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A baseline for soil carbon monitoring in Amazonia

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Abstract
Amazonian forests are important reservoirs of carbon both below and above ground. The status of such C reservoirs is changing, forests are becoming more productive and the net gain in vegetation structures is likely to enter the soil in short term. Nevertheless, the Amazon forest is predicted to suffer from periodic droughts in the near future. This has the potential to convert current C sinks to sources. An Amazonian baseline for detecting soil C changes is now being implemented to allow temporal monitoring of soils. Here we evaluate the feasibility of such an endeavour based on soil C variability in 67 one-hectare plots across Amazonia. We estimated the minimum detectible change (MDC) of background C after a standardized sampling effort as a selection tool for site inclusion into the baseline. Most sites allow precise monitoring of soil C changes with a relatively small number of samples. We estimate that only a 20% change in current background soil C concentrations are needed to allow detection of soil C changes in most of our sites. At an increasing soil C stock of 0.33 Mg C ha^{-1} yr^{-1} in Amazonia, this should allow accurate appraisal of soil C changes on decadal timescales.

Introduction
In the past few decades, Amazonian forests have been observed to be experiencing increasing rates of forest productivity and biomass turnover (Phillips *et al.* 1998; 2004; 2009; Baker *et al.* 2004). Such increment in forest productivity translates into increasing above ground forest biomass at an average rate of 1.22 ± 0.43 Mg/ha/yr (Baker *et al.* 2004). Much of this net increase in vegetation biomass is likely to enter the soils as necromass, thus potentially increasing soil carbon (C) stocks. Nevertheless, Amazonian forests are also subject of a changing environment and likely to become hotter and drier in the near future. Most climate change scenarios predict that severe droughts may become more frequent in the 21st century, particularly for southern Amazonia (i.e. Cox *et al.* 2008), with potential to revert both the net sink for atmosphere CO_2 and the capacity of soils to store C (Phillips *et al.* 2009). Although, it is possible that drought-associated increments in solar radiation could be associated with increasing rates of tropical forest productivity (Huete *et al.* 2006; Saleska *et al.* 2007), a recent study by Phillips *et al.* (2009) showed that severe droughts can reverse the last decades trend to increase biomass, converting forested Amazonia to a net source of C to the atmosphere. Therefore soil C storage is likely to be currently changing in Amazonia, and further dramatic changes in response to climate change might be expected in the near future. As changes in soil C stocks are dynamic and expected to occur at different rates across Amazonia, a network of permanent plots for soil C studies we perceived as needed to act as a definitive baseline for future comparisons. The RAINFOR network, with more than 130 permanent forest plots scattered across Amazonia, provides the needed infrastructure for such a baseline soil carbon network. These plots are inventoried each 4-5 years for above ground biomass and vegetation dynamics (Malhi *et al.* 2002), as well as being inventoried for soil chemical and physical properties (Quesada *et al.* 2009). In its next phase, the RAINFOR project is implementing the first basin wide, Pan Amazonian baseline for soil C monitoring. This paper outlines our goals and preliminary results acquired to date, evaluating the importance of local soil variability in influencing our ability to detect soil C changes in Amazonia.

Method
Study sites
The RAINFOR network has now more than 130 one hectare forest plots scattered across Amazonia. These sites include only pristine forests encompassing a large variety of vegetation forms and soil types common to Amazonia. In this paper we included soil C stocks for 67 sites, in six different South American countries, while another 38 are being processed at the laboratory. Our final goal is to perform soil C inventories in 130 sites, from which 60 will be sampled at high intensity to form a baseline for soil C in Amazonia.

Soil sampling
Two different sampling strategies are being used. At every site a relatively low intensity sampling is carried out for inventory purposes, usually collecting 5 soil cores plus one soil pit per hectare. At each sampling point, soil is retained in the following depths: 0-0.05, 0.05-0.10, 0.10-0.20, 0.20-0.30, 0.30-0.50, 0.50-1.00, 1.00-1.50 and 1.50-2.00 m using an undisturbed soil sampler (Eijkelkamp Agrisearch Equipment BV, Giesbeek, The Netherlands). For soil pits down to 2.0 m depth, soil and three bulk density samples are collected per sampling depth (as for the soil cores).
Once these samples had been analysed for soil C concentration, a subset of 60 study sites was selected to integrate the soil C monitoring baseline. This selection is based primarily on soil C variability, geographical distribution and logistical aspects of each site such as accessibility, site facilities, sampling costs, and level of protection for these areas.
For the baseline, a fixed number of 50 soil cores are being sampled per hectare plot, at the same depth interval as per soil cores (0 – 2.0 m), using a mechanised auger system.
As each forest plot shows specific conditions, the sampling strategy varies from one plot to another. Sites with a higher degree of spatial variability are sampled using random stratified sampling techniques while forests having high degree of homogeneity are sampled using systematic random sampling. No matter the sampling design, sample points are recorded in an X and Y diagram to allow future measurements to be made as closer as possible to the same sampling point used to estimate the baseline.

Soil Analysis
All samples are air dried and have roots, detritus, small rocks and particles over 2 mm removed. Soils are then milled to less than 50 µm and have their moisture correction factor and rock volume determined. Samples are then analysed using an automated elemental analyser (Pella 1990) model Vario Max CN (Elementar Instruments, Germany).
All analysis included in this study were made with 0 - 30 cm data only which is the soil layer more susceptible for changes. However, Fig. 1 shows soil C stocks to the depth of 2 m, to provide better representation of spatial variability of soil C stocks in Amazonia.

Results and discussion
Figure 1 show soil C stocks to 2 m depth derived from field data collected by the RAINFOR project (Quesada *et al.* 2009). Based on such soil C inventories, we calculated the minimum soil C change needed to allow detection with 0.95 power (2-sample t test, based in a sampling effort of 50 cores per ha, Figure 2a). Calculations were also done at power = 0.99 and usually differences between power 0.99 and 0.95 were below 1%, the exception being the sites with minimum detectible change (MDC) > 0.40 at power = 0.95. Differences in these specific sites ranged from 13 to 41%.

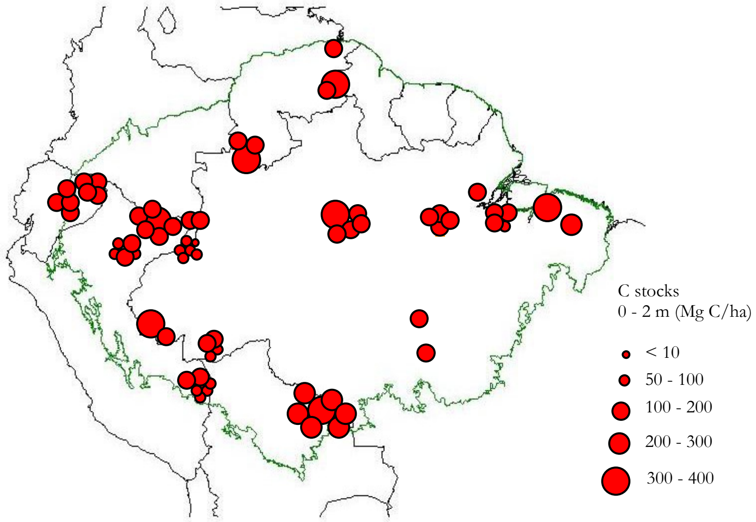

Figure 1. Soil C stocks for 67 sites across Amazonia

Most of our sites had low coefficient of variation for soil C which resulted in a generally low minimum detectible change at power = 0.95. About 45% of our sites had MDC below 0.25 of background C, and 67% were below 0.40.

As there is an abundance of sites with low soil C variability in Amazonia, sites above 0.40 MDC, obtained with 50 cores per plot, can be discarded from baseline studies as there would be little gain in increase the number of cores per plot. For example, doubling the sampling effort to 100 cores would change MDC of 0.45, 0.59 and 0.71 (at 0.95 power, 50 cores per plot) to 0.32, 0.41 and 0.50 respectively - still above the median MDC for all sites in this study (all sites median = 0.28).
We considered that MDC of 0.40 (at 0.95 power) would be the top limit for site inclusion into the baseline. This minimum change is low compared to results from Zhou *et al.* (2006), which reported an approximate 64% increase in background soil C in a 24 years interval in old-growth forests in China. Soil C concentrations changed from 14 mg g^{-1} to 24 mg g^{-1} during this interval at an average rate of increase *ca.* 0.4 mg g^{-1} yr^{-1}.

As for Amazonia, increasing rates of tree mortality and net biomass gain suggest that current soil C stocks should be now increasing. In addition to increments in net biomass gain, carbon inputs to the soil through higher rates of tree mortality should be increasing at about 0.03 Mg C ha^{-1} yr^{-1} (Phillips *et al.* 2004), and assuming this to be a more or less ongoing increase in soil C inputs (dM_i/dt) then the actual rate of increase is the soil carbon pool should be $\tau \circ dM_i/dt$ with a reasonable value for tropical forest soil τ as 10 years (Lloyd 1999). The rate of increase for soil C should thus be of order 0.3 Mg C ha^{-1} yr^{-1}.

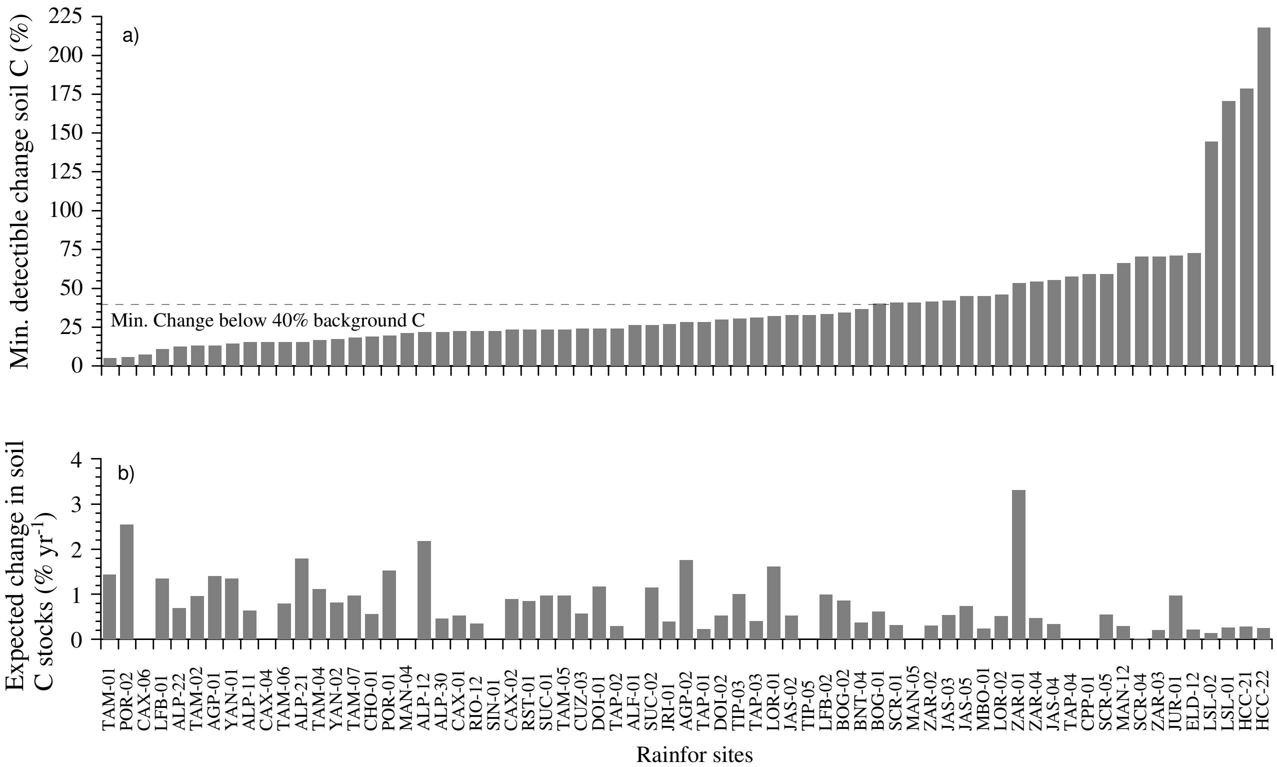

Figure 2. a) Minimum detectible changes in soil C at 0.95 power, based on 2-sample t test, 50 soil cores per hectare. b) Expected rate of change in soil C stocks, for each site in this study. Missing sites in 2b are due to lack of tree mortality rates to date.

The relative increase for soil C for our sites in Amazonia as a result of a stimulation of inputs from above increasing rates of above ground mortality was therefore calculated to be 0.008 of current soil C stocks per year (ranging from 0.0002 to 0.03 of background C stocks, Figure 2b), this disregarding any organic matter input from other vegetation structures except tree boles. Although organic matter input estimations made only with boles involves a time lag for necromass decomposition (which would probably decrease the average rate suggested here in the short term), our calculation ignores the production of lighter vegetation structures such as leaves and twigs which may also be having their productivity and turnover increased. This suggests that higher changes in soil C are likely to occur, possibly close to the values reported by Zhou *et al.* (2006), and imply in the possibility of detecting changes in background soil C in about 20 – 30 years with anticipated increases over 30 years averaging around 0.25 our estimate of current soil stocks . Also, not all sites with MDC 0.4 will be included in the baseline as there are other constraints for such intensive field sampling. A total of 29 sites from this study were selected to form the baseline, and other plots are being currently analysed. It resulted in an even lower MDC level for the baseline, averaging only 0.20 MDC at 0.95 power (0.23 at power 0.99).

Conclusion
Despite difficulties in working on such large and logistically difficult area, Amazonia stands out as an adequate site for soil C monitoring. Most sites allow precise monitoring of soil C changes with a relatively short number of samples. Estimated changes in soil C for the future and low soil C variability are likely to make monitoring of changes feasible in relatively short term (20 or 30 years).

Acknowledgments
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A framework for European soil monitoring

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Abstract
The ENVASSO project has developed a framework for monitoring European soils. 27 indicators were selected for erosion, organic matter decline, contamination, compaction, salinisation, decline in biodiversity, soil sealing, landslides and desertification. A monitoring network with a density of 1 site per 300 km^2 covers most soil type and land use combinations. 20 indicators were qualified for implementation, covering soil erosion by water, decline in soil organic matter, soil contamination, soil sealing, compaction, salinisation and desertification. Methods for monitoring wind erosion, tillage erosion and carbon stocks in peat soils were found to be inadequate. A tiered approach to implementation of soil monitoring is recommended.

Key Words
Monitoring, Indicators.

Introduction
Serious threats to soil throughout Europe have been identified (European Commission 2002, 2006). However, more evidence is needed to support stronger soil protection policy and to target and monitor its implementation. This requires a European soil monitoring network. Some national networks exist (Jones *et al.* 2005) but many have not been re-sampled. The Environmental Assessment of Soil for Monitoring (ENVASSO) project (Kibblewhite *et al.* 2008) aimed to define a European soil monitoring system and describe its potential implementation.

Methods
Indicator selection (Huber *et al.* 2008) was based on international recommendations (OECD 2003). 188 issues and 290 indicators were evaluated. The principal criteria were relevance, methodological soundness, measurability and policy relevance. Priority indicators were chosen (one for each of three key issues per threat) from an initial selection of 27 key issues and 60 indicators. Where possible, performance criteria (e.g. minimum detectable change, background values and indicator thresholds) were defined (Arrouays *et al*. 2008). Methods for within-plot sampling and for parameter measurement were documented (Jones *et al*. 2008). Geo-referenced information was collated on existing soil monitoring networks and inventories that could be re-sampled (Arrouays *et al.* 2008). The number of additional sites needed to adequately monitor soil type and land use combinations was estimated (Morvan *et al*. 2008). Data management requirements were defined (Baritz *et al.* 2008) and a prototype database system (SoDa) constructed with web-based inter-operability. Procedures for estimating 22 indicators were tested in 28 pilot studies (Micheli *et al*. 2008) covering representative regions and land uses.

Results
Of the 27 priority indicators, 20 were qualified (Table 1) but measurement methods are not available or were considered inadequate for 7 others. Better methods are required for continental scale estimation of wind and tillage erosion. Estimation of peat stocks requires reliable methods for estimating the distribution of peat depths and to account for variability in peat composition. A minimum spatial density of 1 site per 300 km^2 is recommended (Morvan *et al.* 2008), which approximates to a regular grid with sampling sites set at nodes 16 to 17 km apart. This is representative of most combinations of soil types and land uses. The site area should be (Arrouays *et al*. 2008) between 100 m^2 and 1 ha with homogeneous soil profile development. A minimum of 4 and preferably between 10 and 100 sub-samples fixed depth sub-samples should be taken, depending on the site area and soil profile variation.

Pedogenic horizon sampling adjacent to the sample area usefully supports site characterization. The time interval between sampling events needs to be long enough to allow for changes that can be detected within measurement errors. Analysis (Arrouays *et al.* 2008; Morvan *et al.* 2008; Saby *et al.* 2008) indicates that a minimum interval of 10 years is required.

Table 1. Threats, issues and indicators

Threat / Issue	Indicator
Qualified ENVASSO indicators	
Soil erosion	
Water erosion	Estimated soil loss by rill, inter-rill, and sheet erosion
Decline in soil organic matter	
Soil organic matter status	Topsoil organic carbon content (measured)
	Soil organic carbon stocks (measured)
Soil contamination	
Diffuse contamination	Heavy metal contents in soils
	Critical load exceedance by S and N
Local soil contamination	Progress in management of contaminated sites
Soil sealing	
Soil sealing	Sealed area
Land consumption	Land take [to urban and infrastructural development]
Brownfield re-development	New settlement area established on previously developed land
Soil compaction	
Compaction, structural degradation	Density
	Air-filled pore volume at specified suction
Causes of compaction	Vulnerability to compaction
Decline in soil biodiversity	
Species diversity	Earthworm diversity and biomass
	Collembola diversity
Soil microbial respiration	Microbial respiration
Soil salinisation	Salt profile
	Exchangeable sodium percentage
	Potential salt sources
Desertification	
	Land area at risk of desertification
	Land are burnt by wildfire
Non-qualified ENVASSO indicators	
Soil erosion	
Wind erosion	Estimated soil loss by wind erosion
Tillage erosion	Estimated soil loss by tillage erosion
Decline in soil organic matter	Peat stock
Landslides	Occurrence of landslide activity
	Volume/mass of displaced material
	Landslide hazard assessment
Desertification	Soil organic carbon content in desertified land

Discussion

Two types of indicators are implemented more easily. Group A are those for which there are existing networks i.e. *topsoil organic carbon contents, heavy metal contents in soils, critical load exceedance by sulphur and nitrogen, salt profile, exchangeable sodium percentage and potential salt sources*. Group B rely on existing remote-sensed data and / or spatial information i.e. *estimated soil loss by rill, inter-rill and sheet erosion, sealed area, land take, vulnerability to compaction, land area at risk of desertification* and *land area burnt by wild fire*. These groups provide a basis for monitoring soil erosion, decline in soil organic matter, soil contamination, soil sealing, compaction, salinisation and desertification. Some indicators require inventories that do not exist in all regions or lack harmonization i.e. *progress in the management of contaminated land, new settlement area established on previously developed land* and *occurrence of landslide activity*. Other indicators are compromised by uncertainties in relating site measurements to estimates over space and time, these are *soil density, air capacity, earthworm diversity, Collembola diversity* and *soil microbial respiration.*

A two-tiered approach to implementation is recommended. The first tier should establish a network with a site density of 1 per 300 km^2, for estimation of indicators in groups A and B. The second tier should be a sub-set of first tier sites with more extended and intensive monitoring, addressing requirements that cannot be implemented feasibly at the first tier, or for which fewer sites are needed. Examples are (1) when measurement procedures are too demanding for general implementation (e.g. some biological, gaseous flux and physical measurements, including measurement of soil erosion), (2) where intensive sampling is needed to describe soil processes to interpret indicator levels and trends (e.g. detailed assessment of sub-soil and lower horizons, or connectivity to landscape processes such as catchment inputs and outputs), (3) special investigations of error sources (e.g. intensive collection and testing of sub-samples to determine an optimum number for application in the first tier), or (4) when performing proficiency exercises to assess variability associated with different field teams (e.g. estimates of stone contents and texture). The second tier network could also provide reference sites for soil typological units.

