cropping.

### Soil stocks and services

The soil services and related stocks considered in this study are listed in Table 1. In this paper, only key soil services and stocks were considered in order to illustrate the method.

**Table 1. Soil services, the minimum set of supporting soil natural capital stocks studied, and their profile measurement.**

Soil services	SNC stocks
Carbon storage	C (t/m <sup>3</sup> to 600 mm depth)
Profile available water storage	Profile available water capacity (mm; within potential rooting depth)
Aeration	Depth to macropores <5%, or depth to dominant grey matrix, whichever is less (mm)
Capillary rise	Capillary rise estimated as water augmentation (Scotter 1989) (mm)
Cation fertility	Sum of bases (exchangeable calcium, potassium, magnesium and sodium) weighted average to 600 mm depth (cmol/kg)

### Results

Stock quality estimates are shown in Table 2. The carbon storage service is promoted by soil development as shown by the contrast between non-eroded and eroded soft rock soils. It is limited in eroded soils by shallowness and by immature topsoil development. Lambert et al. (1984) showed that the eroded sites had only recovered 77 % of their pre-eroded pasture productivity in 80 years. After an additional 25 years, no further recovery occurred on the eroded sites (Rosser and Ross 2009).

The PAW service is limited in all the soils. Lower PAW soils in rain-fed pastures on the soft rock soils cause earlier reduction of pasture in summer and consequent reduction in sheep carrying capacity. Soils with relatively low PAW soils in irrigated maize on the coastal sand soils confer relatively higher costs due to higher irrigation water requirements (Hedley and Yule 2009). The aeration service promotes good pasture growth in deep soft rock soils but in shallower soils limitations to the service reduce production. In coastal sand soils aeration promotes deep root growth in well-drained sites but in poorer drained sites maize will suffer root extension limitations. The capillary rise service is active neither in the soft rock soils nor in the well-drained coastal sand soils. It is active in less well drained soils where it reduces irrigation costs for maize production. The cation fertility service is highest in the eroded soft rock soils where the cations are supplied from parent material sources. Lower stocks within the rooting zone in the other soils limit fertility and must be augmented by appropriate fertilisers at a cost that reflects the natural stock limitations.

**Table 2. Estimated SNC stock adequacy index ranges (adequate stock = 100%) supporting five soil services in two landscapes.**

Soilscapes And major variations	Soil services						
	Carbon storage	Prof. Avail.	Water	Aeration	Capillary rise	Cation fertility	Mean
Soft rock							
Non-eroded – spatial variation (aspect, slope length)	56–94	52–76	50–90	-	52–64	51	
Eroded – temporal & spatial variation	31–40	36–49	36–52	-	73	31	
Coastal sands							
Drainage – spatial variation	71–104	85–190	22–100	0–139	36–40	73	

### Conclusions

- The method presented provides a quantitative estimate of the quality of SNC to support soil services for any given land use.
- The method provides a quantitative expression of the principal variations in quality of soil resources that serve land management in two study areas.
- The stock adequacy index provides a basis for economic and other measures of SNC value. It has potential for use in quantifying the soil assets of areas of land for incorporation into resource economic analyses.

- The stock adequacy index is standardised and can potentially be used as a basis for comparison across a range of SNC stocks and services.
- The method uses land evaluation and soil quality assessment procedures and is capable of being integrated into land resource assessment, using both traditional soil survey and digital soil mapping approaches.
- The results can be interpreted in terms of the activity of soil services, and the costs of reduced services due to SNC limitations.

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# Farmland protection maps for the northern rivers of North Eastern NSW, Australia-an application of soil landscape information

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The Northern Rivers region of NSW comprises the Tweed, Richmond and Brunswick Rivers catchments. Maps depicting those contiguous areas within this region that are considered to be the better agricultural land (not just prime agricultural land) have been derived from published soil landscape information (Morand 1994; 1996; 2001) and mapping currently underway. The maps were used by the NSW Department of Planning 'Northern Rivers Farmland Protection Project' to establish a system of regional agricultural protection through the planning system. The project is a response to the problem of incremental and substantial loss of agricultural land to urban development. Two categories of farmland protection, State and Regional, were developed. State farmland is country that has a relatively unique combination of quality soils and landforms with a favourable climate. Regional farmland is country that is significant from a regional perspective but not necessarily unique. Each category will be subject to its own specific planning rules. A third category is 'other rural land', which may include small pockets of better quality land. National parks and state forests are excluded.

Soil landscape mapping undertaken by DECCW and its predecessor organisations was chosen as the base data for the mapping because:

- There is complete coverage of the Northern Rivers-time and resource constraints did not allow a mapping programme dedicated specifically to farmland protection.
- Each soil landscape is discriminated in terms of soils, landform, geology and, to a lesser extent, vegetation.
- A broad land capability ranking is allocated to each soil landscape.
- Soil type, soil fertility, landform, land capability and natural hazards were the primary factors considered when determining which soil landscapes were to be selected - all these factors are considered in each soil landscape description.
- The Farmland Protection map could be derived from relatively simple manipulation of the soil landscape maps.

## Soils and landforms

State: soils are predominantly Red Ferrosols (Nitisols) forming on Tertiary basalt. Landforms are generally rises to low hills with slopes <15%. Regional: soils are mixed, but include Red Ferrosols (Nitisols) on the steeper basaltic country. Other soils are Brown and Black Dermosols (Phaeozems), Vertosols (Vertisols), Hydrosols (Gleysols, Fluvisols) and Kurosols (Acrisols, Planosols). Landforms can vary from alluvial plains, to rises, to rolling low hills and hills with slopes up to 33%. Other Rural Land: variable, but includes good farmland on narrow alluvial plains where size precludes inclusion within the other categories.