Conclusions

There is a sufficient density of existing sites for continental soil monitoring over much of the European Union and the number of new sites required is relatively limited and confined to a few member states. There is an adequate technical basis to implement a successful monitoring system for a majority of the threats to soil resources. Current methods are inadequate for assessment of carbon stocks in peat soils, wind erosion and tillage erosion. The ENVASSO manual of procedures and protocols (Jones *et al*. 2008) is a valuable reference for future soil monitoring.

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A soil spatial prediction functions (SSPFs) for soil organic carbon in Eastern Australia

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Abstract
The development of spatial soil prediction functions (SSPFs) for soil organic carbon (SOC) in Australian bioregions has the potential to inform pragmatic measurement and monitoring schemes for the terrestrial carbon system. Recent availability of Australia-wide fine resolution datasets of key environmental covariates has created an unparalleled opportunity for the creation of bioregional SSPFs to guide sampling design for farm scale studies that may assist in economical verification efforts for SOC.

Key Words
Soil organic carbon, carbon auditing, spatial soil prediction function, terrain analysis, carbon offsets

Introduction
The predicted consequences of greenhouse gas derived climate change have increasingly turned recent attention to the potential early and positive role of terrestrial carbon sequestration. Whilst great promise is touted and indeed warranted, the challenges of creating verification systems that are at once pragmatic as well as capable of defining a fungible commodity for a given length of time in a dynamic environmental system are manifold. For example, the common truism of high spatial variability in soil organic carbon being a barrier to verification efforts indeed reflects very real soil variation that covers roughly nine orders of magnitude (i.e. see (Crawford *et al.* 2005). It does not follow however, that we lack scientific knowledge of this variation. As a result, the fundamental question of *'where to cut?'* such commodity measures faces all verification and subsequent payment efforts for terrestrial carbon sequestration. In regards to soil organic carbon, the recent availability of global elevation models at fine resolution in combination with national scale gamma radiometric datasets offers an unparalleled opportunity for the development of soil spatial prediction functions (SSPFs) that may assist in more economical measurement and verification efforts. Such a system would provide bioregional or sub-bioregional trends in SOC that could guide sampling design and subsequently be amended by local/farm scale measurements – allowing creation and further refinement of localised SSPF's, creating the basis of a coordinated SOC database and opening opportunities for monitoring of measured local carbon stocks.

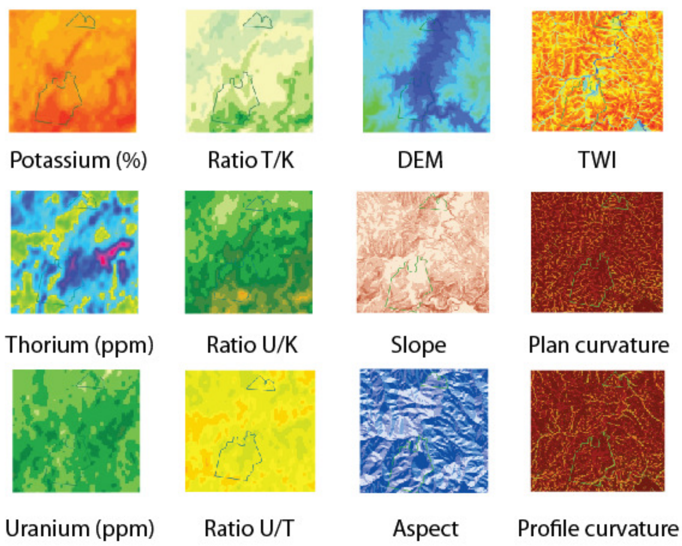

Figure 1. Example environmental attributes utilised in the SSPF for soil organic carbon.

Methods

Datasets utilised for this study include: the joint METI and NASA ASTER Global Digital Elevation Model V001 and its derived terrain attributes at 30 m resolution (Figure 1); the Radiometric Map of Australia at 100 m resolution (Figure 1) and direct observations of soil organic carbon from the Australian Soil Resource Information System (ASRIS) dataset (Figure 2). In addition, 250 m resolution climatic, landuse and SOC% to 30 cm depth as previously estimated (Henderson *et al.* 2005)(Figure 3) were also utilised. Prior farm scale data for SOC within the bioregion have been collated to refine the sub-bioregional SSPF linear trend to create localised and farm scale SOC maps as well as assist in sampling design for future surveys. Additional SOC surveys intended to broaden the variety of production systems, soil types and regional climates represented are in the process of design, collection and analysis and will be integrated into the relevant sub-bioregional SSPF in the near future. A novel approach to landuse classification in regards to local SOC spatial behaviour has also been developed and is currently being refined for farm sale application.

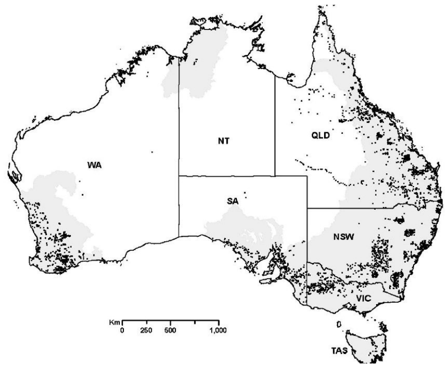

Figure 2. Relative location of topsoil SOC observations within the ASRIS dataset, grey areas indicate extent of prior maps generated from this dataset (Bui *et al.* 2009)

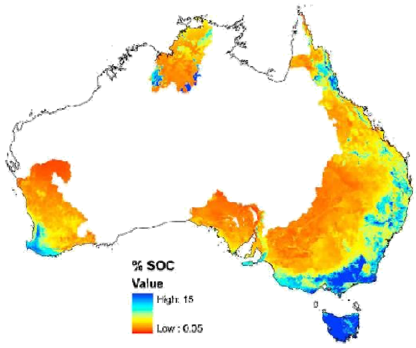

Figure 3. % soil organic carbon at 250 m resolution estimated by Henderson *et al.* 2005 using the ASRIS database

Results and discussion

Utilising the preliminary SOC results from collated farm scale surveys and the previously mentioned covariates a sub-bioregional SSPF was developed for SEH11 (Bathurst) (Figure 6) where *saga_wi* is a variant of the wetness index; *D_a* is dryland annuals; *D_input_pa* is dryland pasture with regular inputs; *D_noput_pa* is dryland pasture with no regular inputs; *D_noput_tree* is tree cover greater than 30% with no regular inputs and *U_th* is the ratio of uranium to potassium as derived from the radiometric dataset. This sub-bioregional SSPF was used to predict the SOC distribution at a 30 m resolution (as depicted in Figure 5) on a farm where no prior SOC survey had been conducted but necessary environmental covariates were available. Stratification of the predicted farm scale SOC variation (Figure 5) was then utilised as the basis of sampling design via stratified simple random sampling (i.e. see (de Gruijter *et al.* 2006) and analyses of the dataset are ongoing. A similar approach is being utilised for additional farm scale surveys within other bioregions.

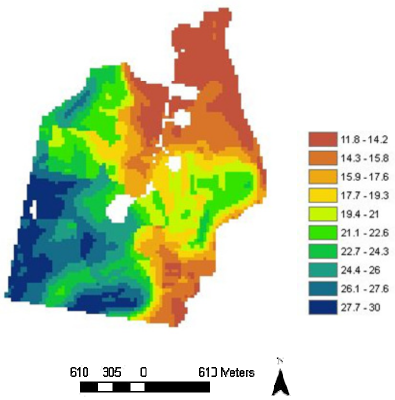

Figure 4. Predicted soil organic carbon ($kg/m^2/m^1$) using sub-bioregional SOC SSPF and local environmental covariates

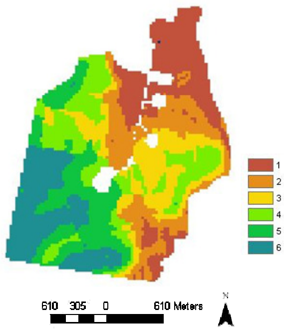

Figure 5. Stratification of SOC content for sampling design based on predicted SOC distribution (Figure 4)

1 – 6 August 2010, Brisbane, Australia. Published on DVD.

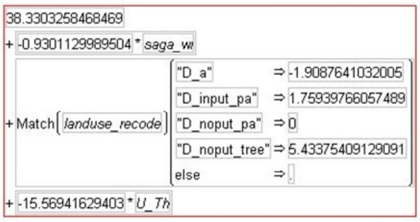

Figure 6. SSPF for SOC (see Figure 5)

Preliminary results indicate that at the local and farm scale controlling factors on soil organic carbon distribution appears to be dominated by landuse, water accumulation in the landscape and the uranium to thorium ratio. More specifically, landuse that involves a perennial component as opposed to annual dominated systems accumulate more SOC within the spatial model. The U:Th ratio also has a strong negative effect on SOC which is likely related to the age of the underlying regolith. Surprisingly, water accumulation is negatively associated with SOC occurrence in this instance. This runs somewhat contrary to the continental scale variation in SOC that appears to be controlled by soil moisture availability as outlined by (Bui *et al.* 2006).

Such fine and comprehensive spatial information of key environmental variables has the potential to support SOC specific SSPFs for sub-bioregional and bioregional scales enabling stratified sampling designs for initial farm scale SOC surveys where no prior soil carbon information exists.

Conclusion

The development of sub-bioregional and bioregional SSPFs for soil organic carbon has the potential to inform pragmatic measurement schemes for the terrestrial carbon system. This is especially pertinent in light of the growing awareness of the dichotomy between the very real opportunities presented by terrestrial carbon sequestration and the practicalities of implementing the currently desired comprehensive-style verification systems. As such, the regional trends in soil organic carbon derived from SSPFs could provide both an indication of local soil carbon potential as well as a foundation database for farm level deviations from regional means.

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ACCURACY OF SOIL ORGANIC CARBON INVENTORIES IN MEDITERRANEAN MOUNTAIN AREAS.

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Abstract
With the objective to quantify the soil organic carbon content of the soils of a model area in the Catalan Pre-Pyrenees a detailed soil inventory was carried out. A nested sampling for organic matter and additional measurements of coarse elements and bulk density were needed to increase the accuracy of the estimations. Among the different profile site characteristics, land use and depth of SOC calculation affect the estimation of carbon stocks, in the sense that surface SOC in forests is higher than in agricultural land. More than half of the stored SOC is found in the first 30 cm, which indicates a high degradation susceptibility.

Key Words
Soil organic carbon, soil organic matter, soil mapping, geostatistics, Catalonia

Introduction
Soil properties play an important role in land use planning activities such as agriculture, erosion control, environmental protection and nature conservation. Soil organic matter is involved in processes like development and stabilization of the aggregates, in the biogeochemical nutrient cycles, and affects water and energy balances. The assessment of soil organic matter pools is essential for the evaluation of the ability of soil to sequester atmospheric carbon and to see whether soil changes induced by climate or land use may affect this ability. In order to do so with the required degree of precision, it is necessary to conduct soil surveys where organic matter contents, bulk densities, rock fragments and soil depths are determined and characterized without bias. Moreover, the sampling has to be georeferenced and be representative enough by working at adequate scales.

In Europe, the extrapolation of soil organic matter contents from the available data was not considered adequate because of lack of georeferences, of different survey methods and lack of standardization of analytical procedures (Jones *et al.,* 2005). As a consequence, the European Union defined (Stolbovoy *et al*., 2005) a protocol for the systematic soil sampling with the objective of detecting changes in soil carbon stocks.

Soil organic carbon content changes with depth. Generally, the highest levels are in the topsoil and they decrease with depth. The overall quantity of organic carbon in a given soil is determined largely by climate and organic inputs but can also be significantly affected by land use (McKenzie *et al.,* 2006).

The aim of this study is to obtain a detailed cartographic soil inventory and know the soil carbon stock in the main soils types and land uses, forest and farming lands, of a model area in the Iberian Pre-Pyrenees (Canalda river basin). This study wants to evaluate the incidence of land use changes on C storage in agro-silvo-pastoral ecosystems along climatic gradients through the determination of the soil potential for carbon storage, the quantification of this C reservoir, and to assess the precision of those estimations by means of (geo)statistical analyses.

Methods
The study area
The study area is located in a sub-basin from the Ribera Salada Basin (Catalan Pre-Pyrenees, NE Spain) called Canalda sub-basin, with an area of 10 km^2 and altitudes between 1100 and 2100 m, with predominant slopes between 10-50%. The parent materials are calcareous conglomerates, calcilutites and limestones. The annual rainfall is 500 to 750 mm, distributed along an altitudinal gradient. Agricultural uses include mainly cereals, potatoes and pastures, and the forest use varies from brook forest environments to subalpine and submediterranean vegetation. Soils are shallow, calcareous and stony, being most of them Inceptisols and Entisols. The area has been subjected to land use changes in the last 100 years, mainly the abandonment of agricultural land and its conversion to pastures or forest.

Cartographic soil inventory
The aim of the soil mapping is to know the soil carbon stock distribution. For this purpose, the CatSIS methodology (Boixadera *et al*., 1989), used in the soil map of Catalonia at a scale 1:25,000 has been applied. It has an intensity of 0.5 observations for cm^2 of final map. This working scale supposes 2 observations each 100 hectares, resulting in more than 20 pits in the model area. The soil classification used has been Soil Taxonomy (SSS 2006). The main steps developed in this soil survey were:
1. Application of photointerpretation and remote sensing;
2. Opening the soil pits in the units defined above;
3. Macromorphological field soil description;
4. Soil sampling and analyses (both physical and chemical);
5. Elaboration of the provisional map;
6. Rectification of the limits in the field by augerings;
7. Elaboration of the definite soil map.

Monitoring soil organic carbon
In each pit the routine soil physical characteristics were described, but in order to quantify carbon stocks some special measurements were carried out to characterize bulk density and coarse elements more in detail. These accompanying measurements are time consuming and rarely performed during routine soil testing (McKenzie *et al*., 2006), but are necessary, at least to depths of 300 mm, to quantify carbon (Porta *et al*., 2005).

Accordingly, the standard soil survey procedure used in the Soil Map of Catalonia was completed with the following determinations:

- Assessment of rock fragment volume. For each horizon, a 2 mm-mesh sieving was conducted in the field, the weight percentage of coarse fragments was measured, and it was converted to volume percentage using the density of quartz. Several kilograms of soil were taken for each horizon, down to a depth of 300 mm or to lithic contacts.
- The bulk density was measured by three methods: core sampling (Nacci *et al.*, 1999) and the aggregate method (Grossman *et al*., 2002) for each horizon; and the hole method (Nacci *et al*., 1999) for surface horizons.
- Organic carbon. It was determined by the standard wet oxidation method (Walkley-Black) and by a total carbon analyser (LECO).

The soil carbon stock was calculated for each soil mapping unit from the average of all pits in one unit. In order to achieve and compare soil carbon stock of different units, the following depth intervals were considered: 0 to 15 cm, 0 to 30 cm, 0 to 50 cm and 0 to 100 cm, disregarding organic horizons due to the low stability of organic matter. The 15 and 30 cm depths are chosen because they represent the stock of carbon susceptible to be influenced by anthropic action. SOC stocks are calculated adding up the SOC of each horizon until the given depth is achieved. The continuity of the profile is indispensable for the calculation of the SOC in depth. The following equation is used for the estimation of the SOC in each horizon (OCh):

$$OCh(Mg/ha) = Organic\ Carbon\ (\%) * Bulk\ density\ (kg/m^3) * Thickness\ (cm) * (1-Coarse\ elements) * 0{,}001$$

The calculation of the SOC down to 15 and 30 cm is done by using the following equations (50 and 100 cm estimations follow the same procedure):

$$SOC\ (15\ cm) = OCh_1 + \left(\frac{OCh_2}{Depth\ h_2} \cdot (15 - Upper\ boundary\ h_2) \right)$$

$$SOC\ (30\ cm) = OCh_1 + OCh_2 + \left(\frac{OCh_3}{Depth\ h_3} \cdot (30 - Upper\ boundary\ h_3) \right)$$

In order to know the precision of the organic matter calculations for each unit, an additional sampling of soil organic matter following a nested design (Figure 1) was conducted around the modal profiles of each soil mapping unit. In this approach, we assumed the rest of the variables to be those of the modal profile.

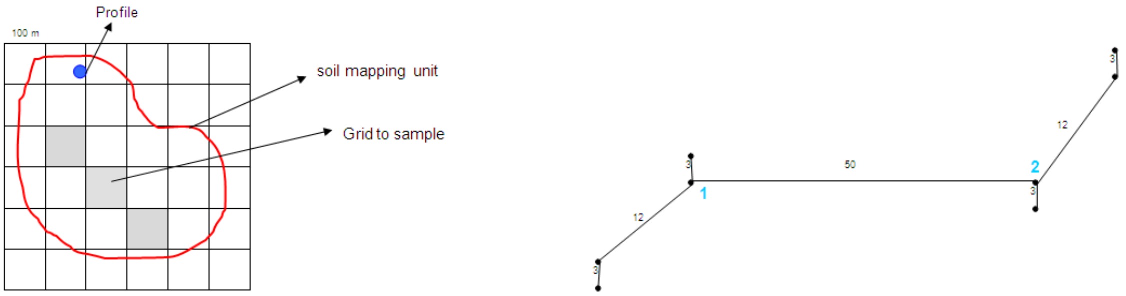

Figure 1. Nested sampling of soil organic matter. Two random starts points (1,2) are selected in each cell

Results

Only some of the results are presented. Table 1 shows some characteristics from the main modal profiles and the estimation of the carbon stocks. The depth is not the main factor that determines SOC, but the fact that many soils have buried horizons and a fluventic character.

Table 1. General characteristics to calculate SOC from some modal profiles. Coarse elements and bulk density are the interval values of the different horizons in each profile.

Modal profile	Depth	Horizon sequence	Coarse elements (%)	Bulk density (kg/m^3)	OCh (Mg/ha)	SOC (Mg/ha)
Typic Haplustepts	0-60	A_1 - B_{w1} - B_{w2} - R	64.04 - 73.13	1487.2 - 1655.2	A_1 - B_{w1} - B_{w2} 14 - 33.1 - 8.5	55.7
Typic Ustifluvents	0-110	A_1 - A_2 - AB - A_b - R	43.3 - 68.5	1352.3 - 1554.6	A_1 - A_2 - AB - A_b 27.2 - 26.7 - 33 - 39.6	126.7
Lithic Ustorthents	0-40	O_i - O_a - A_1 - A_2 - R	25 - 63.5	1000 - 1533.6	O_i - O_a - A_1 - A_2 24.6 - 53.8 - 33.7 - 49.7	83.5
Udic Calciustepts	0-180	A_{p1} - A_{p2} - B_w - B_k - C_1 - C_k	64.5 - 92.88	1390 - 1737.4	A_{p1} - A_{p2} - B_w - B_k - C_1 40.1 - 9.8 - 3.3 - 9 - 14.1	76.6
Typic Hapludolls	0-80	O - A_1 - B_{w1} - B_{w2}	20.19 - 70.64	1000 - 1782.7	A_1 - B_{w1} - B_{w2} 57.7 - 12.8 - 33.7	105.3
Typic Udorthents	0-81	O - A_1 - A_2 - B_k/R	46.92 - 84.23	1000 - 1686.9	A_1 - A_2 - B_k/R 54.8 - 41.9 - 34.4	131.1
Typic Calciustepts	0-120	O_i - O_a - A_1 - B_w - B_k - C	78.89 - 91.68	1000 - 1711.8	A_1 - B_w - B_k - C 28.6 - 31.5 - 15.5 - 16	91.9
Typic Eutrudepts	0-80	O - A_1 - B_{w1} - B_{w2} - B_k/C_k - C_{k1}	50.75 - 84.82	1063.6 - 1763.2	O - A_1 - B_{w1} - B_{w2} - B_k/C_k 39.3 - 9.4 - 2 - 2.5 - 19.8	33.9
Entic Hapludolls	0-140	A_1 - B_{w1} - B_{w2} - B_{w3} - B_{wk}	51.10 - 81.10	1346.7 - 2353.5	A_1 - B_{w1} - B_{w2} - B_{w3} - B_{wk} 64.9 - 23.4 - 6.1 - 6.9 - 6.4	107.9
Typic Calciustolls	0-100	O_i - A_1 - A_2 - B_w - B_k	78.88 - 81.11	1000 - 1700	O_i - A_1 - A_2 - B_w - B_k 21 - 27.4 - 18.7 - 10.7 - 11.6	68.6

The SOC values down to each depth are shown in Figure 2. The number of samples decreases with depth, because some soils are shallow and stony. More than half of the SOC is stored in the first 30 cm, which stresses the importance of A horizons in soils of mountain areas.

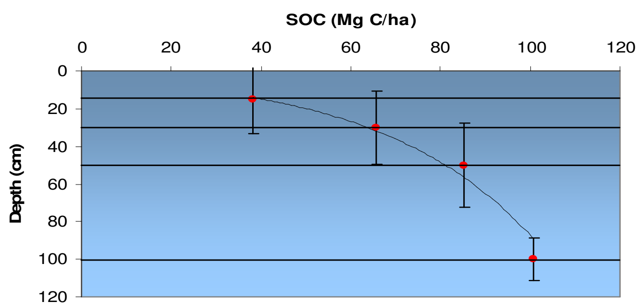

Figure 2. The SOC values down to each depth. The number of samples is 22 (10 cm), 22 (30 cm), 21 (50 cm) and 14 (100 cm). Bars show the standard deviation.

The data was checked for differences between altitudes and land use. A non parametric method (Kruskal-Wallis and Kolmogorov-Smirnov tests) was applied, since the SOC data do not follow a normal distribution. The SOC values did not show any significant difference between altitudes, but was higher at 15 cm under forest (P<0.05). Table 3 shows the descriptive statistics for this analysis.

Table 3. Soil organic carbon stocks to different depth (Mg/ha) according to land use

Land use	Variable	Number of profiles	Average	Median	Maximum	Minimum	Standard deviation	Variation Coefficient (%)
Agricultural	SOC15 (Mg/ha)	6	21,24	21,65	32,47	7,39	9,27	43,64
	SOC30 (Mg/ha)	6	55,67	53,72	76,67	45,08	11,65	20,92
	SOC50 (Mg/ha)	6	71,00	72,26	92,05	51,67	15,31	21,57
	SOC100 (Mg/ha)	5	86,93	97,05	102,81	65,23	18,45	21,22
Forest	SOC15 (Mg/ha)	16	44,61	33,59	187,07	11,53	41,25	92,46
	SOC30 (Mg/ha)	16	69,25	58,84	205,80	14,05	45,29	65,39
	SOC50 (Mg/ha)	15	91,08	70,06	219,01	24,88	51,31	56,34
	SOC100 (Mg/ha)	9	108,30	113,55	135,31	67,79	22,26	20,56

Conclusion

When comparing SOC between soil mapping units the soil depth has to be defined, in order to obtain unbiased estimations resulting from soil surveys not oriented to carbon inventories. Our results show that the standard depth of 1 meter in most of SOC inventories may not be necessary to increase the accuracy, since more than 50% of SOC is found in the upper 30 cm. Land use is affecting SOC storage, in the sense that agricultural soils have less SOC than forest soils at the surface. This fact increases the vulnerability of forest soils because they can lose the capacity of storing carbon after a forest fire or erosion episodes more rapidly.

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Assessment of soil carbon stores at the farm scale in Tasmania, Australia

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Abstract
The measured farm soil carbon stores in the upper 30 cm of soil were 371 to 699 T/ha CO_2 equivalents across the three farms assessed. The highest value occurred on predominantly Dermosols and Ferrosols, which have high clay contents, are under perennial irrigated pasture for dairying, and have a mean annual rainfall of 1242 mm. The lower soil carbon stores occurred on Kurosols, Sodosols and Tenosols which have sandy loam surface textures, are used for cropping and have mean annual rainfalls of 560 – 760 mm. This study demonstrates that farmers are custodians of a large 'bank' of soil carbon which is susceptible to degradation and conversion into CO_2 if management is not sustainable. The calculated farm carbon storage in the upper 30 cm of soils varied depending on the scale of investigation. Broad scale assessment using information from the Australian Soil Resource Information System ranged from being 25 - 82% less than that determined from farm scale information.

Key Words
Carbon stores, assessment scale

Introduction
Recent concern over the contribution made by agriculture to greenhouse gas production has lead to interest in soil carbon as a potential store for atmospheric carbon (Izaurralde *et al.,* 2001; Scholes and Noble, 2001). There is also increasing pressure and demand for estimates of current soil organic carbon stocks as well for information on how different farming enterprises can be managed in order to minimise their carbon footprint. It is important, however, that accurate and reliable data are used as the basis for these estimates in order to minimise errors. Some farmers are interested in the 'big picture' of carbon stores and fluxes on their farm rather than just their emissions and sequestrations which are the focus of current carbon accounting using existing carbon calculators such as FullCAM (Richards *et al.,* 2005). The farmers involved in this study wanted to see information which is already routinely compiled as part of a well delivered property management plan e.g. farm map, soil types, land use areas (cropping, pastures) used to demonstrate the valuable role that farmers play as carbon stewards.
The objectives of this research were to measure current on-farm soil carbon stores by physical measurement, and compare different scales of assessment of farm carbon stores

Methods
Three pilot farms were used that were representative of enterprise types in Tasmania. Seventeen sites were selected on each of the three farms. These sites were representative of the soils and topography previously mapped on the properties. Not all mapped polygons were sampled and some large polygons or soils with multiple polygons had multiple samples taken. Measurement of soil carbon was undertaken according to the protocols of McKenzie and Dixon (2006). Sampling was carried out in September. Samples were collected in areas sufficiently far from fence lines, gateways and headlands to avoid these edge effects. Five soil cores were taken along a 60 m transect using a 50 mm diameter push auger. Cores were combined to form a single composite specimen for each of 3 depths, 0-50 mm, 50-100 mm and 100–300 mm. These samples were dried at 40^oC for at least 48 hours, ground to pass a 2 mm sieve, and stored in air-tight containers. The samples were then analysed for total carbon by dry furnace combustion (Rayment & Higginson, 1992). Bulk density was measured at the sampling site in order to calculate the mass of soil organic carbon (area and depth). Stainless steel cylinders, 60 mm long and 60 mm in diameter, were hammered into the soil at the starting point of the composite sampling transect. Cores were collected from 0-60 mm, 50-110 mm and 150-210 mm depth. Cores with soil intact were excavated and trimmed before the contents were emptied into plastic bags, dried at 105^oC, and then weighed. The mass of carbon stored at each sampling depth, to a total depth of 30 cm, was calculated and converted to carbon dioxide equivalents (CO_2e) by multiplying by 44/12. This carbon dioxide mass value was then multiplied by the area of mapped polygons of each soil type occurring on the farm and the totals summed for the entire farm property.

Assessment of farm carbon stores was compared using the farm scale maps and regional scale information on the Australian soil resource information system (ASRIS) which is accessible via the world wide web. Farm scale soil map information used for Farm A was 1: 10 000 scale land capability mapping at subclass and unit level, for Farm B soil map information was a combination of 1: 10 000 scale and 1: 100 000 scale soil type or complex mapping, and for Farm C 1: 25 000 scale land capability at subclass and unit level was used. The mass of carbon was calculated on a per hectare basis using the data collected from the 17 sites on each farm. These values were then multiplied by the by the mapped areas of the soils or land capability units they represent to give a total farm carbon store in CO_2e.

Soil attribution for ASRIS in Tasmania has been completed for the North West and North East regions providing soil information for two thirds of the agricultural land of Tasmania at ASRIS Land District level which is equivalent to a scale of 1: 250,000. Information accessed from ASRIS for the 1-2 polygons mapped on each farm included the soil carbon content (%) and bulk density for each layer to a depth of 300 mm. The information provided on ASRIS is an integrated value of soil properties for each polygon, based on up to 6 soil components for each polygon attributed in the database. The mass of carbon was calculated from the ASRIS data on per hectare basis which was then multiplied by the area of farm represented by the corresponding polygon to give a total farm carbon store.

Results and discussion

The soil carbon stored on each of the pilot farms in the upper 30 cm of soil (Table 1) was calculated as: Farm A 170,454 T CO_2e; Farm B 302,300 T CO_2e; Farm C 213,445 T CO_2e (Farm A 46,487 T C; Farm B 82,445 T C; Farm C 58,212 T C). The farm total soil carbon stores are large and indicate that farmers are custodians of a large amount of soil carbon. Farm B was the largest property (753 ha) and had the greatest soil carbon store. Farm C was the smallest property (305 ha) but it had a greater soil carbon store than Farm A (460 ha), which is likely to be due to a combination of soil type, land use and climate. The soil carbon stores amount to 371, 401 and 699 T/ha CO_2e (101, 109, 191 T C/ha) for Farms A, B and C respectively. The highest per hectare value on Farm C occurred on predominantly Ferrosols and Dermosols, which have high clay contents, are under perennial irrigated pasture for dairying, and have a mean annual rainfall of 1242 mm. Farms A and B are both dominated by Kurosols, Sodosols and Tenosols which have sandy loam surface textures, are used for cropping and have mean annual rainfalls of 766 mm (Farm A) and 562 mm (Farm B). Under Tasmanian conditions, clay textured soils (Dermosols, Ferrosols, Vertosols) have been found to have greater soil carbon contents than sandy textured soils (Kurosols, Sodosols, Tenosols) and perennial plant systems such as permanent pasture result in greater soil carbon contents than cropping systems due to greater inputs over the long term (Cotching, 2009).

The farm carbon storage in the upper 30 cm of soils determined from broad scale data obtained from the ASRIS web site is compared with that determined at the farm scale in this study (Table 1). The storage values for Farm A are the most similar, but the broad scale ASRIS value is 25% less than that determined from farm scale information. The difference for Farm C is 33% and for Farm B it is 82%. The differences are similar or much greater than those found by Frogbrook *et al.* (2009) who found differences of 8% and 45% for areas in Scotland and Wales respectively when comparing field survey data with information from the national UK database. The differences in this study are likely to be mainly due to soils mapped at the farm scale are not included as components in the broad scale ASRIS information. This is most obvious on Farm B which had the largest discrepancy, where 246 ha of Vertosols (33% of total farm area) were mapped at the farm scale but these were not identified in the ASRIS data (Table 1). Clay rich Vertosols have much higher soil carbon contents than sandy loam to loamy sand textured Kurosols and Tenosols in Tasmania (Cotching *et al*. 2002). Other factors likely to have contributed to the differences are attributed depth, soil carbon and/or bulk density values in the ASRIS data are derived from similarly mapped land system polygons which are not representative of soils on these specific farms.

The ASRIS data used in this study was drawn from detailed soil maps and/or soil profile descriptions with all necessary analytical data which cover 9.9% of Tasmania. These areas are concentrated in the more intensively used agricultural areas in the northern Midlands and the northwest coast and cover the three pilot farms used in this research. The soil carbon data was either based on a single measurement in the land unit tract, or based on direct measurements of similar soils in the same land unit type.

Table 1. Farm soil carbon storage determined from ASRIS data and farm scale measurements.

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Farm	Map unit area (ha)	Soil order# composition (%)	Soil layer	Layer thickness (m)	Soil organic carbon (%)	Bulk density (T/m^3)	Carbon (T/ha)	ASRIS soil carbon (T CO_2e)	ASRIS farm soil carbon (T CO_2e)	Farm scale soil carbon (T CO_2e)*
A	460	De 45, Ku 30, Hy 25	1	0.24	2.05	1.4	68.9			
			2	0.06	0.80	1.4	6.7	127,512	127,512	170,454
B	22	De 60, Ch 40	1	0.14	1.30	1.2	21.8			
			2	0.16	0.37	1.3	7.7	2,339		
	696	So 65, Te 35	1	0.14	0.88	1.3	16.0			
			2	0.16	0.11	1.4	2.5	47,161		
	38	Te 100	1	0.14	1.30	1.2	21.8			
			2	0.16	0.37	1.3	7.7	4,115	53,616	302,300
C	42	De 100	1	0.13	8.22	0.8	85.5			
			2	0.04	4.55	1.0	18.2			
			3	0.13	1.94	1.2	30.3	20,432		
	263	Fe 100	1	0.15	5.83	0.9	78.7			
			2	0.15	2.92	1.1	48.2	122,546	142,978	214,463

*data from Table 1; # Ch=Chromosol, De=Dermosol, Fe=Ferrosol, Ku=Kurosol, Hy=Hydrosol, So=Sodosol, Te=Tenosol

This means that for the reconnaissance scale ASRIS data, at best soil carbon data from one site has been applied to the whole polygon, and so it is not surprising that there are significant differences with the greater intensity of sampling and attribution used in this study. In other agricultural areas of Tasmania where reconnaissance soil maps and profile descriptions with incomplete analytical data were available (17.5%), or in native and plantation forestry areas and agricultural areas with low intensity use (29%) where reconnaissance soil maps and/or knowledge of similar soils in similar environments are available, the reliability of ASRIS data for calculating soil carbon storage is likely to be relied on with even less confidence. The discrepancies in this study are disturbingly large and indicate that the use of broad scale information within ASRIS, which is the Australian national database, can lead to large errors in calculating on-farm soil carbon storage.

Conclusions

The measured farm soil carbon stores were 371 to 699 T/ha CO_2e (101 – 191 T C/ha) across the three farms assessed in this pilot. The highest value occurred on a farm with predominantly Ferrosols and Dermosols, which have high clay contents, are under perennial irrigated pasture for dairying, and have a mean annual rainfall of 1242 mm. The lower soil carbon stores occurred on Kurosols, Sodosols and Tenosols which have sandy loam surface textures, are used for cropping and have mean annual rainfalls of 560 – 760 mm. The largest property (753 ha) and had the greatest soil carbon store (302,300 T CO_2e) but the smallest property (305 ha) had a greater soil carbon store (213,445 T CO_2e) than the 460 ha property (170,454 T CO_2e) due to a combination of soil type, land use and climate. This study demonstrates that farmers are custodians of a large 'bank' of soil carbon which is susceptible to degradation and conversion into CO_2 if management is not sustainable.

The calculated farm carbon storage in the upper 30 cm of soils varied depending on the scale of investigation. Broad scale assessment using ASRIS information ranged from being 25 - 82% less than that determined from farm scale information. The differences are disturbingly large and indicate that the use of currently available broad scale information can lead to large errors in calculating farm soil carbon storage. The result is perhaps unsurprising given that the ASRIS data is of relatively small resolution. It must be emphasized that this study sampled three farms in the north and northeast of Tasmania. Additional sampling from other locations, where there are a range of soil types encompassing other land use types and topography, would further contribute to improving the estimates of the amount of carbon held on farms in Tasmania.

Acknowledgements

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Assessment of the spatial relationship between soil properties and topography over a landscape

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Abstract
The objective of this study was to assess the spatial relationships between soil properties and topography over a watershed cultivated with sugarcane. Soil samples were collected at 0-20cm depth from 244 data points approximatelly evenly distributed over the entire watershed. The coordinates of the sampling points were recorded with a DGPS. In each sampling point, average topographic height, slope and aspect were calculated from the digital elevation model in a GIS environment. Soil samples were analysed for physical and chemical routine characterizations. The spatial dependence of each individual variable as well as the relationship between them were evaluated with semivariogram and cross semivariograms. Soil phosphorus and calcium showed extremelly high variability owing to the variety of soil types and depths over the watershed. All variables studied were spatially dependent. There was spatial dependence between slope percentage and total sand, natural clay, phosphorus, calcium, cation exchange capacity and bases saturation. A positive correlation was characterized between slope and total sand and a negative one between slope and the other five variables. These results indicate the possibility to map those properties using cokriging with slope as the auxiliary variable.

Key Words
Terrain slope, topographic variables, semivariogram, cross semivariogram

Introduction
Depending upon the scale of the measurements and the complexity of the environment from which the data are collected, classical statistics fails to represent the data. In most of these situations, geostatistics is the appropriate tool to describe the spatial variability and the relationships between data. It is generally recognized that soils can vary widely as a function of their position on the landscape, parent material variability, erosion history, and cultivation. The amount of variation over an area depends on many environmental conditions and how they have been acting on soil properties over time. Spatial variability of soil properties has been long known to exist and has to be taken into account every time field sampling is performed. Beckett and Webster (1971) presented a very comprehensive review with deep discussion of soil variability on soil chemical properties. Soil variability can also occur as a result of agricultural management, land use and erosion (Vieira, 2000). Ceddia *et al.* (2009) related some soil physical attributes with topography over a landscape and concluded that it was viable to use cokriging with topography as an auxiliary variable to map sand, clay and water retention parameters. The objective of this study was to assess the spatial relationships between soil properties and topographic attributes over a watershed cultivated with sugarcane. The fundamental assumption is that mapping soil properties using auxiliary variables may help to understand the processes occurring over a landscape and improve the precision of the constructed maps.