## Conclusion

The Farmland Protection Project is a good example of how the considerable amount and variety of information collected for soil landscape maps can be utilised to produce strategic planning maps where time and resources are limited. Although there is a substantial input of time and resources into producing soil landscape maps and reports, their versatility and adaptability for providing more than just soil information is a powerful feature. The soil landscape concept is particularly conducive to the type of application illustrated on this poster.

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# Hydrogeological landscapes – an expert system for salinity management

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## Abstract

Salinisation of land and rivers is a major environmental problem in Australia and around the world. The correct management options to counter this threat are paramount if salinity is to be brought under control. The Hydrogeological Landscape (HGL) concept provides a structure for the understanding of how salinity manifests itself in the landscape and how differences in salinity are expressed across the landscape. The HGL framework is an expert management system that integrates the spatial distribution of salinity processes with the most effective management options for any given area. In the Braidwood area of NSW, Australia, 20 HGLs have been identified and assigned specific management options based on the manner in which salinity is expressed within the landscape.

## Key Words

Salinity, salt stores, water quality, salinity management, EC, Braidwood.

## Introduction

A Hydrogeological Landscape (HGL) spatially defines areas of similar salt stores and pathways to salt mobilisation. The process of HGL determination relies on the integration of a number of factors: geology, soils, slope, regolith depth, and climate; an understand of the differences in salinity development (“plumbing”); and, the impacts (land salinity/ salt load/ EC) in landscapes (Wilford *et al.* in press). Information sources such as soils maps, site characterization, salinity site maps, hydrogeological data, surface and groundwater data are incorporated into standard templates. Each HGL has been assessed for a range of salinity characteristics: including Salt Land, Salt Load (export) and Water Quality as well as overall hazard.

A number of HGL projects are currently underway including the Hydrogeological Landscapes for the Southern Rivers Catchment Management Authority, Braidwood 1:100,000 map sheet (in prep). The Braidwood 1:100,000 sheet study area lies in the southern tablelands of NSW, Australia and covers approximately 2500 km<sup>2</sup>. It is bounded by latitudes 35° 00’S and 35° 30’S and longitudes of 149° 30’E and 150° 00’E. The mapped area encompasses the towns of Braidwood, Windellama, Tarago, Lake Bathurst and Mongarlowe.

## Methods

The methodology used to arrive at a HGL involved a structured comparison of salinity characteristics. These included water pathways through the landscape; salt stores; relative mobility of salt within the landscape; salinisation processes; and, salt signatures within streams. Concept models were developed to describe unique characteristics within each HGL. A multi-staged approach was used to arrive at the HGL units. Firstly existing information was assessed. Information sources such as soils maps, site characterisation, salinity site maps, hydrogeological data, surface and groundwater data were incorporated into standard HGL templates.

The project relied upon a number of different disciplines and skill sets to obtain an integrated understanding of the landscape. Groups involved in the project include geologists, hydrologists, geomorphologists, pedologists, land resource planners and local extension staff. In the Braidwood area the prime method of HGL determination was lithologic boundaries followed by terrain, soils, climate and local knowledge. The Braidwood 1:100,000 Geology map (Fitzherbert *et al.* (in prep)) was used to separate the map sheet into major lithological groups that had similar hydrological properties, regolith depth and weathering characteristics. Bedrock structures including dykes, faults and major lineaments were also used to delineate

HGLs. Field reconnaissance confirmed the depths of bedrock and any salinity manifestations. Soil landscapes (Jenkins 1996) were used to better understand the terrain and the surficial deposits such as the ancient Shoalhaven floodplain sediments and aeolian sand deposits. Field reconnaissance backed up with regolith bore logs confirmed the depths of soils, the nature of soil materials and the presence of any saline scalds. The soil landscapes were also used as a basis for splitting terrain based on modal slope (relief and slope inclination).

Climatic zones were drawn on the provisional HGL map and field reconnaissance was used to ascertain the critical climatic gradients. The vegetation mapping and classification of Keith (2006) was used to cross check provisional HGL units. Groundwater flow systems were determined from the geology, soil landscapes, field work and expert panel assessment. HGL units were verified against field observation, EC measurements, historical bore log data, expert knowledge, local knowledge and known saline site mapping. Once HGL units were established and verified landscape functions were assigned (Table 1). A landscape may provide one or more functions in a catchment context. Catchment scale management involves understanding how functions are maintained, improved or degraded. It is important to consider the full range of salinity and hydrology functions to understand which mix of strategies (Table 2) and related management actions (Table 3) are appropriate for salinity management. Some strategies and management actions could have negative offsite impacts to catchment management unless their applicability to functions is understood.

**Table 1. Landscape function descriptions for the Braidwood area.**

Function	Description
A	The landscape provides fresh water runoff as an important water source
B	The landscape provides fresh water runoff as an important dilutions flow source.
C	The landscape provides important base flow to local streams
D	The landscape generates saltloads which enter the streams and are redistributed in the catchment
E	The landscape receives and stores saltload through irrigation or surface flow.
F	The landscape generates high salinity concentration water
G	The landscape contains important land based assets which are being impacted by salinity processes.
H	The landscape contains high hazard for generating sodic and saline sediment.
I	The landscape contains high hazard for acid sulfate processes.

Management strategies are aimed at maintaining or improving the landscape functions. One or more strategies may be applicable to any landscape in order to maintain or improve the function of the landscape (Table 2).