Material and Methods
The experimental area is located on a watershed named Ceveiro near Piracicaba, SP, Brazil. Soil samples were collected at 0-20cm depth from 244 data points evenly distributed over the entire wastershed. The coordinates of the sampling points were recorded with a DGPS, and their position within the watershed are shown in Figure 1. In this figure, the symbol code classification represents the average topographical height or altitude according to five classes of equal number. Average topographic heights or altitudes (m), slopes (% and degrees) and aspects (degrees) were calculated from digital elevation model in a GIS environment having as feature definition image the sampling points. Soil samples were analyzed for physical and chemical routine characterizations. The spatial dependence of each individual variable as well as the relationship between them were evaluated with semivariogram and cross semivariograms. A total of twenty-nine soil and topographic variables were analysed.

The Geostatistical Approach
The spatial dependence of soil properties, according to Vieira *et al.* (1983), can be evaluated by examining the

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semivariogram. If the semivariogram increases with distance and stabilizes at the a priori variance value, it means that the variable under study is spatially correlated and all neighbours within the correlation range can be used to interpolate values where they were not measured. Moreover the spatial relationship between variables can be evaluated using the cross semivariogram. Semivariogram modeling is the foundation for geostatistical analysis, and can also be the most difficult and time consuming portion of the analysis. In part, this is due to the computationally intensive calculations, but it is also due to the difficulty in defining semivariogram models which reasonably honor the experimental semivariograms (McBratney and Webster, 1986). The models fitted are described by the parameters C_0, which express the nugget effect, C_1, the structural variance, and *a,* the range of spatial dependence. In this work, the models were fit by using least squares minimization and judging the coefficient of determination. Whenever there was any doubt on the parameters and model fit, the jack knifing procedure was used to validate the model, according to Vieira (2000).

Results and discussion

The descriptive statistical parameters for ten soil and topographic variables which showed spatial correlation between pairs are illustrated in Table 1. It can be seen that most of the variables express an extremely high variability. The highest of all is for phosphorus content (P) with a CV of 335%. Obviously that depends on the dimension of the area under study. Ceddia *et al.* (2009) have found 65% for clay content on a 2.64ha field, although Siqueira et al. (2008) have found 35% for clay content for a 3.52ha field in. These results indicate that the terrain variability depends not only on the field size but also in the sampling intensity with respect to the size. The obtained semivariograms for topographic variables (Figure 2) revealed that for four studied variables data were best fit by the exponential model. The range, which marks the limit of spatial dependence, is around 180m for slope, 900m for aspect, and 1,700.00 for altitude. On Figure 3, semivariograms for the soil variables show that range varies around 2,000m for six studied variables. The best fits were obtained for total sand (R2= 0.9725), natural clay (r2=0.9468), and base saturation (r2= 0.9120). The semivariograms for phosphorous (P), calcium (Ca), and cations exchange capacity (CEC) were adjusted employing the spherical model. The poorer adjust was for P (r2=0.4589), followed by Ca (r2= 0.7909) and CEC (r2=0.8563). The cross semivariograms revealed a positive correlation between total sand and slope, which could be related with the occurrence of Ultisols and Alfisols on the highest altitudes at the northern and northwestern borders of the watershed, as described by Weill & Sparovek (2008). The four chemical variables, P, Ca, CEC, and base saturation (V) have showed a negative correlation with slope. This is related with the fact that shallower soils, like Entisols and Inceptisols, which maintain a major influence from the parent material, occur at more gentle slopes in Ceveiro watershed. Moreover, the tendency to bases accumulation in the lower part of the hillside is also known. In relation to the natural clay, the negative correlation with slope could be due to the fact that in the watershed the soil as a whole and the sandy topsoil of the Ultisols and Alfisols are deeper and the textural gradients tend to be higher as the slope declines.

Conclusions

The geostatistical approach could enhance the comprehension of some processes occurring over the Ceveiro watershed. Some soil properties could be mapped using cokriging with slope as the auxiliary variable.

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Table 1. Descriptive statistics for topographic and soil variables in Ceveiro watershed.

Variable	Mean	Variance	C.V.	Minimum	Maximum	Skewness	Kurtosis
Altitude	508.70	646.60	5.00	464.10	582.50	0.659	0.046
Slope (%)	10.21	39.00	61.15	0.00	35.36	0.812	0.814
Slope (degrees)	5.81	12.31	60.42	0.00	19.47	0.757	0.624
Aspect (degrees)	169.60	9614.00	57.81	0.00	355.5	-0.258	-1.062
Total sand (%)	61.12	554.00	38.51	11.00	94.00	-0.605	-1.091
Natural clay (%)	12.36	102.20	81.82	0.00	46.00	1.074	0.183
P (mg kg^{-1})	15.69	2777.00	335.80	1.00	770.00	12.26	172.50
Ca ($mmol_c$ kg^{-1})	27.54	801.00	102.80	1.00	205.00	2.175	7.079
CEC ($mmol_c$ kg^{-1})	78.87	2184.00	59.25	12.80	246.50	1.268	1.261
V (%)	43.91	437.20	47.62	6.00	96.00	0.465	-0.654

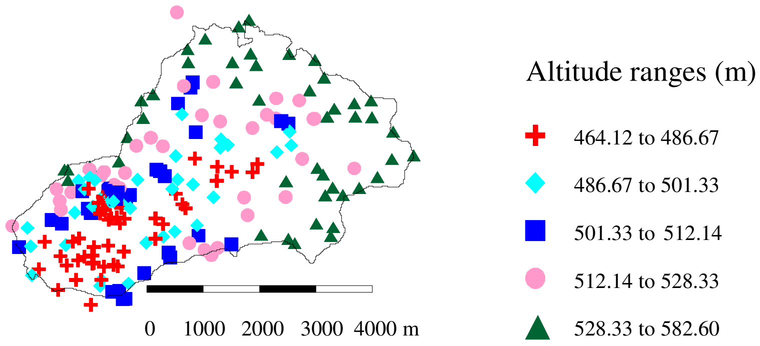

Figure 1. Location of sampling points within the Ceveiro wastershed (SP, Brazil) with topographic height.

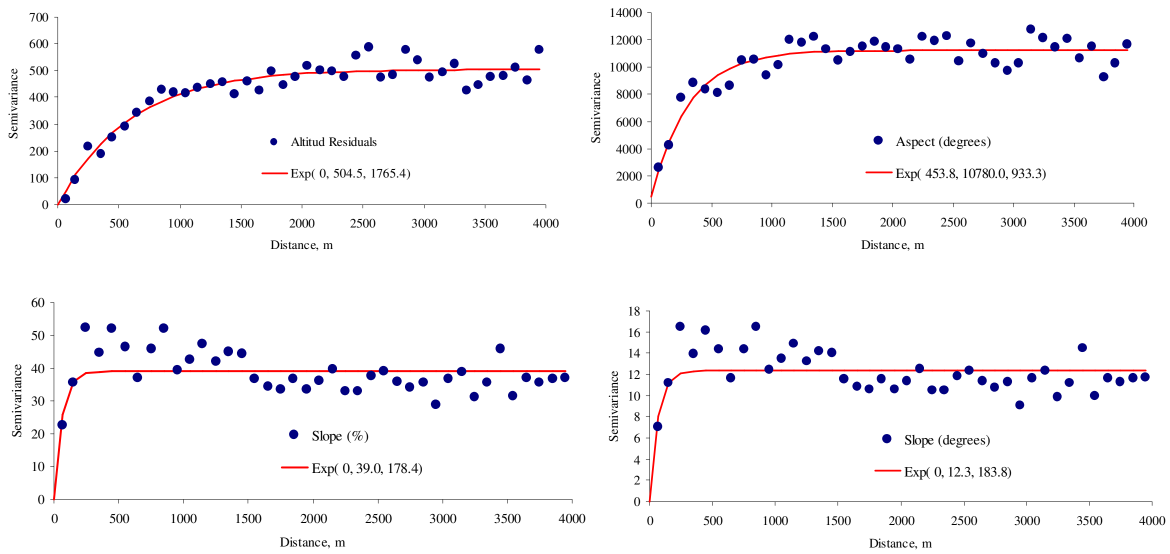

Figure 2. Semivariograms for the topographic variables: altitude residuals, aspect, and slope.

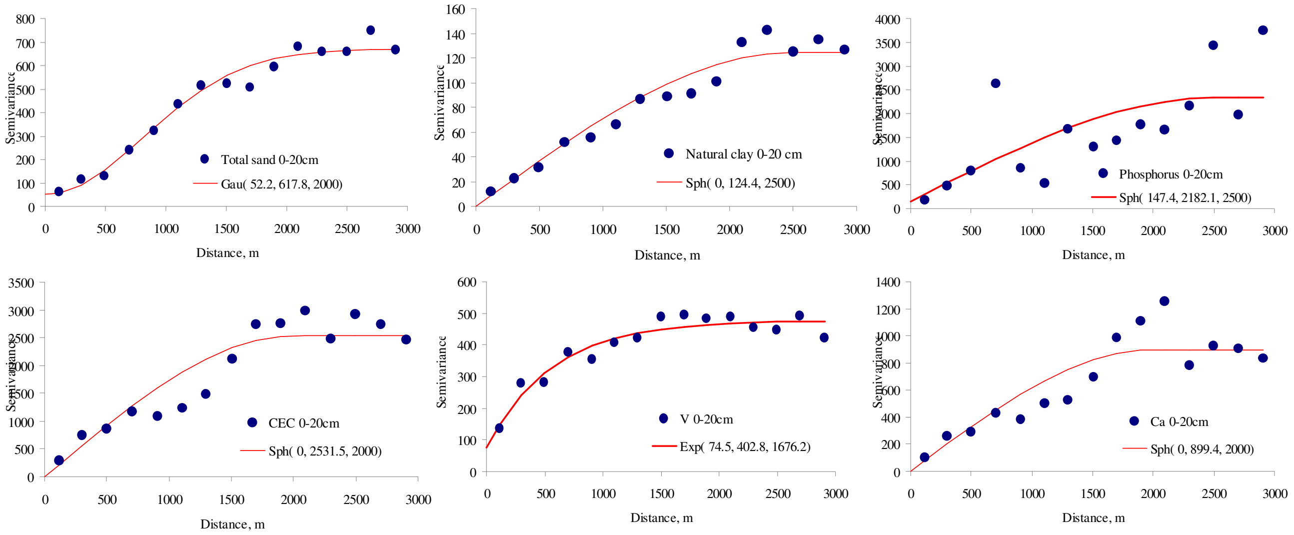

Figure 3. Semivariograms for soil variables: total sand, natural clay, phosphorus(P), calcium(Ca), cation exchange capacity (CTC), and base saturation (V).

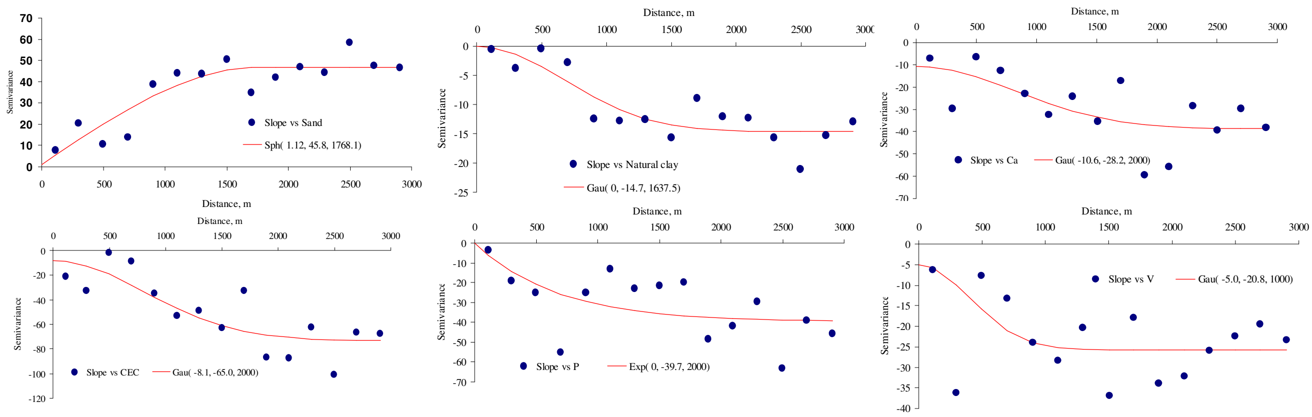

Figure 4. Cross semivariograms between slope and soil properties.

Comparison of soil quality targets and background concentrations in soil of the Waikato Region, New Zealand.

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Abstract
Current targets used for assessing soil quality in New Zealand were compared with 95% confidence intervals of measurements of background soils from the Waikato region, New Zealand. Indicators analysed included pH, total C and N, Olsen P, anaerobically mineralised N, bulk density, macroporosity, aggregate stability, As, Cd, Cr, Cu, Hg, Pb, Ni and Zn.

While background concentrations are easily applied to some measurements (metals), for other soil quality measurements (particularly those with optimum ranges rather than just maximum allowable limits such as bulk density and macroporosity) the concept is more difficult to apply. Background sites met the targets for pH, total C, aggregate stability, Zn, Cu, Pb, As, Cr Ni, Cd and Hg. Some background sites didn’t meet the targets for Olsen P, as New Zealand soils are naturally low in phosphorous. Also, some sites didn’t meet the targets for bulk density and macroporosity indicating an erosion risk. Targets for total N and anaerobically mineralised N on their own are inadequate to assess N status and the N leaching risk, and trends in these measurements may be more valuable than meeting an actual target.

Key Words
Background, soil quality, indicators.

Introduction
Regional environmental authorities in New Zealand have been monitoring soil quality since 1995. A set of indicators was agreed upon and the majority of target values were set by “expert” opinion (or for metals, adoption of guidelines from NZWWA). Currently these target values are being reviewed. Expert opinion target values were based on a “fitness for use” standard and one concern has been the applicability of these target values to indigenous systems.

Background concentrations of soil properties provide a reference for assessing the impacts of anthropogenic activity, including changes in land uses. The Ministry for the Environment’s Contaminated Land Management Guideline number 4 (MfE 2006) defines background concentration as: “An estimate of the natural concentration of a substance (element, compound or mixture) that would exist in the absence of any anthropogenic input, usually on a regional, sub-regional or catchment basis”. For chemical elements in soils, the background concentration is expected to show some broad-scale variation depending on the nature of the geochemical parent materials. Other factors that affect background levels are natural surface inputs (volcanic ash, dust, fluvial deposition and atmospheric aerosol deposition) and disturbances such as a tree turnover. Consequently, background soil properties can vary widely. A site is considered to be above background concentrations when the concentration of a contaminant is clearly higher than its background concentration. Factors such as the confidence limit (95% CL) of the background concentration, the number of samples collected and their representativeness, observed or expected variability associated with sampling and analysis, and applicable guideline values are considered in the assessment. In this study, background levels for soil quality indicators from indigenous sites throughout the Waikato region were assessed and compared with suggested target values.

Methods
Background sites were identified on the basis of their current land use and what is known of their land use history. They were long-term forest or wetlands, with minimal influence from anthropogenic activities for the life of the vegetation or longer. Some of these sites may have been logged or cleared by early generations, but atmospheric inputs in New Zealand soils are relatively low, and for the most part these sites are regarded as being close enough to background to serve as a useful point of comparison. Sampling for chemical and biochemical indicators consisted of 25 soil cores (0-100 mm, 25mm diameter) over a 50 m transect, which are combined to form composites for analysis (Sparling *et al*. 2002). Sampling for physical indicators consisted of 3 soil cores (10-90 mm, 100mm diameter), which are individually analysed and the results averaged for each site.

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Samples are analysed for an established set of soil quality chemical and physical parameters following Sparling *et al*. (2002) and for trace elements following EPA 200.2 (total recoverable metals hydrochloric/nitric acid digestion). Measurements were made at IANZ-accredited laboratories (soil quality chemistry at Landcare Research, Palmerston North, soil quality physical parameters at Landcare Research, Hamilton, aggregate analysis at Plant & Food Research, Lincoln and elemental analysis by ICP-MS at Hill Laboratories, Hamilton).

Results and discussion

Background measurements of soil chemical and physical soil quality indicators found in the Waikato region were identified and 95% confidence limits fitted (Table 1). The 95% confidence limits of pH, aggregate stability, Zn, Cu, Pb, As, Cr Ni, Cd and Hg fitted into clear categories when compared with current targets for soil quality parameters for forests.

Table 1. Average background concentrations of soil quality indicators in Waikato mineral soils.

Element	Average	95% CL	n	Current target rating
pH	5.2	4.5-5.7	36	Optimal[1]
Total C (%)	8.3	3.5-16.0	36	Normal to ample[1]
Total N (%)	0.45	0.18-0.91	36	Depleted to high[1]
Olsen P (mg/kg)	6.6	1.1-14.9	36	Very low to adequate[1]
Anaerobically mineralised N (mg/kg)	117	46-225	35	Adequate to excessive[1]
Bulk Density (t/ha)	0.69	0.41-1.01	35	Very loose to adequate[1]
Macroporosity (% at -10 kPa)	22	8-39	35	Low to high[1]
Aggregate stability (MWD, mm)	2.21	1.96-2.62	13	Optimal[2]
Zn (mg/kg)	29.9	11.2-57.4	26	Below guidelines[3]
Cu (mg/kg)	13.8	5.1-27.3	26	Below guidelines[3]
Pb (mg/kg)	10.3	3.1-25.6	25	Below guidelines[3]
As (mg/kg)	4.9	0.70-9.0	26	Below guidelines[3]
Cr (mg/kg)	4.56	0.75-10.7	26	Below guidelines[3]
Ni (mg/kg)	3.18	0.69-9.19	25	Below guidelines[3]
Cd (mg/kg)	0.12	0.04-.28	26	Below guidelines[3]
Hg (mg/kg)	0.11	0.05-0.27	26	Below guidelines[3]

[1] Sparling *et al*. (2003)
[2] Beare *et al*. (2006)
[3] NZWWA (2003)

Total carbon fitted across two categories but both these are considered as meeting soil quality targets.

Total N ranged from one extreme to the other. Too little total N in managed land uses restricts production due to N deficiency and too much increases the risk of N leaching. Too little labile N is not likely to be harmful in native systems as they are often adapted to low nutrient conditions. Too much available N however has been linked to forest degradation in northern hemisphere forests subjected to high atmospheric N deposition. Total N on its own appears inadequate to measure nitrogen status for soil quality assessment (Taylor 2009). Trends in total N may be more valuable than meeting an actual target.

Olsen P ranged from very low to adequate, supporting the view that New Zealand soils are regarded as being naturally low in P (McLaren & Cameron 1990). The very low rating is outside the soil quality target, but many indigenous forests are considered P limited (Parfitt *et al*. 2005) and it is possible that higher P levels could result in changes in competitive dynamics between species.

Anaerobically mineralised N ranged from adequate to excessive. Too little anaerobically mineralised N restricts production, while too much (excessive rating) indicates an increased risk of nitrogen transfer. However, anaerobically mineralised N was strongly positively correlated with total carbon ($R=0.696$, $p<0.0001$), which reduces the N leaching risk due to sorption of N. This result supports the view that high anaerobically mineralised N on its own is inadequate to measure the risk of nitrogen transfer to water or the atmosphere

(Taylor 2009). Trends in anaerobically mineralised N may be more valuable than meeting an actual target.

Bulk density ranged from very loose (low bulk density) to adequate and this property is strongly influenced by soil type (Table 2). The target range is met if bulk density is rated loose or adequate. Targets are not met if soils are rated very loose or compact (no background sites were rated compact). Soils of the Waikato region, generally, have low bulk density and these soils could have an increased erosion risk if cleared of vegetation. This risk is identified by the current soil quality targets for bulk density. However, it is debatable if low bulk density soils are non-desirable as these soils are in their natural state and have successfully supported the native vegetation for centuries. To say that these soils do not meet soil quality targets is, therefore, incorrect. The bulk density indicator is providing warning of the fragility of some of the soils of the Waikato region.

Table 2. Mean surface (0-100 mm) bulk density and macroporosity (-10kPa) measurements of different soil orders averaged over all land uses in Waikato soils

Soil Order[1]	Bulk density (t/ha)		Macroporosity (-10 kPa) (%)		n
	Average	95% CL	Average	95% CL	
Podzols	0.53	0.42-0.64	30	16-40	10
Pumice Soils	0.64	0.49-0.80	24	7-43	34
Allophanic Soils	0.70	0.49-0.88	16	4-36	56
Gley Soils	0.85	0.57-1.04	13	6-23	24
Recent Soils	0.87	0.71-1.04	17	6-47	9
Brown Soils	0.91	0.76-1.11	13	6-23	27
Granular Soils	0.97	0.72-1.32	15	2-28	17

[1] Hewitt 2002, New Zealand Soil Classification

Macroporosity had a very wide range of categories, from low to high. Also, like bulk density, macroporosity is strongly influenced by soil type (Table 2) and the two measurement are strongly negatively correlated (r=0.732, p=<0.0001). Targets are met if the macroporosity rating is low or adequate. Targets are not met if the rating is very low or high. Very low macroporosity inhibits soil aeration and plant root growth, while high macroporosity is indicative of an increased erosion risk, similar to low bulk density. A similar argument to that for bulk density applies. It is questionable if high macroporosity soils are non-desirable as these soils are in their natural state and have successfully supported the native vegetation for centuries. To say that these soils do not meet soil quality targets is, therefore, incorrect. Like bulk density, the macroporosity indicator is providing warning of the fragility of some of the soils of the Waikato region.

Conclusion

Background concentrations are easily applied to some measurements (e.g. metals) but for other soil quality measurements (e.g. bulk density and macroporosity) the concept is more difficult to apply. Background sites met the targets for pH, total C, aggregate stability, Zn, Cu, Pb, As, Cr Ni, Cd and Hg. Some background sites didn't meet the targets for Olsen P, as New Zealand soils are naturally low in phosphorous. Also, some sites didn't meet the targets for bulk density and macroporosity. However, it is questionable if these soils are non-desirable as they are in their natural state and have successfully supported the native vegetation for centuries. Targets for total N and anaerobically mineralised N on their own are inadequate to assess nitrogen status and risk of transfer to other parts of the environment. Trends in anaerobically mineralised N over time may be more valuable than meeting an actual target.

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Comparison of soil quality targets and background concentrations in soil of the Waikato Region, New Zealand

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While background concentrations are easily applied to some measurements (metals), for other soil quality measurements (particularly those with optimum ranges rather than just maximum allowable limits such as bulk density and macroporosity) the concept is more difficult to apply. Background sites met the targets for pH, total C, aggregate stability, Zn, Cu, Pb, As, Cr Ni, Cd and Hg. Some background sites didn't meet the targets for Olsen P, as New Zealand soils are naturally low in phosphorous. Also, some sites didn't meet the targets for bulk density and macroporosity indicating an erosion risk. Targets for total N and anaerobically mineralised N on their own are inadequate to assess N status and the N leaching risk, and trends in these measurements may be more valuable than meeting an actual target.

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Introduction
Regional environmental authorities in New Zealand have been monitoring soil quality since 1995. A set of indicators was agreed upon and the majority of target values were set by "expert" opinion (or for metals, adoption of guidelines from NZWWA). Currently these target values are being reviewed. Expert opinion target values were based on a "fitness for use" standard and one concern has been the applicability of these target values to indigenous systems.

Background concentrations of soil properties provide a reference for assessing the impacts of anthropogenic activity, including changes in land uses. The Ministry for the Environment's Contaminated Land Management Guideline number 4 (MfE 2006) defines background concentration as: "An estimate of the natural concentration of a substance (element, compound or mixture) that would exist in the absence of any anthropogenic input, usually on a regional, sub-regional or catchment basis". For chemical elements in soils, the background concentration is expected to show some broad-scale variation depending on the nature of the geochemical parent materials. Other factors that affect background levels are natural surface inputs (volcanic ash, dust, fluvial deposition and atmospheric aerosol deposition) and disturbances such as a tree turnover. Consequently, background soil properties can vary widely. A site is considered to be above background concentrations when the concentration of a contaminant is clearly higher than its background concentration. Factors such as the confidence limit (95% CL) of the background concentration, the number of samples collected and their representativeness, observed or expected variability associated with sampling and analysis, and applicable guideline values are considered in the assessment. In this study, background levels for soil quality indicators from indigenous sites throughout the Waikato region were assessed and compared with suggested target values.

Methods
Background sites were identified on the basis of their current land use and what is known of their land use history. They were long-term forest or wetlands, with minimal influence from anthropogenic activities for the life of the vegetation or longer. Some of these sites may have been logged or cleared by early generations, but atmospheric inputs in New Zealand soils are relatively low, and for the most part these sites are regarded as being close enough to background to serve as a useful point of comparison. Sampling for chemical and biochemical indicators consisted of 25 soil cores (0-100 mm, 25mm diameter) over a 50 m transect, which are combined to form composites for analysis (Sparling *et al*. 2002). Sampling for physical indicators consisted of 3 soil cores (10-90 mm, 100mm diameter), which are individually analysed and the results averaged for each site.

Samples are analysed for an established set of soil quality chemical and physical parameters following Sparling *et al*. (2002) and for trace elements following EPA 200.2 (total recoverable metals hydrochloric/nitric acid digestion). Measurements were made at IANZ-accredited laboratories (soil quality chemistry at Landcare Research, Palmerston North, soil quality physical parameters at Landcare Research, Hamilton, aggregate analysis at Plant & Food Research, Lincoln and elemental analysis by ICP-MS at Hill Laboratories, Hamilton).

Results and discussion

Background measurements of soil chemical and physical soil quality indicators found in the Waikato region were identified and 95% confidence limits fitted (Table 1). The 95% confidence limits of pH, aggregate stability, Zn, Cu, Pb, As, Cr Ni, Cd and Hg fitted into clear categories when compared with current targets for soil quality parameters for forests.

Table 1. Average background concentrations of soil quality indicators in Waikato mineral soils.

Element	Average	95% CL	n	Current target rating
pH	5.2	4.5-5.7	36	Optimal[1]
Total C (%)	8.3	3.5-16.0	36	Normal to ample[1]
Total N (%)	0.45	0.18-0.91	36	Depleted to high[1]
Olsen P (mg/kg)	6.6	1.1-14.9	36	Very low to adequate[1]
Anaerobically mineralised N (mg/kg)	117	46-225	35	Adequate to excessive[1]
Bulk Density (t/ha)	0.69	0.41-1.01	35	Very loose to adequate[1]
Macroporosity (% at -10 kPa)	22	8-39	35	Low to high[1]
Aggregate stability (MWD, mm)	2.21	1.96-2.62	13	Optimal[2]
Zn (mg/kg)	29.9	11.2-57.4	26	Below guidelines[3]
Cu (mg/kg)	13.8	5.1-27.3	26	Below guidelines[3]
Pb (mg/kg)	10.3	3.1-25.6	25	Below guidelines[3]
As (mg/kg)	4.9	0.70-9.0	26	Below guidelines[3]
Cr (mg/kg)	4.56	0.75-10.7	26	Below guidelines[3]
Ni (mg/kg)	3.18	0.69-9.19	25	Below guidelines[3]
Cd (mg/kg)	0.12	0.04-.28	26	Below guidelines[3]
Hg (mg/kg)	0.11	0.05-0.27	26	Below guidelines[3]

[1] Sparling *et al*. (2003)
[2] Beare *et al*. (2006)
[3] NZWWA (2003)

Total carbon fitted across two categories but both these are considered as meeting soil quality targets.

Total N ranged from one extreme to the other. Too little total N in managed land uses restricts production due to N deficiency and too much increases the risk of N leaching. Too little labile N is not likely to be harmful in native systems as they are often adapted to low nutrient conditions. Too much available N however has been linked to forest degradation in northern hemisphere forests subjected to high atmospheric N deposition. Total N on its own appears inadequate to measure nitrogen status for soil quality assessment (Taylor 2009). Trends in total N may be more valuable than meeting an actual target.

Olsen P ranged from very low to adequate, supporting the view that New Zealand soils are regarded as being naturally low in P (McLaren & Cameron 1990). The very low rating is outside the soil quality target, but many indigenous forests are considered P limited (Parfitt *et al*. 2005) and it is possible that higher P levels could result in changes in competitive dynamics between species.

Anaerobically mineralised N ranged from adequate to excessive. Too little anaerobically mineralised N restricts production, while too much (excessive rating) indicates an increased risk of nitrogen transfer. However, anaerobically mineralised N was strongly positively correlated with total carbon ($R=0.696$, $p<0.0001$), which reduces the N leaching risk due to sorption of N. This result supports the view that high anaerobically mineralised N on its own is inadequate to measure the risk of nitrogen transfer to water or the atmosphere

(Taylor 2009). Trends in anaerobically mineralised N may be more valuable than meeting an actual target.

Bulk density ranged from very loose (low bulk density) to adequate and this property is strongly influenced by soil type (Table 2). The target range is met if bulk density is rated loose or adequate. Targets are not met if soils are rated very loose or compact (no background sites were rated compact). Soils of the Waikato region, generally, have low bulk density and these soils could have an increased erosion risk if cleared of vegetation. This risk is identified by the current soil quality targets for bulk density. However, it is debatable if low bulk density soils are non-desirable as these soils are in their natural state and have successfully supported the native vegetation for centuries. To say that these soils do not meet soil quality targets is, therefore, incorrect. The bulk density indicator is providing warning of the fragility of some of the soils of the Waikato region.

Table 2. Mean surface (0-100 mm) bulk density and macroporosity (-10kPa) measurements of different soil orders averaged over all land uses in Waikato soils

Soil Order[1]	Bulk density (t/ha)		Macroporosity (-10 kPa) (%)		n
	Average	95% CL	Average	95% CL	
Podzols	0.53	0.42-0.64	30	16-40	10
Pumice Soils	0.64	0.49-0.80	24	7-43	34
Allophanic Soils	0.70	0.49-0.88	16	4-36	56
Gley Soils	0.85	0.57-1.04	13	6-23	24
Recent Soils	0.87	0.71-1.04	17	6-47	9
Brown Soils	0.91	0.76-1.11	13	6-23	27
Granular Soils	0.97	0.72-1.32	15	2-28	17

[1] Hewitt 2002, New Zealand Soil Classification

Macroporosity had a very wide range of categories, from low to high. Also, like bulk density, macroporosity is strongly influenced by soil type (Table 2) and the two measurement are strongly negatively correlated (r=0.732, p=<0.0001). Targets are met if the macroporosity rating is low or adequate. Targets are not met if the rating is very low or high. Very low macroporosity inhibits soil aeration and plant root growth, while high macroporosity is indicative of an increased erosion risk, similar to low bulk density. A similar argument to that for bulk density applies. It is questionable if high macroporosity soils are non-desirable as these soils are in their natural state and have successfully supported the native vegetation for centuries. To say that these soils do not meet soil quality targets is, therefore, incorrect. Like bulk density, the macroporosity indicator is providing warning of the fragility of some of the soils of the Waikato region.

Conclusion

Background concentrations are easily applied to some measurements (e.g. metals) but for other soil quality measurements (e.g. bulk density and macroporosity) the concept is more difficult to apply. Background sites met the targets for pH, total C, aggregate stability, Zn, Cu, Pb, As, Cr Ni, Cd and Hg. Some background sites didn't meet the targets for Olsen P, as New Zealand soils are naturally low in phosphorous. Also, some sites didn't meet the targets for bulk density and macroporosity. However, it is questionable if these soils are non-desirable as they are in their natural state and have successfully supported the native vegetation for centuries. Targets for total N and anaerobically mineralised N on their own are inadequate to assess nitrogen status and risk of transfer to other parts of the environment. Trends in anaerobically mineralised N over time may be more valuable than meeting an actual target.

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Design-based and model-based sampling strategies for soil monitoring

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Abstract
This paper explains the fundamental differences between the design-based and the model-based approach for sampling. In soil monitoring four combinations of these two approaches are possible, a fully design-based approach, a fully model-based approach, and two mixed, design-based and model-based approaches. The choice between these four approaches is crucial in designing a sampling scheme for monitoring. Another important choice is the pattern type of the observations in space-time, differing in how many and when sampling locations are revisited. Five basic types are described. Two case studies are then described, and the choices of the statistical approach and the pattern type are motivated.

Key Words
Probability sampling, trend monitoring, compliance monitoring, validity.

Two fundamentally different sampling approaches
In sampling for soil *survey* two fundamentally different approaches can be followed: a design-based or a model-based approach (Särndal *et al.* 1992; de Gruijter and ter Braak 1990). In a design-based approach sampling locations are selected by probability sampling, and the statistical inference (e.g. estimation of spatial mean) is based on the sampling design. In a model-based sampling approach there are no requirements on the method for selecting sampling locations, and typically are selected by purposive (targeted) sampling, for instance on a centred grid. In statistical inference a model for the spatial variation is introduced, e.g. an ordinary kriging model, assuming a constant (unknown) mean, or a universal kriging model in which the mean is modelled as a linear function of one or more predictors. Besides the deterministic part for the mean, a kriging model contains a stochastic part describing the variance and covariance of the residuals of the mean. Note that the source of randomness is different in the two approaches. In the design-based approach the selection of the sampling locations is random, whereas in the model-based approach randomness is introduced via the model of spatial variation. In the design-based approach no such model is used. This has important consequences for the interpretation of measures of uncertainty about estimates, e.g. the variance of the estimation (prediction) error.

To quantify our uncertainty about estimates (predictions) in both approaches a chance experiment is repeated many times (not in reality but in mind). However, as the source of randomness differs between the two approaches, this chance experiment also differs. In the design-based approach the chance experiment entails repeated selection of sampling locations with a random sampling design, whereas in the model-based approach a long series of values is generated at all locations in the area with a model, i.e., a series of `fields' (model-realizations) is simulated. Note that in the design-based approach only one `field' is considered, being the 'field' actually sampled, and in the model-based approach only one sample is considered, being the sample actually selected.

Choosing between the two approaches is one of the most important decisions in designing sampling schemes (de Gruijter *et al.* 2006). Brus and de Gruijter (1997) discuss pros and cons of both approaches, and give rules for choosing. Broadly speaking, a design-based approach is the best choice if interest is (parameters of) the spatial cumulative distribution function (SCDF) for the area as a whole or for a restricted number of subareas, and besides validity of the result really matters (validity more important than efficiency). A model-based approach is the best option if interest is in a map depicting the values of many small areas, e.g. pixels, and we want to predict these values as precise as possible (efficiency more important than validity). We try to increase the precision by postulating a model, which may invoke discussions on the validity of the result, as several assumptions in modelling are made.

Sampling for monitoring
In sampling for soil *monitoring* one dimension is added, the time dimension. Besides *where* to observe the soil and at how many locations, we must decide on *when* to observe it, and how frequent. Similar to sampling

locations, sampling times can be selected either by probability sampling or by non-probability sampling, the former enhancing design-based statistical inference, for instance of (parameters) of the temporal cumulative distribution function (TCDF), the latter asking for model-based inference. Having two options for spatial sampling and two options for sampling in time, four combinations for sampling in space *and* time are obtained:

1. D_SD_T: both sampling locations and sampling times are selected by probability sampling, and statistical inference is entirely design-based
2. D_SM_T: sampling locations are selected by probability sampling, but sampling times are not. Inference involves both design-based and model-based inference. The model used in model-based inference is a time-series model
3. M_SD_T: sampling locations are not selected by sampling, but sampling times are. Design-based and model-based inference. The model now is a spatial model
4. M_SM_T: neither sampling locations nor sampling times are selected by probability sampling, and a full space-time model is used in the inference

The choice between these four statistical approaches is crucial in designing sampling schemes for monitoring. Another important choice is the type of sampling pattern in space-time. Several basic types can be distinguished, differing in how many locations and when sampling locations are revisited (Figure 1). In a *static* pattern sampling locations are fixed (static), but the observations are not synchronized. In a *synchronous* pattern the observations are synchronized, i.e., all locations are observed in short periods of time (sampling rounds). However, the sampling locations observed differ between the sampling rounds, there are no revisits. In a *static-synchronous* pattern in all sampling rounds the same sampling locations are observed, i.e., all locations are always revisited. Two compromise patterns in between a synchronous and a static-synchronous pattern can be thought of. In a *rotational* pattern, part of the sampling locations of the previous sampling round is revisited, the other part is replaced by new locations (sampling with partial replacement). In a *serially alternating* pattern no locations are revisited until a given sampling round, after which all locations of the first round are revisited *et cetera*. Note that the method of selection of sampling locations or times (probability or non-probability) is not part of the definition of the pattern types. In figure 1 locations are selected randomly, but sampling times are not.