**Table 2. Management strategies for the Braidwood area.**

	Management strategy
Strategy 1	Buffer the salt store – keep it dry and still
Strategy 2	Intercept the shallow lateral flow and shallow groundwater
Strategy 3	Stop discrete landscape recharge
Strategy 4	Discharge rehabilitation and management
Strategy 5	Increase agricultural production to dry out the landscape and reduce
Strategy 6	Dry out the Landscape with diffuse actions over most of the landscape
Strategy 7	Access and use of groundwater to change water balance
Strategy 8	Maximising recharge to dilute water tables with engineering actions
Strategy 9	Minimising recharge with engineering actions
Strategy 10	Maintaining and maximising runoff
Strategy 11	Manage & avoid acid sulfate hazards

Management actions for salinity deliver on the strategies at an operational level. One or more management actions may be needed to deliver on any strategy. A management action which is highly suitable for delivering on a particular strategy may be unsuitable to deliver on a different strategy.

There are over 100 defined management actions and new management actions are added as required. Management actions are grouped into several categories (Table 3). A feature of the HGLs is the apportioning of management areas (MA) so that specific landform elements within a landscape can be targeted within any given HGL (Figure 1).

## Results

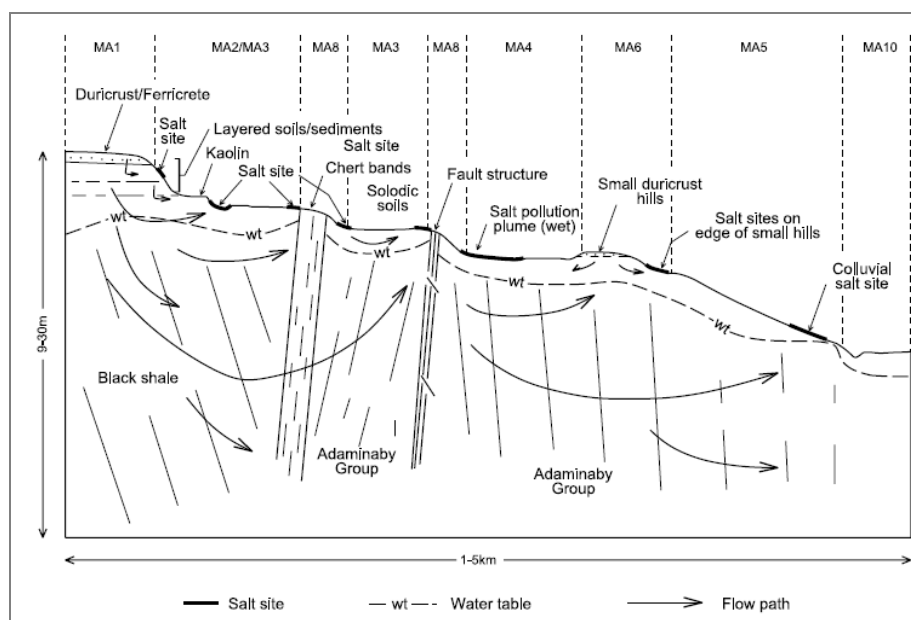
By not only denoting the functions for each HGL, but also ranking them for Salinity Hazard (Figure 2), land managers can target the correct works to an area and are able to prioritise the landscapes most at risk. The

Very High Hazard HGLs, (Spa Road, Nadgigomar and Budjong Creek) all have demonstrated onsite and offsite salinity impacts. Every flow line examined in these HGLs exhibited salinity.

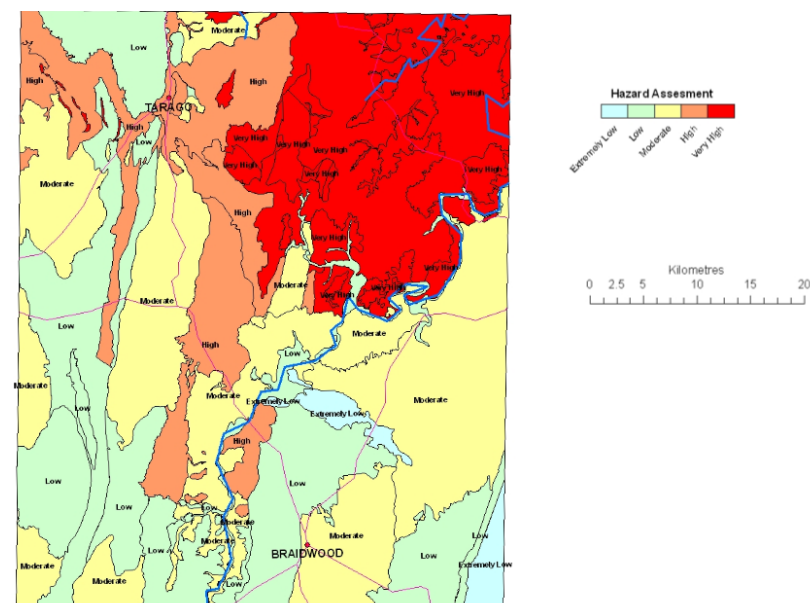
The functions, strategies and salinity hazard vary widely across the landscape (Table 4). Some HGLs such as Mongarlowe, Palarang and Butmaroo Range provide sources of fresh water and dilution flow. Strategies for these landscapes, for one or more management area, will include maintaining and maximising runoff. Actions include not planting excessive amounts of woody vegetation as this action will compromise the fresh water contribution from this HGL.

**Table 3. Management Action groups and an example management action from the Braidwood area.**

Management action group	Example Management Action
Vegetation for ecosystem service VE	VE2 - Interception planting of trees to target shallow groundwater
Vegetation for production VP	VP9 - Perennial horticulture to manage recharge
Farming systems FS	FS7 - Controlled Traffic farming systems
Engineering E	E3 - Diversion banks to avoid recharge on low areas
Irrigation systems IS	IS3 - Effluent disposal systems specific to site conditions
Soil Ameliorants SA	SA5 - Address soil biological health by application of compost
Saltland rehabilitation SR	SR6 - Water ponding on dry scalds



**Figure 1. Management Areas (MA) cross section for the Spa Road HGL, Braidwood 1:100,000 map sheet.**



**Figure 2. Braidwood 1:100 000 Salinity Hazard Map.**

HGLs such as Spa Road and Long Flat North generate salt loads which enter streams. In such cases strategies include reducing discrete landscape recharge, rehabilitation and management of discharge sites and the interception of shallow lateral flow and shallow groundwater. For these HGLs management actions for one or more management area will include maintaining and improving native pastures to manage recharge and the rehabilitation of salt land to minimise onsite and offsite degradation.

**Table 4. Function, strategy and hazard analysis.**

No	HGL	Function	Strategy	Hazard	Confidence
1	Mongarlowe	A, B,C	4,10	Low	High
2	Cookanulla	D,G	3,4,6	Medium	Medium
3	Spa Road	D,G,F,H,I	3,4,6,2,11	Very High	High
3	Moura Creek	D,G	4,6,2	Low	Medium
5	Budjong Creek	D,G,H,I	3,4,6,2,11	Very High	High
6	Palerang	A,B	4,10	Very Low	Low
7	Bobbaduck Hills	D,E,G,I	10,4,2,11	High	Medium
8	Long Flat North	D,F,G	4,2,6	High	Medium
9	Long Flat South	A,B	10,4	Medium	Low
11	Butmaroo Range	A,B,C	10	Low	High
12	Mulloon	A,B,D	1,2,4,6,7	Medium	Medium
13	Braidwood East	A,B,D,G	1,4,6,7,10	Medium	High
14	Braidwood West	A,B,G	1,2,4,7,10	Low	High
15	Hollow Wood	A,G	10,4	Low	Low
16	Merimbula-Minuma	A,B,	10	Very Low	Medium
17	Nadgigomar	D,G,H,I	1,2,3,4,6,11	Very High	Medium
19	Illogen Park	D,G	1,2,4,6,10	High	Low
20	Euradux	A	4,6,10	Very Low	Medium
22	Lake Bathurst	D,E,F,G	1,4,6	High	High
24	Larbert	A,B,G	1,2,3,4,6	Low	High

## Conclusion

The main HGL features are the identification of salinity processes relevant to each parcel of land, the specific management actions to be undertaken relevant to landscape function and the prioritisation of actions based on salinity hazard. Management actions to be avoided within a HGL and within management areas are highlighted. The lists of prescriptive management actions for each part of the landscape allows land resource managers to better target on ground rehabilitations and mitigation works.

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# Roots and earthworms under grass, clover and a grass-clover mixture

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## Abstract

White clover has a lower root biomass and a higher abundance of earthworms than grass. This might have an impact on the ecosystem services soil structure maintenance and water regulation when white clover is introduced in a grass-clover mixture. We investigated the root biomass, the abundance of earthworms and a selection of soil physical parameters in white clover, grass-clover, and grass with and without inorganic N fertilizer. The treatment with clover-only had a lower root biomass, a lower C/N-ratio of the roots, a higher abundance of earthworms, a higher number of earthworm burrows, a lower penetration resistance at the 20-30 cm soil layer and a lower proportion of crumbs in the soil, than the other treatments. This confirms the literature that pure clover stimulates the ecosystem services of water regulation, but is less conducive to soil structure maintenance. However, the grass-clover mixture did not differ significantly from the grass treatments, but differed from pure clover in a higher percentage of soil crumbs. We infer that, when clover is introduced in grassland to reduce the reliance on inorganic fertilizer, the mixture of grass and clover maintains the positive impact of grass roots on soil structure but only may show a positive effect of clover-only on water regulation with a higher clover percentage in the dry matter than in our experiment.

## Key Words

Roots, earthworms, grassland, clover, ecosystem services.

## Introduction

In sustainable grassland the focus is on ecosystem services like soil structure maintenance and water regulation, because of the perennial nature of the crop with no regular cultivation coupled with the compaction from animal trampling and tractor usage. For these ecosystem services, roots and soil biota play an important role. When sustainable grassland systems are developed it is important to know which effect management measures have on roots, soil biota and the functioning of the soil-plant system. One of the management measures that may contribute to sustainable grassland systems is the introduction of white clover (*Trifolium repens*) with its ability to fix atmospheric N<sub>2</sub> in symbiosis with *Rhizobium* bacteria. However, it is well documented that the root density of white clover is considerably lower than that of grass (Robinson and Jacques 1958; Young *et al.* 1958; Evans 1977; Tisdall and Oades 1979). Since the organic material released by living or decomposing roots stabilizes aggregates directly or indirectly by providing nutrients to micro-organisms, the lower root density could have a direct impact on soil structure maintenance. Robinson and Jacques (1958) measured a lower percentage of stable soil aggregates in white clover than in perennial ryegrass. On the other hand, Sears (1950) and Van Eekeren *et al.* (2005) found a higher earthworm biomass in a grass-clover mixture than in grass-only swards. Earthworms are known for their positive effect on soil structure and water regulation through their burrowing activity and earthworm burrows characteristics (Hoogerkamp *et al.* 1983; Clements *et al.* 1991). Mytton *et al.* (1993) found higher drainage rates in white clover than in perennial ryegrass. Altogether this would suggest that with the introduction of white clover in grassland, soil structure maintenance could deteriorate, while water regulation would improve.