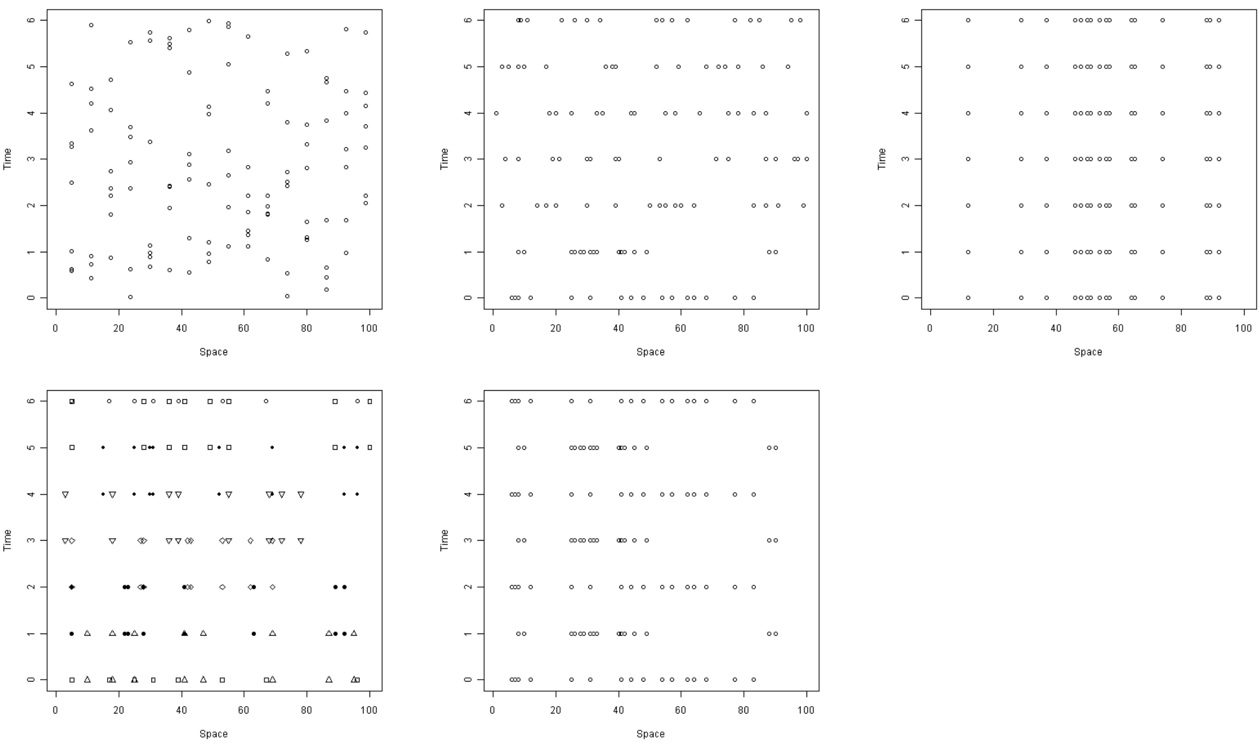

Figure 1. Basic types of sampling pattern in space-time. From top to bottom and from left to right: static, synchronous, static-synchronous, rotational (matching proportion 50%), and serially alternating (after de Gruijter *et al* (2006))

Case studies

I will illustrate now the above mentioned possibilities with two case studies, one on compliance monitoring of surface water quality, the second one on trend monitoring of indicators for soil eutrophication and acidification under forests. The choices with regard to statistical approach and pattern type that have been made will be motivated. In the first case study a fully design-based approach D_SD_T was chosen (Brus and Knotters 2008; Knotters and Brus 2010). The reason is that in compliance monitoring the quality of the result, being the conclusion whether the surface water quality complies with legal standards or not, must be beyond discussion. In other words, the validity of the result is very important. Moreover, interest was in a global target quantity, being the space-time mean concentration during a summer half-year. There was no need for spatial mapping of concentrations. As a pattern type a synchronous pattern was chosen, in which the sampling locations of a given round were selected *independently* from the locations of any other round. This independent synchronous sampling enables *design-unbiased* estimation of the sampling variance. For a static-synchronous pattern an unbiased estimator of the sampling variance does not exist. This is due to the two-fold alignment in space-time (Figure 1), i.e., the sampling locations of a given round are not selected independently from the locations of the other rounds, they even coincide with the locations of other rounds. This is entirely comparable to systematic random sampling in space (random grid sampling), for which an unbiased estimator of the sampling variance does not exist either. Both sampling locations and sampling times were selected by stratified simple random sampling. The stratification along both the spatial and the time axis improved the coverage of the space-time universe.

In trend-monitoring of soil eutrophication and acidification indicators a hybrid, design-based and model-based approach D_SM_T was chosen (Brus and de Gruijter 2010). So sampling locations were selected randomly, but sampling times were selected non-randomly, with the first sampling round at the start, the last one at the end of the monitoring period. The reason for selecting locations randomly is based on the chosen target quantity, being the temporal trend *of the spatial mean*. The available budget allowed for twenty locations per sampling round, which makes high-resolution mapping of temporal trends unfeasible. For estimating a temporal trend, random selection of times is suboptimal. For estimating a linear trend it would be optimal to do half of the total number of observations at the start, the other half at the end. However, this prevents us from checking for a non-linear trend. A rotational pattern type was chosen. This choice was somewhat arbitrary as we lacked knowledge of the relative efficiency of space-time pattern types for estimating the selected target quantity. However, we were reluctant to revisit in all sampling rounds twenty locations only, which would lead to a rather poor coverage of the space-time universe. To estimate the temporal trend of the spatial means, first the spatial means at the sampling rounds and their sampling variances and covariances were estimated by model-free, design-based inference. The estimated spatial means were then plotted against time. A linear trend was then fitted by generalized least squares. The covariance matrix used in GLS is the sum of the matrix with sampling variances and covariances of the estimated spatial mans and a matrix with model variances and covariances of the (errorless) spatial means. The hybrid approach enables quantification of the contribution to the total uncertainty about the trend of the sampling error in the estimated spatial means and of the model inadequacy error.

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Designing a soil pH monitoring network for the Western Australian wheatbelt

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Abstract
Soil acidification is recognised as becoming the major land degradation issue in Australia in the coming decades. It happens slowly, within a complex mosaic of variable soils and agricultural management, and it can take decades for related environmental consequences to become apparent. Monitoring efforts are needed to quantify the changes occurring and inform policy and management. We discuss the design of a soil pH monitoring network for south-western Western Australia. We describe the important criteria used in designing the network, present the methodology developed and site selection process, and highlight some of the challenges of carrying out a long term soil monitoring programme.

Key Words
Sampling design, monitoring, soil pH, acidification.

Introduction
The negative impacts of 60 years of intensive agricultural practices on soil condition are beginning to show in Western Australia, as in other parts of the world. At the same time, regional and global demand for food production is increasing, and climatic constraints on agriculture are predicted to increase (Hennessy *et al.* 2007; FAO 2009). Acidification of agricultural land in Australia is considered a larger risk to sustainable production than salinity (Government of Western Australia 2007), and declines in soil pH are expensive and difficult to reverse, making detection of the problem and proactive management paramount. Natural resource managers in Western Australia require a purposively designed soil pH monitoring network to identify areas at higher risk of soil acidification and to detect changes in soil pH through time. This monitoring network, in conjunction with field experimental work and acidification modelling, will identify trends and help shape state natural resource management policy and regional extension programmes, as well as directly inform industry and land managers.

The wheatbelt in southwestern Australia includes approximately 13 Mha under dryland agriculture, two-thirds of which are at risk of acidification (Government of Western Australia 2007). Key regional impacts of soil acidity include: lower plant yields and related profits, fewer suitable planting options, irreversible degradation of clays and colloids in the soil, further reducing fertility; accelerated erosion related to lower density groundcover; and worst case scenario, severe erosion preventing future agricultural production (NLWRA 2001). The processes that decrease soil pH in agricultural systems are removal of base cations from the soil system (generally through plant uptake and removal of vegetation, or leaching), and application of ammonium-based nitrogen fertiliser at rates in excess of plant requirements. Liming causes soil pH to increase, though its effectiveness varies with soil texture, type of lime, and amount applied. This combination of variable loss and gain complicates detection of long term trends in soil pH. Here we describe the design of a soil monitoring network to characterise the current status and future trends in soil pH for the wheatbelt of south western Western Australia.

Design criteria
To provide the most flexible framework for assessing soil condition, a monitoring network must be both as general and as representative of natural and human-affected conditions as possible but still meet the funders budgetary constraints. Careful planning is required to ensure that a relatively small sample of sites selected to form the network allows estimation of the general spatial and temporal trends in soil pH. A number of decisions are required to define the focus, required outputs and reliability of the monitoring programme.

The goal is to estimate the change in pH (ΔpH) over the wheatbelt as a whole, and within the wheatbelt with as much spatial detail relevant to regional planning as possible. Design-based inference was chosen so that an *unbiased* estimate of the mean ΔpH (not pH itself) across the whole wheat belt could be obtained. Using design-based methods also ensures the estimates of ΔpH made using the network will be adequate for regulatory practices, if required in the future (de Gruijter *et al.* 2006).

The processes causing soil acidification are slow, and the high spatial and seasonal variability in pH can make detection of trends difficult. Change is undetectable in less than five years even in the most severe cases of degradation, and can be more consistently determined over a ten year period (McKenzie and Dixon, 2006). This long term degradation process requires an equally long-term monitoring programme. The planned duration for the monitoring network is 20 years, and sampling will occur once every five years. The same sites will be sampled through time, helping to minimise the spatial variability between time pairs which can often confound interpretation of trends (McKenzie *et al.* 2002).

The final set of design criteria were:

- *Target population*: soils vulnerable to acidification (low buffering capacity) under rotational cropping, in the low rainfall region (< 500 mm/yr)
- *Target quantity*: the mean change in pH over the sampling interval
- *Reporting units*: the region as a whole, and mapped soil-landscape zones which reflect physiographic and cropping system differences in the region and help to partition variability in soil pH plus provide broad-scale spatial patterns in results
- *Quality measure*: High confidence (>95%) for the region as a whole of detecting a 0.2 pH unit change from the mean, and moderate confidence (70-90%) of detecting a 0.3 pH unit change within the subregions (26 soil landscape zones)
- *Logistics*: Must keep costs and labour requirements as constant as possible throughout the life of the survey (e.g. spread out timing of sampling); fix total sites for affordability
- *Flexibility*: Design network for acidification only, but allow flexibility to augment the system to monitor additional processes in the future; ability to adjust network in reaction to ongoing findings

Calculation of the number of sites needed

The sites need to be randomly selected (probability sampling) as we are using design-based inference (de Gruijter *et al.* 2006). The following equation, based on standard statistical sampling theory, was used to estimate the number of samples (n) required to detect the specified minimum detectable pH change (y) at a certain confidence level (100*(1- α)%) (Saby *et al.* 2008, equation 5):

$$n > \frac{2z_{\alpha}^{2}s^{2}}{y^{2}} \tag{1}$$

where z_α is the value of the standardized Normal distribution at probability α, and s^2 is an estimate of the variance of pH. This equation assumes that the soil is sampled from the same site on subsequent monitoring visits, the variability of pH does not change over time and that the correlation between two sampling times is small. Best current estimates of soil pH and its variability were used as very little reliable information was available on ΔpH and its variability in the region. Soils with low buffering capacity were targeted; within these soil types, the current estimate of pH gives an indication of those soils that are at greatest risk of falling below the recommended pH levels for cropping.

Table 1. Summary for the 0-10cm layer of pH variability and estimated number of monitoring sites required per reporting unit to detect changes of: 0.2 pH units with 95% confidence (N1), and 0.3 pH units with 85% confidence (N2). A subset of soil landscape zones (8 of 26) shows how N changes with pH variance.

Reporting Unit	Count	Mean	Variance	N1	N2
Wheatbelt	42585	5.18	0.449	86	37
Zones					
253	4008	5.00	0.126	24	6
243	719	5.02	0.220	42	10
220	227	5.66	0.273	52	13
256	7989	5.15	0.319	61	15
258	12111	5.19	0.417	80	19
250	985	5.27	0.689	132	32
221	87	6.41	0.955	183	44
246	2668	6.01	1.191	229	55
Total for 8 of the 26 zones:				803	194

A commercial pH dataset (Precision Soil Tech (Perth WA) unpublished data) was used to characterise current pH status. Samples were collected between 2006 and 2008 and consist of 10 cores at 0-10, 10-20, and 20-30 -cm depths sampled in an 8-m arc, bulked, mixed and subsampled by layer (Gazey *et al.* 2007). The sample count, mean, and variance for each reporting unit are shown in Table 1, along with the estimated number of sample sites required for different levels of detection (see caption). The funding available dictated approximately 400 sites in total could be sampled; the programme goals were met with this total by distributing the sites across soil landscape zones using a detectable change of 0.3 pH units, and 85% confidence. The 400 sites will permit estimation of a 0.2 pH unit change for the wheatbelt as a whole with 99.9% confidence.

Selection of monitoring sites

Once the number of sites was determined by reporting unit, the highest resolution soil map unit polygons (Schoknecht *et al.* 2004) were randomly selected for sampling, weighted toward polygons with larger areas of vulnerable soils (probabilities proportional to size: de Gruijter *et al*. 2006). Areas outside the target population were masked out (e.g. urban areas, remnant vegetation, saline areas, water bodies). The masked, selected polygons were overlain with property boundaries. If only one sample was required per polygon, then the property with the largest target area was selected. If more samples were allocated to the polygon, or the first property owner refused access, then the property with the next largest area was selected, and so on. A list of property identifiers and percent coverage of vulnerable soil types in the map unit polygon was prepared for the field surveyors. The exact location of the site is determined in the field.

Within site sampling

The soil monitoring site is defined as a plot of land 25 x 25 m, following recommendations in McKenzie et al (2002). The within site sampling method is as follows: The south-west corner of each site is georeferenced with a standard GPS, and the sampling grid is laid out from there. Each grid cell is 5x5m (= 25 sampling cells). Each grid cell is sampled at four depth intervals, 0-5, 5-10, 10-20 and 20-30. Total number of samples per site = observations x depths = 25 x 4 = 100. The samples will be bulked by depth, split for laboratory analysis, and the remainder archived. The site and the soil profile will be described according to MacDonald *et al.* (2009) and the soil classified according to the WA classification and the Australian Soil Classification. Sampling will be done during the dry summer season when pH is most stable, and to reduce the seasonal signal year to year (Conyers 1997). In addition to pH, soil carbon and bulk density will be measured at least for the first 5 years of the programme. The soils targeted as vulnerable to acidification are unlikely to be those most vulnerable to changes in soil carbon, nevertheless they will contribute to a more complete assessment of current soil carbon status in WA.

Sampling strategy

The number of sites (approximately 400) and time interval for sampling (5 years) were determined, but sampling all sites in a season then waiting 5 years to repeat the exercise was considered too difficult to maintain continuity of funding and skilled personnel. Instead, a rotational sampling programme was planned in which the same sites are resampled at the same time interval, but a subset of the sites are visited each year. The selected sites were divided into five panels, and each year one panel will be monitored, with a repeat visit in 5 years time. Having some sites sampled in consecutive years ("partially augmenting" the sample) connects the panels as for an experimental design, and increases the power for trend detection. In fact, nearly equivalent trend detection results are obtained with fewer repeat visits to sites using this type of partially augmented rather than an "always revisit" plan (Urquhart and Kincaid 1999). The design as a whole is a 5-year period partially augmented serially alternating rotational pattern (Urquhart and Kincaid 1999; de Gruijter *et al.* 2006).

Will the number of sites really be adequate?

Generally speaking, more sites improve the precision of estimation, but the number of sites in the network must be balanced with affordability. Whether or not a smaller network would have been adequate was of real concern, particularly when ten years of data must be collected before the first results on ΔpH are available. We deliberately over estimated the number of sites required in the following ways:

- Data used to calculate number of sites: Used all currently available pH data even though we were only interested in data for vulnerable soils, so the variance was likely higher than that of the target population. Lower variance means fewer sites actually needed to meet quality criteria.
- Equation used to calculate sites: Equation 1 assumes no correlation between sampling times, whereas we would realistically expect a positive correlation, reducing the estimate of s^2 thus reducing the number of samples required. This was shown to be true for a small catchment within the wheatbelt with appropriate data where the correlation over 7 years was approximately 0.3.

- Distribution of change: The number of sites required were calculated for a two-tailed distribution, but primary interest is in detecting negative trends (lower tail only), which would reduce sample requirements.
- Minimum of ten samples: All reporting units were assigned a minimum of 10 sites to assist in assessing the variability in ΔpH once two sampling rotations are completed.

After each sampling season, the variability in pH for every reporting unit will be recalculated; if after the first full sampling (5 years) the variability is different from that calculated from the existing database, then sites will be added or eliminated from the programme without negative impacts on the survey results.

Conclusion

We have illustrated the steps involved in designing a long-term, spatially extensive monitoring programme to assess changes in soil pH in Western Australia. The network design chosen was the most efficient scheme to answer specific questions about particular reporting units within the practical limitations set out. However this design was based on a target population of soils vulnerable to acidification which will limit its ability to answer future questions on other soil properties or soil functions. As this network was designed using design-based methods it will be flexible enough to extend the target population to all soils and add sites to the network to address future unforseen questions if funding becomes available.

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Determination of Soil Erosion Using Laser Scanners

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Abstract
Recent advances in laser scanning techniques have allowed for a wide variety of applications and therefore the adaption of laser scanners (LS), both airborne laser scanners (ALS) and terrestrial laser scanners (TLS), is increasing in many science disciplines. Soil erosion is not an exception, where advances in soil erosion detection and measurement techniques are a crucial step in the reduction of errors and uncertainties in soil erosion models. In this paper, topographic datasets captured by different laser scanning devices were tested in order to quantify the soil erosion by water using different software-types. An experimental protocol based on iterative processes of data capturing and processing on field plots are the essential points of the methodology. In the selected experimental site, five plots were defined and scanned before and after removing manually some soil volume, as would occur during any soil erosion process. Later on, the eroded materials were weighted at the laboratory and their volume calculated considering their bulk density. The scanned dataset (i.e. point cloud) was also adjusted and calibrated at the maximum possible resolution in relation to the device capacity (i.e. accuracy and spot size divergence). Subsequently, soil erosion in each plot was calculated with the available software-types and both laser-calculated and manual-weighted results were compared from each scan using different software-types. Results revealed that soil erosion measured with laser scanning techniques is good when adequate calibration at adequate spatial resolution is performed. Moreover, the combination of hardware and software has led to a variety of results which highlight the importance of the algorithm used by each software-type. Furthermore, soil erosion measured with TLS vary considerably in relation to the software used, and thus the values reported without indication to the software might be doubtful and should be used with caution in hydrological modelling.

Key Words
Laser scanners (LS); Terrestrial laser scanners (TLS); bulk density; point cloud; experimental protocol.

Introduction
In general, and in order to quantify soil erosion by water in the field, different methods are often used, e.g. profile-meter, erosion pins, runoff-erosion plots, etc., according to the dominant hydrological processes and/or their spatial distribution. Methods based on sediment quantification collected by automatic samplers coupled to gauging stations have severe inconveniences when hyper-concentrated flows clog the gauging devices and sensors like those in badlands (Solé-Benet et al., 2003). Whereas classic methods based on microtopographic variations (erosion pins, profile-meters) are precise at local scale, the extrapolation of their results to large scales (i.e. up-scaling) implies errors and/or uncertainties. In this work, we rehearse a non-invasive technique, based on micro-topography surveying by laser scanning techniques (Buckley et al., 2008), scantly used in soil erosion quantification. Precisely, different types of Terrestrial Laser Scanners (TLS) and different software have been tested for their suitability in the volumetric quantification of soil material (e.g. regolith or rock) exported by water erosion.