In the present field study, we measured the root biomass, the abundance of earthworms and a selection of soil physical parameters in white clover-only, a grass-clover mixture, and grass-only with and without inorganic N fertilizer. Our objectives were (1) to measure the effect of white clover, perennial ryegrass and a grass-clover mixture on the root biomass and abundance of earthworms, and (2) to explore the relevance of changes for the ecosystem services soil structure maintenance and water regulation.

## Methods

### *Sampling site and experimental design*

The experiment was established in spring 2004 on a free-draining sandy loam soil (7.2-7.5 % clay (< 2 µm)) in the east of the Netherlands (52°26'N, 6°08'E). Four treatments were established in a completely

randomized block design of six blocks:

GN1 : Grass with inorganic N fertilizer;

GN0 : Grass without N fertilizer;

GCN0 : Grass-clover without N fertilizer;

CN0 : Clover without N fertilizer.

The former production pasture (mainly based on *Lolium perenne*) was killed in March 2004 with 3 L/ha Roundup<sup>®</sup> Max (Monsanto Company, St Louis, USA), after which the sward was ploughed and prepared for sowing. On 26 April, the different treatments were applied. The seed used was 35 kg *L. perenne* L./ha (cvs. Plenty and Roy) for the grass-only treatments (GN1 and GN0), 30 kg *L. perenne* L./ha and 5 kg *T. repens* L./ha (cv. Alice) for the treatment GCN0 and 10 kg *T. repens* L./ha for the treatment CN0. In order to get approximately the same quantity and quality (C/N ratio) in the above- and below-ground biomass in GN1 and GCN0, inorganic fertilizer (calcium ammonium nitrate 27%) was applied on GN1 at a rate of 150 kg N/ha. The percentage clover dry matter in 2005 was on average 26% for GCN0 and 75% for CN0.

### *Soil sampling and analysis*

On 16 December 2005, two growing seasons after the start of the experiment, soil samples were taken for determination of root biomass, earthworm biomass, earthworm and earthworm burrow abundance, and soil structure. Three soil cores (0-10 cm,  $\phi$  8.5 cm) per plot were taken to determine the root biomass. The soil in the samples was thoroughly rinsed with water, after which the roots were oven-dried at 70 °C and the dry matter of the roots was measured. After drying, the individual samples of roots were bulked together per treatment and analyzed for ash content and total N. Root biomass was expressed as grams of ash-free dry matter (AFDM). Earthworms were sampled in two blocks (20 x 20 x 20 cm) per plot. The blocks were transferred to the laboratory where the earthworms were hand-sorted, counted, weighed and fixed in alcohol prior to identification. Numbers and biomass were expressed per m<sup>2</sup>. Before the blocks were sorted for earthworms, in one block per plot the earthworm burrows with a diameter >2 mm were counted on horizontal surfaces (20x20 cm) exposed at 10 cm and 20 cm depth. Bulk density was measured in the 5-10 cm layer below the soil surface, in three undisturbed ring samples containing 100 cm<sup>3</sup> soil. Penetration resistance was measured with a penetrometer (Eijkelpamp, Giesbeek, The Netherlands) with a cone diameter of 2 cm<sup>2</sup> and a 60° apex angle. Cone resistance was recorded per cm of soil depth and expressed as an average value of 6 penetrations per plot in the soil layers of 0-10 cm, 10-20 cm, 20-30 cm. Soil structure was determined in 1 block (20 x 20 x 20 cm) per plot. The soil was divided by visual observation into crumbs, sub-angular blocky elements and angular blocky elements (FAO 2006). These were weighed and expressed as a percentage of total fresh soil weight.

### *Statistical analysis*

The effects of grass-only and fertilization, clover-only, and the mixture of grass-clover on the measured parameters were tested using one-way ANOV, using the GENSTAT statistical software (8<sup>th</sup> Edition, VSN International, Hemel Hempstead, UK)

## **Results**

CN0 had significantly lower grass root biomass and significantly higher clover root biomass than the other treatments (Table 1). The ranking of treatments in terms of total root biomass was comparable to that of grass root biomass. In terms of the total N in the root biomass, CN0 was significant lower than the other treatments. The C/N ratio in the total root biomass was lowest for CN0 and highest for GN0. GN1 and GCN0 were intermediate. Earthworm abundance was significantly higher in CN0 than in the other treatments (Table 1). CN0 had the highest earthworm biomass, GN1 and GN0 the lowest. Earthworm numbers and biomass were negatively correlated with the C/N ratio of the root biomass ( $r=-0.59$ ,  $P=0.002$  and  $r=-0.52$ ,  $P=0.01$ , respectively). The number of earthworm burrows at 10 cm depth was significantly higher in CN0 than in the other treatments. At 20 cm depth, the number of earthworm burrows was highest in the two treatments with clover (GCN0 and CN0), but it was not significant different from GN1. The number of burrows at 10 cm and 20 cm depth was positively correlated with the earthworm biomass ( $r=+0.50$ ,  $P=0.012$  and  $r=+0.49$ ,  $P=0.015$ , respectively).

Bulk density was not significantly different between the treatments. The penetration resistance in all soil layers was lower in clover-only (CN0) than in the grass-only with inorganic N fertilizer (GN1), but this was only statistically significant in the soil layer at 20-30 cm depth. The penetration resistance at 20-30 cm was negatively correlated with earthworm biomass ( $r=-0.47$ ,  $P=0.02$ ). The proportion of crumbs was significantly

higher in GN0 than CN0 (Table 1). GN1 and GCN0 took an intermediate position. The CN0 had the highest proportion of angular blocky elements. The proportion of crumbs was negatively correlated with clover root biomass ( $r=-0.53$ ,  $P=0.008$ ), but no significant correlation was present with grass or total root biomass.