Materials and Methods
The following TLS types (and makers between brackets) have been used: ScanStation 2 (by Leica), Ilris-3D (by Optech), and LS-800 (by Faro). The datasets captured by these devices were analysed by 5 distinct software-types: Polyworks, I-Site Studio, Cyclone, Faro-Scene and JRC-3D- Reconstructor. The Faro TLS device uses the Phase-based measurement principle, which is a priori more precise but with less range than the other devices that use time-of-flight technology. All the experimental work was performed in the same selected sector of a hillslope in the Tabernas Desert badlands (figure 1). It was a bare marly area with some lichens (Canton et al., 2004) about 30 m x 30 m, with 20º slope.

Experimental procedures
Once the experimental site was georeferenced by specific fixed targets, a first scanning was performed with an horizontal resolution (x,y) of 5 mm (grid spacing), in order to obtain the point cloud used in the generation of

the first Digital Surface Model (DSM). This relief surface forms the base surface level and was named the "a" surface. Later on, all DSMs are filtered in order to generate the Digital Terrain models (DTMs) used to construct the terrain surface using the Triangulated Irregular Networks (TINs). It is important to underline that all scanning processes were carried out from the same reference point, prepared initially in order to minimize the differences between devices due to local factors, e.g. slope, distance, divergence of the laser spot, etc.; that is, equal conditions for all TLS. Next, an artificial (i.e. manual) erosion process was done in five plots using chisel, hammer and shovel, extracting every time between 1 and 2 L of soil or regolith material. The location of plots was chosen to represent variable conditions of the topographic relief (steepness, roughness of surface cover, soil humidity) (Figure 1). The manually-exported materials were stored in bags for further weighting at the laboratory. The extracted volumes were defined in relation to the variable bulk density (BD) of the site, which varies between 1.2 to 1.4 kg/L (measured with both the excavation and the cylinder methods). Consequently field measurements had an error of ± 0.12 kg/L. A second scanning was performed at the same resolution and, once processed it generated the second surface "b". Therefore, the difference between the two surfaces ("a" and "b") was assumed to provide a good estimation of the extracted volumes (i.e. eroded plots). This experimental protocol was repeated with every TLS instrument. The generated surfaces (i.e. TINs) by different software-types have provided different values for each plot based on the algorithm used to generate the TIN surfaces and the total number of points in the dataset (i.e. point cloud).

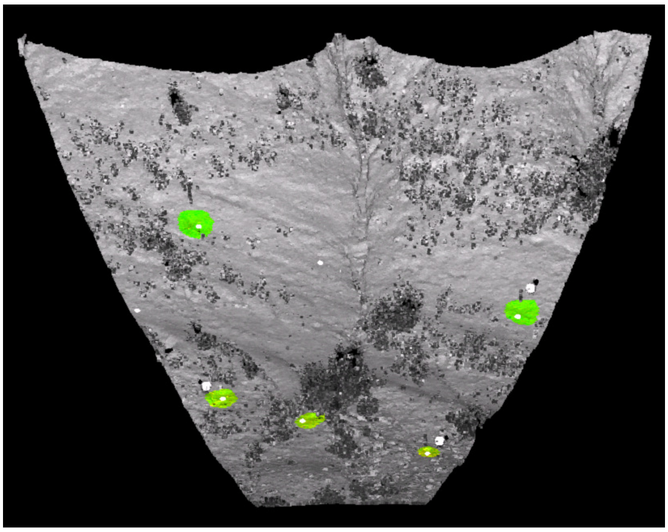

Figure 1. Map of the study area,about 30 m x 30 m, compiled by a TLS cloud of points. Manually-eroded plots are marked in green.

Results and discussion

The TLS measurements generate four types of major error: *i*) errors of the TLS itself, which is related to the TLS maker; *ii*) errors related to the horizontal resolution of the scanning process; iii) errors related to the treatment and filtering process; and *iv*) errors related to the algorithm used by each software in the TINs generation. In general, measurement errors produced by different scans of the same instrument (hardware) on the same area and under the same conditions of processing and treatment are minimal, and could be neglected. However, the different algorithms used by distinct software to generate the surface TINs from the same point cloud could lead to significant variations that should be evaluated.

Results revealed that both hardware (TLS) and software types are of paramount importance: results from each instrument vary in relation to the software used, and vice versa. Figure 2 shows the results of the eroded volumes measured and calculated from two devices, ScanStation 2 and Ilris-3D, and 4 types of softwares (Cyclone, Polyworks, Reconstructor, and I-Site). These results reveal a significant variation between real measured volumes (weighted in laboratory) and calculated ones (i.e. defined TINs). In figure 2, the green line of each curve, which represents the calculated values (TINs differences), highlights the clear variations for each instrument and software-type. Theoretically, the best approximation for TLS calculated volumes are achieved when all the points in the green line are located between minimum and maximum laboratory-measured values; this best fit was not achieved in any of the plots. However, the combination ScanStation 2 - Cyclone (figure 2 a) has three points between minimum and maximum values, indicating a good approximation to the laboratory-measured volumes. On the other hand, the same point cloud generated by the same LS device but processed by another software produced moderately different results (Figures 2 c, d, e and f). It is worthy to underline that all the devices and software-types are quite close to real eroded data, indicating the high capacity and strong potential of this new technology.

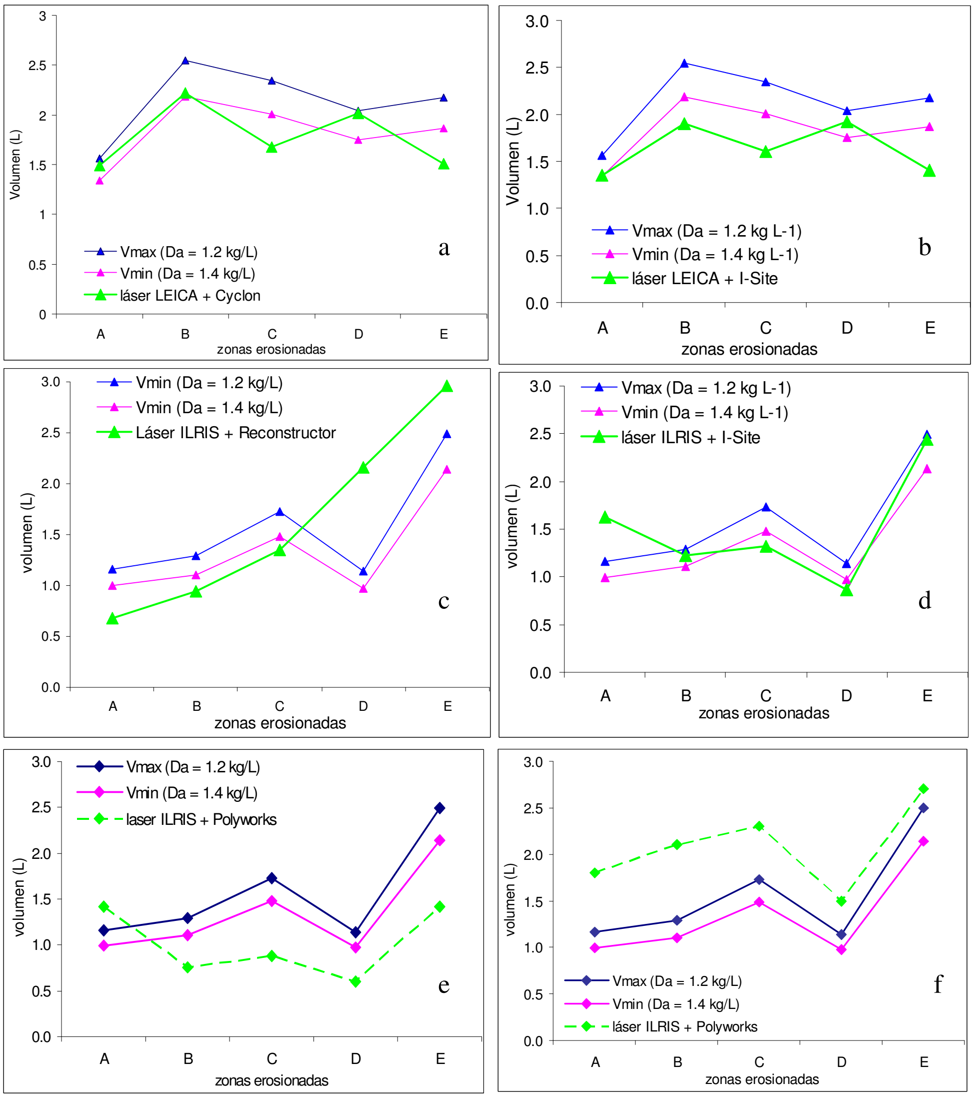

Figure 2. Volumes measured and calculated by different TLS and software-types (green dots linked by discontinuous lines). Blue and red lines dots and lines represent maximum and minimum laboratory-values.

Conclusions

Laser scanners offer a highly effective method for collecting massive volumes of precise, high-resolution 3D information for microtopographic detection, and hence surface variations. Monitoring processes of soil erosion using TLS techniques in specific parts of the landscape susceptible to erosion processes is faster and more precise than other topographic methods. Without any specific pre-evaluation of appropriateness or suitability of the different tested devices, it is possible to emphasize that all TLS achieve a sufficiently dense point cloud to generate a TIN that fit well (with a good precision and accuracy) to minimum landform surface details. However, some combinations of hardware and software achieve better results (better fit to real data) than others. Results revealed that datasets obtained from the same device (LS) but modelled by different software-types are slightly variable, which highlights the importance of the algorithm used in the surface construction process (i.e. triangulation process), a crucial step in the volumetric calculation of the eroded surfaces. When coupling TLS microtopographic data with detailed soil surface mapping in terms of soil type, cover type, etc. the TLS could be an indispensable tool for studying detailed mechanisms of soil surface erosion. Finally, due to its measuring accuracy, its high point density, and its measurement speed, TLS increasingly represents an alternative to and/or an additional option for traditional methods for data capturing and soil erosion detection.

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Developing methods to detect change in the soil resource of a country with a Northern, Temperate Boreal Climate (Scotland).

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Abstract
Many nations and regions are currently developing soil monitoring schemes to assess the status of their soil resource. In Scotland, we have used an existing national scale, objective soil sampling scheme to assess different methods of sampling soils for the purposes of monitoring change, to assess new soil indicators (including indicators of physical soil quality) and to develop novel, rapid methods to assess changes in the soil resource. Early results have shown the value of archived soil material in detecting change over a prolonged (up to 30 years) period of time and indicate that the ability to detect change is influenced by sampling method and analytical methods.

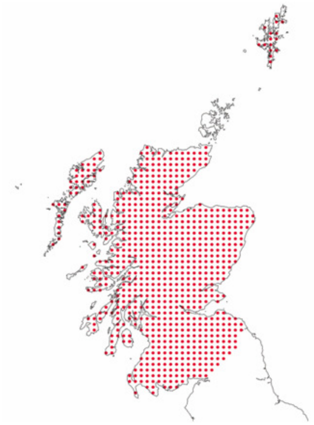
Key Words
Monitoring, soil indicators, carbon, sampling

Introduction
Soils are under threat from economic, social and biophysical drivers. Soil monitoring schemes are being designed to determine the magnitude and direction of any change in order to more effectively manage the soil resource. However, detection of change is not straight forward. In Scotland, we are using an existing national scale sampling scheme to test three soil monitoring sampling systems currently in use in the UK. In particular we aim to determine their scientific and technical suitability, logistics and ability to detect change out-with normal variation in soil properties. Further, the re-sampling scheme will be used to determine suitable soil indicators for soils developed under a northern, temperate, Boreal climate.

Methods
From 1978-1988, the soil profile at 721 sites on a 10km grid (Figure 1) were described, sampled and analysed for a wide range of soil properties. This National Soil Inventory of Scotland (NSIS) provided an objective sampling scheme of a wide range of Scottish soil types and habitats. Re-sampling the NSIS is ideal for testing methods to determine changes in soil properties over time. In particular we aim to:

1. determine evidence of change in C content and macronutrients
2. compare sampling methods such as:
    - fixed depth sampling vs traditional sampling by soil horizon (pedological)
    - point sampling vs composite sampling over an defined area
3. measure new soil attributes to test their suitability as indicators of soil quality
4. develop and test new methods for assessing soil quality
5. assess the value of archival material in the detection and interpretation of change

In order to maintain this objectivity, we decided on a sampling scheme that allowed us to resample 25% of the original inventory over a 3 year period and was based on a 20km grid pattern aligned to the British Ordnance Survey grid (Figure 2). However, using a 20km grid meant that there were four possible sampling grids. In order to test if any of these would bias the results, specific soil and environmental attributes such as carbon content (Figure 3), base status and slope were examined for each of the four potential grids using data from the existing 10km NSIS dataset. Statistical analyses verified that there was no significant bias inherent in any of these grids. The final grid selected for the 2007-2009 re-sampling (NSIS2) had sampling locations at 20km intervals from the Ordnance Survey false origin used in UK topographic maps and has sample locations on the main outlying islands (Figure 2).



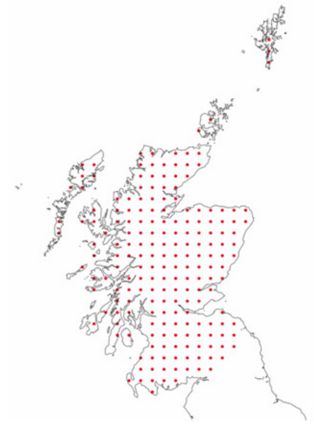


Figure 1. Original NSIS1 10km grid

Figure 2. 2007–2009 NSIS2 20 km grid sampling sampling design. design.

Prior to implementing the sampling programme, strict protocols were written and field- tested to ensure compatibility between each of the lead surveyors. All samples were taken in spring to reduce the inter-annual variation in soil biological properties and each sampling team began by sampling the lowland sites and gradually moving to the more upland sites so that as many soils were sampled before the onset of the growing season as possible. Navigation to the site was undertaken by experienced soil surveyors using both new and original air photography to relocate the original sampling sites. Certain site characteristics such as slope, aspect and soil type had to match the original records (within acceptable error limits) before the site was deemed to be correctly located.

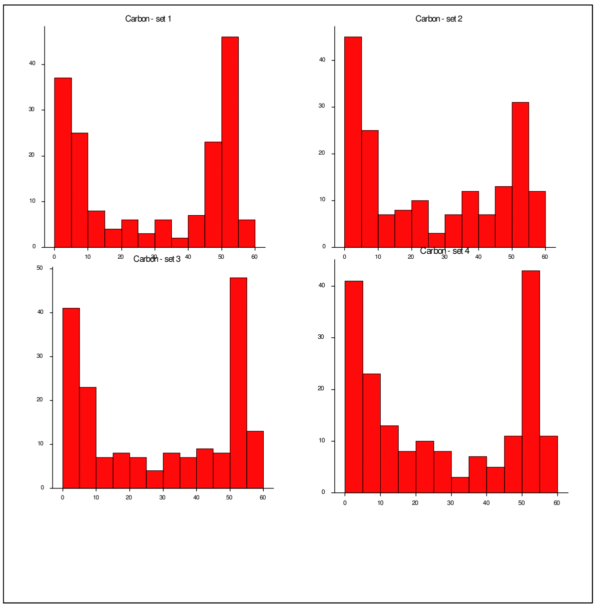


Figure 3. Frequency Distribution of topsoil carbon contents for each of the 4 possible 20km grids sampling schemes.

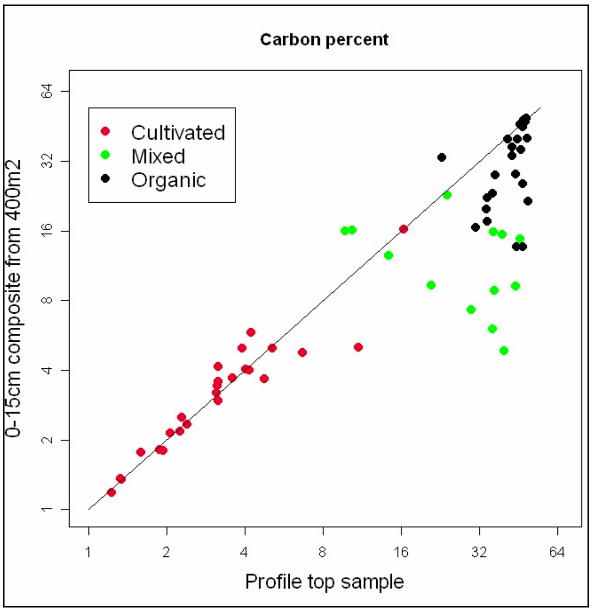


Figure 4. %C in cultivated, mixed organo-mineral and organic soils as determined from 0-15cm composite samples (y axis) and from profile bulk top horizon sample.

At each site, three main sampling methods currently in use in the UK were tested:

- Pedological horizon-based sampling
- Single, central 0-15cm core
- Composite auger samples 0-15cm

A soil profile was excavated to between 80 and 100cm and each of the main horizons were sampled by taking a bulk, disturbed sample from approximately a 10cm band around the mid point of each of these horizons. A single, centrally positioned vertical core was taken from the surface to a depth of 15cm and a composite 0-15cm auger sample was taken at 3-5m interval within a 20m grid centred on the main pit.

Triplicate samples to determine bulk density were taken from all soils and horizons. Cores to determine moisture retention and clods to determine aggregate stability were taken from mineral soils only. At each site there was also a sample design to assess short range variability in soil properties in order to help determine if detected changes are real or simply due to the inherent variation in soils. This comprised a further 4 smaller profile pits at varying, randomly set distances (up to 16m) from the main pit and aligned to the four cardinal compass points. Here, bulk density and bulk samples were taken from the uppermost, main horizon.

Results

Table 1 shows the number of samples collected over the three year field campaign and the attributes to be measured and evaluated. The soil samples are being analysed for the same range of properties that were determined in the original sampling scheme (%C, pH, micro and macronutrients, %N and particle size, base saturation) in order to determine if change has occurred and if this is within the local site variation. This latter component is vital to ensure that false results are not reported.

As well as analysing the recent samples, the archived soil material from the original sample scheme has also been re-analysed in order to minimise the effects of changes in analytical techniques and machines. This is a key component in the detection of change as early results appeared to indicate that carbon values were declining in Scottish soils, however, re-analysis of the archival material alongside the recent soil samples, showed that this change was not significant.

A number of soil indicators not currently in use in the UK are being assessed including Least Limiting Water Range (LLWR), aggregate stability and more novel, rapid methods of assessing various soil properties through the use of XRD and NIR techniques. These methods are quick to implement and, if suitable and robust regression equations can be developed between these attributes and attributes that are more costly and time consuming to measure, they may be useful as surrogate indicators that can be applied quickly and cheaply to a large number of samples.