**Table 1. Root biomass, earthworm abundance, earthworm burrow number and soil structure in grass with added inorganic N fertilizer (GN1), grass without N fertilizer (GN0), grass-clover without N fertilizer (GCN0) and clover without N fertilizer (CN0).**

Parameter	Unit	Treatments				P-value
		GN1	GN0	GCN0	CN0	
<i>Roots biomass 0-10 cm</i>						
Grass	g AFDM/m <sup>2</sup>	169a	217a	177a	12b	<0.001
Clover	g AFDM/m <sup>2</sup>	0c	1c	16b	62a	<0.001
Total	g AFDM/m <sup>2</sup>	169a	218a	193a	73b	<0.001
Total N	g N/m <sup>2</sup>	4.0a	4.1a	4.5a	2.6b	0.043
C/N		21.0b	26.3a	21.3b	14.2c	<0.001
<i>Earthworms</i>						
Total number	n/m <sup>2</sup>	322b	326b	359b	480a	0.002
Total biomass	g/m <sup>2</sup>	82b	76b	110ab	135a	0.009
<i>Earthworm burrows</i>						
10 cm depth	n/m <sup>2</sup>	58b	67b	138b	225a	0.002
20 cm depth	n/m <sup>2</sup>	50ab	8b	113a	121a	0.023
Bulk density	g/cm <sup>3</sup>	1.47	1.42	1.49	1.47	0.098
<i>Penetration resistance</i>						
0-10 cm	mPa	1.48	1.44	1.46	1.39	0.776
10-20 cm	mPa	1.46	1.45	1.40	1.34	0.368
20-30 cm	mPa	2.51a	2.39ab	2.45ab	2.13b	0.036
<i>Soil structure 0-10 cm</i>						
Crumb	%	39bc	53a	50ab	32c	0.006
Sub-angular	%	13	9	12	5	0.094
Angular	%	47b	38b	38b	62a	0.009

Values followed by the same letter within a row are not statistically different at the 5% error level for the main treatment effect.

## Discussion

In line with other research (Robinson and Jacques, 1958; Young *et al.* 1958; Evans, 1977; Tisdall and Oades, 1979), the root biomass in clover-only (75% clover in the dry matter in 2005) was less than in grass-only. However, the mixture of grass and clover (26% clover in the dry matter in 2005) had the same root biomass as grass-only. Although the soil structure (measured as proportion of crumbs) was only correlated with clover root biomass and not with grass or total root biomass, the soil structure followed the same pattern; the soil structure in clover-only (measured as a proportion of crumbs) was less developed than in grass-only and the grass-clover mixture. This is in line with other research (Robinson and Jacques, 1958; Tisdall and Oades, 1979) in which perennial ryegrass had a higher soil aggregate stability than white clover-only. Since the grass root mass and the soil structure in the grass-clover mixture were comparable with the grass-only treatments, we suggest that the soil structure of clover mixed with grass is maintained at the same level. Further research on soil aggregate stability is needed for confirmation.

The earthworm biomass was higher (70%) in clover-only (CN0) than in grass-only (GN1 and GN0), with the mixture of grass and clover in an intermediate position. Sears (1950) and Van Eekeren *et al.* (2005) found a higher earthworm biomass in a grass-clover mixture than in grass-only swards. Thus, introduction of clover in a grass sward results in higher earthworm population densities. The negative relationship between the C/N-ratio of the root biomass and the total abundance of earthworms, suggests that the quality of the litter rather than the quantity played a prominent role in the higher abundance of earthworms. Water regulation as an ecosystem service in grasslands is greatly influenced by earthworms (Clements *et al.* 1991; Bouché and Al-Addan 1997). Especially earthworm burrows can increase water infiltration (Edwards and Shipitalo 1998). In our experiment, the numbers of earthworm burrows at 10 and 20 cm depth were highest in clover-only. Furthermore, clover-only showed the lowest penetration resistance at 20-30 cm, suggesting improved water infiltration. These data are consistent with results of Mytton *et al.* (1993), who found that white clover-only drained more rapidly than grass-only. In their research, soil moisture curves indicated a more free-draining structure in clover than in grass due to a higher ratio of macro- to micro-pores (Mytton *et al.* 1993).

For both drainage and soil moisture characteristics, Mytton *et al.* (1993) found that a grass-clover mixture (> 50% clover in the DM) took an intermediate position between the monocultures of grass and clover. In our research, the mixture of grass-clover (GCN0), with 26% clover in the DM, showed a higher number of earthworm burrows and a lower penetration resistance than grass-only with fertilization (GN1), but differences were not significant. This suggests that a positive effect of clover on water infiltration was not apparent in our grass-clover mixture. With a higher clover percentage in the dry matter this might be different.

## Conclusions

The treatment with clover-only had a lower root biomass, a lower C/N-ratio of the roots, a higher abundance of earthworms, a higher number of earthworm burrows, a lower penetration resistance at the 20-30 cm soil layer and a lower proportion of crumbs in the soil, than the other treatments. This confirms the literature that pure clover stimulates the ecosystem services of water regulation, but is less conducive to soil structure maintenance. However, the grass-clover mixture did not differ significantly from the grass treatments, but differed from pure clover in a higher percentage of soil crumbs. We infer that, when clover is introduced in grassland to reduce the reliance on inorganic fertilizer, the mixture of grass and clover maintains the positive impact of grass roots on soil structure but only may show a positive effect of clover-only on water regulation with a higher clover percentage in the dry matter than in our experiment.

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# Soil protection and strategic goals in local environmental planning

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## Abstract

Strategical Environmental Planning is a newly developed interdisciplinary planning tool for sustainable urban development. A regular environmental reporting is to document the long-term developments in the quality of the local environment. Purpose is to develop an instrument with specific relation to space and time so that environment-related data processing, planning and control can be simplified. Besides the interests of soil protection, the environmental goods species/habitats, water, climate and human health are also considered.

## Key Words

Environmental quality goals, monitoring, soil protection, Strategical Environmental Planning, surface claim, sustainable urban development.

## Introduction

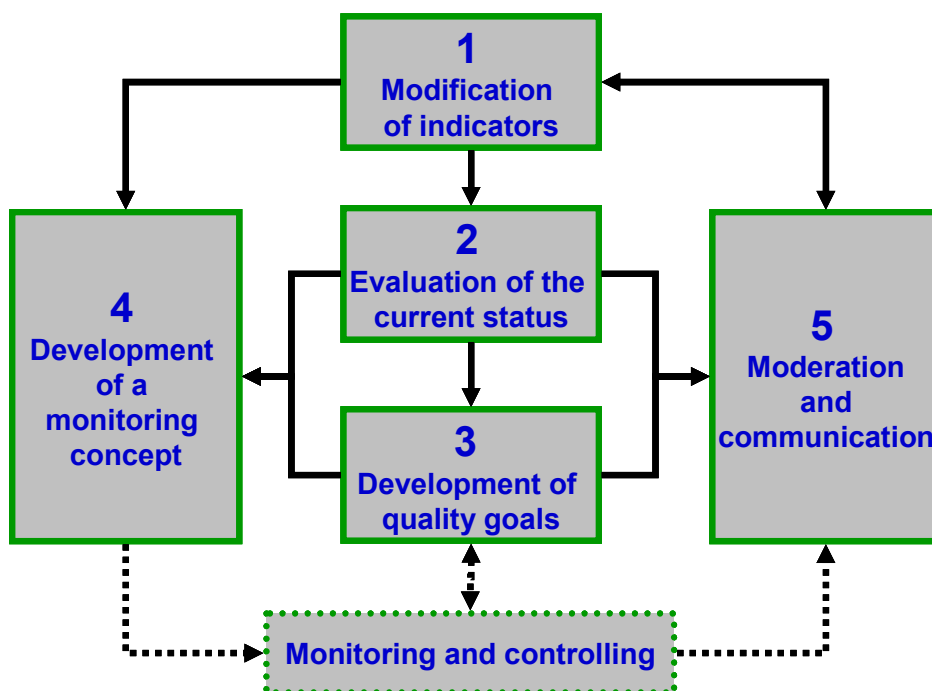
Daily nearly 113 hectares of surface are newly built up in Germany – for residential, industrial and recreation areas and traffic. Every German claims 564 square metres of space – trend rising according to statistics. On the other hand, about 180,000 hectares of derelict land are potentially available in urban areas. In the Ruhr Area - an urban area with a long history of coal mining and steel production in Germany - these are often more or less contaminated industrial brown fields. Through high space consumption and intensive land utilisation areas with ecosystem service functions are increasingly lost. Therefore, the city of Bochum has initiated the development of a new tool for the comprehensive documentation and evaluation of ecosystem services for the city area. This will allow to identify priority areas where measures to improve the environmental quality will render the highest returns in terms of ecosystem services, with special respect to soil protection.

## Methods

Five steps of the Strategical Environmental Planning form the basis for monitoring and controlling the environmental indicators (Figure 1). During the first step, the available data was collected and evaluated in terms of its suitability to serve as an indicator for the status of the single environmental goods (soil, flora/fauna, surface/ground water, air/climate, human health). One important criteria for all data sets was that they need to be available with geographically explicit references so they can be documented and updated regularly with GIS. At the same times, specific quality goals were developed for each environmental good. In the second step, the current status of each environmental good was compared to the goals and deficiencies identified and documented in the GIS. Based on the produced maps, priority areas for action were identified in the next step. These priority areas were selected in such a way, that single measures (like unsealing or restoring vegetation) would positively affect as many environmentally goods as possible. Finally, a monitoring concept was developed that will ensure that all relevant parameters are evaluated at regular intervals at a specified spatial resolution. The whole process was conducted in close cooperation with representatives from all relevant administrative bodies of the local city council, with whom regular meetings were held and who participated in the identification of relevant parameters and their evaluation. In this way it was ensured that the new planning tool is known and accepted by the regulators.

## Results

For soil quality, the presence and remediation status of contaminated sites and the ability of soils to fulfil its natural functionality were identified as the relevant criteria. While data on contaminated sites and on surface sealing was available and only needed to be transformed into a GIS-compatible format, data on soil functionalities was missing. For the rural parts of the city (about 25% of the city area), agricultural soil maps on a scale of 1:5000 are available and the data was transformed into formats that allowed the calculation of relevant soil functionalities such as water retention capacity, pollutant filtering capacity or agricultural production potential. For the other parts of the city, soil maps are not available and soil functionality was estimated by deriving soil properties from numerous sources in which the degree of anthropogenic influence



**Figure 1. The five steps of the Strategic Environmental Planning**

was estimated (historical maps, age of housing/industry, degree of destruction during WW II, aerial photographs). Data for most other environmental goods was also available and incorporated into the GIS. A concept of "environmental corridors" was developed, that stretch through the whole city and are not restricted to areas that are currently not built up. These corridors are to serve as priority action areas, where measures aiming at improving the environmental quality should be focused. Such measures may include upgrading the environmental functionality of derelict industrial sites by removing surface seals or by supporting revegetation. In built-up areas within the corridors, new constructions will be obliged to minimize surface sealing and include green roofs to contribute to soil functionality, local climate and habitat connectivity without profoundly interfering with current city planning. The results of the inventories and their evaluation are documented and updated in a GIS that is available to all administrative bodies of the city council. Part of the results will also be made available to the public through a web-GIS on the city's homepage.

### **Conclusion**

By providing a GIS-based data set for the spatially resolved documentation and evaluation of the status of environmental goods the importance of soil protection has been established securely within the overall evaluation of ecosystem services for the city of Bochum. By providing an atlas that not only documents the status but also the long-term goals for the environmental goods, a tool has been developed that is a base for pro-active planning of measures to improve environmental quality as opposed to the current policy, where environmental issues are only evaluated through EIAs in reaction to proposed building projects

# What are the opportunities for enhancing ecosystem services from soils through management of soil carbon?

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## Abstract

There is a worldwide need for more balanced decision-making in planning the use and management of our soil resources. Such planning needs to fully consider the value of all ecosystem goods and services supported by soil. However, within the context of economic evaluation, soils have been a poor relation when compared to other natural resources such as biodiversity and water. As a consequence there is a dearth of information on the cost/benefit implications of changing soil management to enhance the delivery of ecosystem services. Here we discuss the opportunities to enhance the delivery of ecosystem services from soils through managing soil carbon using case-studies from Scotland. We examine soils inherent capacity to store carbon in agricultural and semi-natural habitats and reflect on the limitations on achieving this biophysical potential. We then contrast these with the potential costs, ancillary benefits of, and uncertainties in, management options. Finally we discuss potential trade-offs and implications of climate change in the development and achievement of land use policies which involve soil carbon sequestration.

## Key Words

Carbon capacity, management, valuation, ecosystem services.

## Introduction

The emergence of sustainable development on the policy agenda and a broader view of the importance of soils to fulfil societal needs has brought with it expectations that soil scientists can not only deliver the right sort of information to inform a broad range of land user and policy requirements for soil use and management but that we can also contribute to the wider debate on future options for a sustainable planet. Soil is truly the ultimate model for multi-functionality since there is an expectation that soil resource can be managed to help mitigate climate change, to expand urbanisation, to increase agricultural production to address food insecurities, to provide recreational services, to maintain reliable water supplies, all while maintaining soil quality and protecting our natural environment. This is a tall order and meeting this challenge will require a sound appreciation of the values that people place on the ecosystem goods and services they need and want. At a national level, many countries, the UK included, are championing an ecosystem service approach “to secure a diverse, healthy and resilient natural environment, which provides the basis for everyone’s well-being, health and prosperity now and in the future”. Assessing ecosystem service values goes beyond the bounds of soil science and requires an effective interaction between natural and social science disciplines. In Scotland, we have been developing interdisciplinary research to tackle valuation of natural resources, including soils, as part of the Scottish Government’s research programme on Environment, Rural Land Use and Stewardship. In this paper, we identify the opportunities for soil science to harmonise with the developments of the ecosystem services approach, in particular the interface with social and economic disciplines to support more holistic valuations of our fundamentally non-renewable soils resource.

Soil organic matter (SOM) content is a fundamental property of soil because it determines the soil's capacity to deliver many of its functions, including storing, retaining and transforming water, nutrients and contaminants as well as sustaining biodiversity and carbon sequestration and providing nutrients for biomass production. Thus, loss or increase of soil organic matter could have multiple and diverse environmental, social and economic consequences. Although most soils are managed specifically to optimise the delivery of one or two functions, management to achieve these goals in turn may compromise soils ability to perform the other functions. In our research we developing and applying approaches to characterise and quantify trade-offs in managing soil organic matter, and specifically carbon, to meet specific policy objectives.

## Our approach

### *Capacity for soil carbon sequestration*

Scottish soils are estimated to contain approximately  $3 \times 10^9$  tonnes carbon, which is the majority of the soil carbon stock of the entire UK. This stock is associated with a wide diversity of soil types reflecting climate and topography, which also accounts for the wide range of functions associated with Scottish soils (Scottish Government 2009). We explore the capacity to increase soil C in across this range of soil types using data from the National Soil Inventory for Scotland (NSIS). This inherent biophysical capacity to sequester soil carbon is reviewed against current soil C stocks under different land uses and the likely reasons of these differences, including land use change, management and pollution.

### *Cost/benefits in soil C management options*

Using two contrasting case-studies, we explore the opportunities to enhance soil carbon sequestration in agricultural and semi-natural habitats and the scientific uncertainties around the success of management options. We report the results of a choice experiment study from 600 households to investigate the costs and benefits of a policy-driven management programme to enhance soil carbon sequestration in Scotland.

### *Potential trade-offs and implications of climate change in soil carbon sequestration*

We investigate the role of ancillary or co-effects on economically driven decision-making for soil C management in the land use and environmental sector. Finally opportunities for, and costs/benefits of, soil C sequestration are considered in the context of future climate change by reviewing potential impacts on the biophysical capacity for Scottish soils to store carbon (c.f. Brown *et al.* 2008; Figure 1); the uncertainties associated with soil C management options and potential trade-offs between different land use policy goals to enhance, protect and restore C sequestration.



**Figure 1. Location of prime agricultural land (LCA classes 1, 2 and 3.1) a) current b) predicted under 2050's UKCIP02 Med-High Emissions (Macaulay Institute, work in progress; Scottish Soil Framework, 2009)**

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