There is also a major programme to develop methods to assess biological indicators and we are currently undertaking DNA analyses of the recent soil samples. Extracted DNA is also being archived at -80°C, providing a resource that can be utilised in the future, for example, as new assays are developed or as new questions are raised.

Our initial comparisons of the three sampling techniques have indicated that, for organo-mineral soils with thin (<15cm) organic layers overlying mineral horizons, composite auger sampling at 0-15cm produces results with too great a range to be of value in soil monitoring although this method may be suited to monitoring in homogenised cultivated horizons (Figure 4). Its applicability to monitoring in organic soils with >15 cm of organic material is still questionable due to the lack of information of overall thickness of the organic material.

Conclusion

Field sampling on a 20km grid has resulted in over 1400 disturbed, bulk soil samples from a wide range of Scottish mineral and organic soils that have been collected using strict protocols. Preliminary analyses have shown that archived soil material is invaluable to substantiate any apparent changes in soil properties over time. It is also becoming apparent that some sampling methods are not suitable for all soils. In Scotland, we are taking a rigorous approach to the analyses and the interpretation of results that inevitably extends the period between sampling and result reporting. However, despite the pressure from policy makers in governmental organisations and other stakeholders, it is vital that the results are robust and defensible.

Table 1 – Sampling of NSIS2: Indicators in **bold** are those suggested by UK-Soil Indicators Consortium as useful indicators of soil quality in UK.

Method/sample	**Number of samples**	**Analyses undertaken**	**What this will tell us?**
Pedological horizon-based sampling	701	LOI, **pH**, **%C**, **%N**, particle size, **macro** and **micronutrients**	a) Assessing changes in soil C content and stock. b) Assessing the effect of restrictions of sulphur emissions on soil pH or base status.
		XRD, FTIR, alkanes	c) High resolution functional analysis – new and cheaper methods to measure indicators. d) Assessing accuracy of site relocation. e) Have organic fractions/land use changed?
Bulk density	2087	**Bulk Density**	f) Determining carbon stocks and volume of available nutrients. g) Assessing soil compaction
Moisture release	198 (mineral soils only)	Least limiting water range (LLWR)	h) Determining the ability of soil to supply water for plants
Aggregate stability	132 (mineral soils only)	Aggregate stability	i) Determining susceptibility to erosion
Topsoil variability	725	LOI, **pH**, **%C**, **%N**; **bulk density**	j) Are detected changes in these properties within site variability?
Single central 0-15cm core	183	LOI, **%C**, **%N**, **pH, bulk Density**	k) Comparison of sampling methods - single core vs Horizon-based sampling or composite sampling
Composite auger samples 0-15cm	183	LOI, **pH**, **%C**, **%N**	l) Comparison of sampling methods - composite sampling vs Horizon-based sampling or single core
Soil/air interface core 0-5cm	183	**pH %C**, **%N** and **Metals**	m) Atmospheric pollutant concentration in near surface sample

Erosion processes on intensively farmed land in the Czech Republic: comparison of alternative research methods

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Abstract
Soil erosion and deposition of sediments are natural processes caused, in CZ conditions, by water and wind. However, these processes are also increasingly affected by human activity in the countryside. Research in the cadastral area of Čejkovice in South Moravia, CZ, where typical chernozem is the dominant soil type, showed problems typical for large areas of such soil type and similar farming intensity. To evaluate the extent of erosion two alternative methods were used – digital visual image interpretation and classification of areas of erosion from aerial photographs, as well as measuring radionuclide ^{137}Cs in soil samples. Before using these methods a model of potential erosion within the area was created, using the Universal Soil Loss Equation (USLE). The outputs of the research were compared in terms of exactness, general suitability and dis/advantages of different approaches. The application of remote sensing data, together with a field survey, seems to be very prospective and effective.

Key Words
Erosion, aerial photographs, visual image interpretation, image classification, radionuclide ^{137}Cs, field survey.

Introduction
Soil erosion and deposition of sediments are natural processes caused, in CZ conditions, by water and wind. However, these processes have also been increasingly accelerated by human activity. In the Czech Republic erosion is a serious problem: about 50% of arable land is endangered by water erosion and 10% by wind erosion (Janeček 2007; Šarapatka *et al.* 2002). Soil is a non-renewable natural resource; therefore its degradation due to surface erosion has far-reaching environmental and economic consequences. Moreover, the extent and impact of erosion is not just a local problem, neither is it a problem of a few recent years. It often means irreversible changes, taking place over decades, without adequate solution. Besides degrading the soil, erosion also reduces soil fertility and crop yield, causes pollution in both surface and ground water sources and increases sedimentation of loosened material in fields, alluvial meadows, river basins and water reservoirs where it contributes to water eutrophication and contamination (Zapata 2002).

Methods
Čejkovice area in South Moravia, CZ, was chosen as the pilot area for studying water erosion, its dominant soil type being typical chernozem which reflects the problems characteristic for large areas of such soil type and similar farming intensity.
The following methods were used to evaluate the extent of erosion:
A widely used method of modelling potential erosion, using the USLE equation (Wischmeier and Smith 1978). Processing the input data from the studied area was simplified by using geoinformatic methods. Commonly available commercial ArcGIS software was used for a GIS model of potential erosion.
As an alternative approach, digital visual image interpretation and classification of areas of erosion from aerial photographs. Visual evaluation of photographs is based on vectorization, i.e. drawing borders of eroded areas directly in the software environment. Aerial photographs were used in mapping earlier, but only the development of IT and geographic information systems enabled relatively fast and exact analysis of this type of data from Earth remote sensing (Fulajtár 2001). Preparation of photographs, orthorectification, creating seamless orthophoto mosaic and image analysis (unsupervised image classification) were carried out in ERDAS IMAGINE program. Another, alternatively used method was that of field survey – measuring the amount of ^{137}Cs radionuclides from soil samples in an transect, perpendicular to the slope affected by erosion. The estimation of ^{137}Cs redistribution is generally based on comparing the measured values of this radioisotope (total activity per unit area) in a certain sampling point (soil sample) with reference value representing cumulative atmospheric fall-out on the site (Zapata 2002).

Results

In this part we present only two research methods, i.e. visual image interpretation and classification, and the method of measuring radionuclides, both of which we have compared with the classical method for studying erosion.

Visual image interpretation and classification
In an aerial photograph, an erosion-disturbed soil profile is shown as a set of light areas of variable intensity on a darker background (on extreme sites even the opposite). The main problem of such a method can be in the vegetation cover (crops) of the area. Visibility can be improved by e.g. adjusting contrast. Another option is a combination of a higher number of time-spaced images where the affected areas are plant-free. The presence of special vegetation such as vineyards, orchards or hop-fields are specific to fertile warm areas where the affected areas can be adequately detected by manual-visual interpretation. Despite being time-consuming, non-objective and having deficiencies, this technique is quite exact. Its non-objective character is the result of the spatial complexity of the erosion process, creation of colluvisols and the limited possibility for an exact interpretation of the colour shades with the human eye. Defining non/eroded areas can be subjective, especially on a large-scale map.

Automation of several phases is an advantage and, in some respects also a disadvantage of the process of undirected image classification (Lillesand *et al.* 2007). This often means that areas are marked which have nothing to do with erosion, the operator cannot directly intervene in the process. However, objectivity of results is an undoubted advantage – the results should be identical when using identical software data and classification. Generally, the exactness of the identification, compared with visual interpretation, is of a very good standard. A combination of field survey, consideration of topography – a digital terrain model and the use of analytical tools of directed image classification can further improve - thus various stages of degradation could even be distinguished. This could eliminate the afore-mentioned disadvantages of interpretation. Another advantage of automation is in rapid processing, the option of working with large areas of land and practically excluding subjective evaluation of data. An exact detection of spectral characteristics (colour shades) of every single pixel of an image is another plus as this is not possible in other methods.

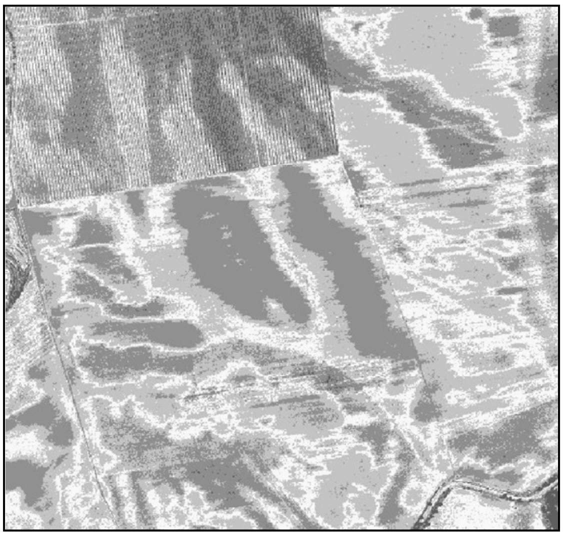 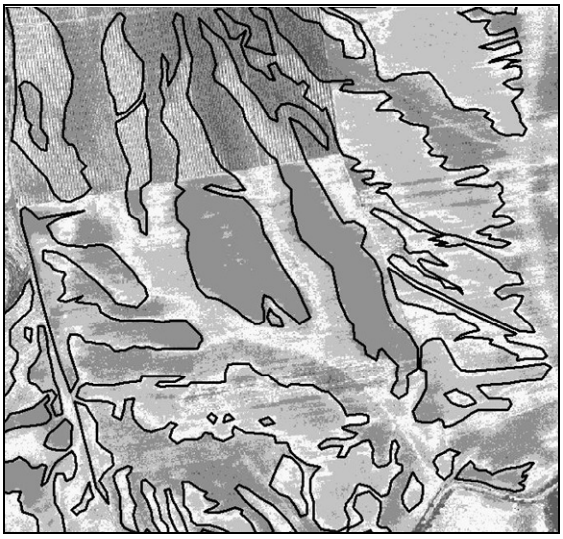

Figure 1 and 2. A layer of land analysed via unsupervised classification; right: projection with a visual interpretation vector layer.

Radionuclide measurement method
Large scale measurement of radionuclides is hardly feasible and generally demanding, from the sample-taking phase to evaluation of results. Creating erosion maps is also more difficult in comparison to classic field survey (Fulájtar 2000). The undeniable advantage is in the possibility to monitor soil erosion dynamics, both in time and in the soil profile. This allows the realistic estimation of the cubic volume of material moved by erosion.

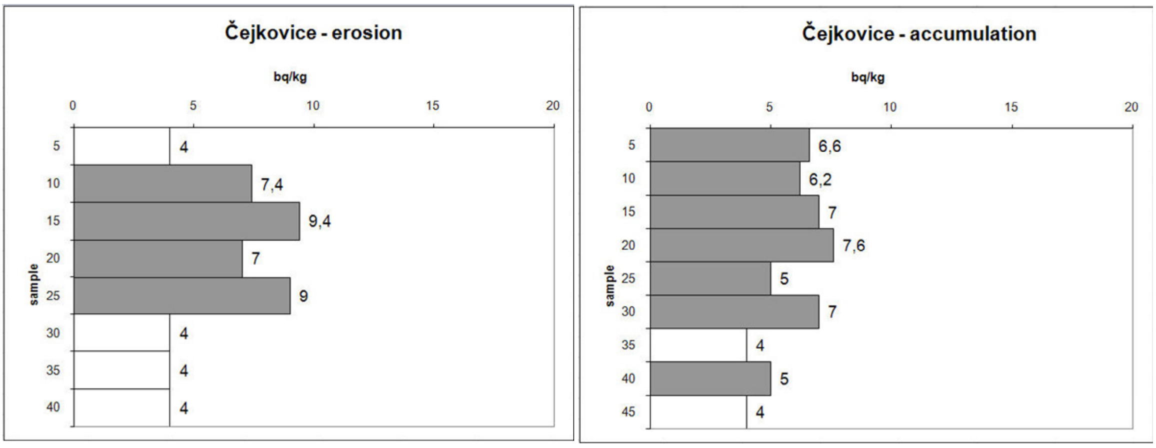

Figure 3 and 4. The distribution of Cesium in the soil profile confirms the advanced stage of the colluviation process (erosion), when the older layers without measurable radioisotope content cover younger layers with traces of ^{137}Cs deposited within about 40 years.
An accumulative sample shows increased surface horizon and reduced radionuclide content in the soil profile due to erosion and consequential deposition of sediments (soil).

The results provide a relatively reliable estimation of the extent of soil redistribution within the last 40 years. The computerized data obtained is compatible with current mathematic models, GIS applications and geostatic methods dealing with loss or accumulation of soil (Harmon and Doe 2001). This method combined with a field survey allowed the verification of results of the methods described above. The main goal was to consider alternatives of field work with the real state of soil.

Conclusion

Knowledge of the spatial layout of eroded areas is an important condition for effective soil protection, particularly against the effects of water erosion in an intensively farmed landscape. Still, relevant information on such a degradation process is often not mapped in detail. The research showed that the classic research methods can be successfully supplemented with results of the afore-mentioned methods, especially visual interpretation and classification of aerial photographs. Application of remote sensing data is very prospective, although a certain amount of subjectivity in processing such data (vectorization) must be resolved, as with the final specification of results of automated unsupervised image classification. Both approaches are being developed at present and their deficiencies should be suppressed while the spectrum of advantages is developed. The results of all so-called GIS methods can be verified in detail and compared to field survey supplemented by soil sampling and consequential measuring of ^{137}Cs radioisotope content, on the basis of which the erosion dynamics can be evaluated throughout the soil profile and partly in a time scale.

Acknowledgement

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Evaluating the effects of land management systems on soil characteristics: some confounding problems in experimental design

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Abstract
A series of research projects over the last decade have had some unexpected outcomes, at least partly from the effects of confounding by uncontrolled variation in factors outside the experimental design. The most pervasive is the overarching effect of the land manager, their paradigm, commitment, motivation, competence and application. To compare land management systems, most experimental designs compare two or more adjacent or close by operations, and take great care taken to standardise the non variable biophysical factors (soil types underlying geology, etc) and system factors, (seed, fertiliser regimes, tillage, livestock type and number, overall grazing rates, etc), but how does the researcher standardise for land manager variation? Even fence line contrasts between systems on the same landholders land suffer from the attitudes and motivation of the land manager. The other, and an equally difficult issue to control relates to seemingly inconsequential differences not considered in the experimental design. For instance the contrast of two or more land management systems that examines the effects of grazing rotation, but the researcher finds that the ways in which grazing pressure are calculated is by different means; or a land management system contrast where a paddock in one system is cut for hay and in the other is grazed. In recent research all these effects have been felt and on some occasions the experiment was unable to be used to come to a sensible conclusion because of the confounding effect of uncontrolled variation.

Key Words
Land management systems, experimental design, confounding factors.

Introduction
The increasing interest in carbon (C) sequestration and the ever escalating political debate on the role of agriculture, not to mention inclusion or exclusion in a future protocol has led to a burgeoning interest among research students in soil C (SC). Interest amongst likeminded bodies such as Greening Australia and the NSW Government bureaucracy has led to a number of cooperative research projects, but the sites selected and the “treatments” led to a number of uncontrolled variations. Landholders are being encouraged to consider SC in roles as different as soil physical characteristics (soil water holding capacity, infiltration capacity, aggregate stability, to mention a few), soil chemistry (nutrient storage, buffering and uptake) and soil biology and ecology (as habitat, a substrate, a food source, plant exudates, VAM and other fungi, bacteria and viruses, and other micro organisms). The problem is that many of the more conservative and sustainable land management systems have little more than anecdotal evidence to back them and land managers are committing their financial future on decisions based on very little data, not to mention the dearth of analysis. Political discussion about long term C storage in soils with substances such as biochar and organic matter conversion in processes such as pyrolysis are examples where the validation experiments have not even been planned at this stage.

Addition of organic material as living organisms (agricultural and grazing management) and as already decomposing materials such as mulches and compost can be incorporated in already existing land management systems without large changes to landholder paradigms. These management systems seem to hold the greatest returns, because the changes made to increase C inputs, storage and cycling have huge benefits to the agricultural and grazing enterprises without risking returns and any additional financial returns from C sequestration would truly be cream on the cake. To this end projects examining a range of land management systems has been carried out. Each of those described turned up another example of an uncontrolled variation and a concomitant confounding of results, fortunately for the researcher in each case a rescue in the form of some more intricate statistical analysis, addition of extra data sets, or a careful redirection of the design and analysis brought home a worthy and viable project.

Methods
A range of projects conducted recently will be used to illustrate the examples of confounding factors.

Results and discussion

One example will suffice for the first set of issues – land manager commitment. In a recent study a researcher (Cheeseman 2006) set out to compare Biodynamic grazing systems with conventional systems on 5 different paired landholdings. In fence line contrasts Cheeseman set about comparing a range of soil attributes relevant to productivity and sustainability including; infiltration, bulk density, soil microbial respiration, SC and macro invertebrate counts. Cheeseman makes the point in his conclusions that he could explain the differences across the fence lines but being a Biodynamic manager or conventional operator ended up being only one of the explanatory variables. What is now called "operator variance" in our research circle has come to include the problem of land manager commitment to the system being managed, levels of competence in carrying out that system, levels of enthusiasm, financial commitment, and what many an innovative land manager has called "passion" on one side of the fence can often have a much greater effect than the actual system being implemented. The work of Cheeseman (2006) also had issues with land managers deciding to cut hay rather than grazing after agreeing to take part in the "experiment", forgetting that an area had been fertilised some 5 or more years earlier, limed or had a potent herbicide application, not to mention the inclusion or not, of rotational grazing without describing such likelihoods to the potential researcher.

In a project designed to examine the addition of C to the landscape by growing trees (deep rooted woody perennials) found in addition that direct seeded windbreaks and woodlots may have a C sequestration value, but a much more substantial return was from a "subsidiary effect" of the additional farm income that could be made available as a result of sheltering livestock and crops in the harsh weather conditions on the tablelands and slopes (Read 2008). The project also became an "effects of grazing", or not, of the windbreaks, when it turned out that some of the land managers regularly grazed their belts while others had only ever had stock in when fences went down. The effects were increased SC in the grazed belts, more than likely due to the more rapid decomposition of litter from the nutrients and microbes in excreta and from stock trampling and breaking up the litter. The incorporation of additional C into soils in pastures and crops resulting from hydration in flood plains under Natural Sequence Farming (NSF) techniques showed some benefits to SC but when the other benefits of increased biomass production, water retention on farm, increased yields of crops and pastures and the reduction in erosion by wind and water were all considered (Weber 2008), it appears that a difference in grazing rates had a greater effect than did "hydration". The saving grace was that one paddock on the NSF property was quite inaccessible and had a grazing history in the previous few years that was much more similar to the non NSF paddocks and in this one there were measurable and significant positive changes to a range of soil attributes, and a chance to make meaningful comparisons.

Changes to grazing systems such as Time Control Grazing not only benefit soil C but they allow sustainable increases in overall carrying capacity of livestock, reduced soil erosion, increased infiltration of rainfall, increased soil water storage, and better soil structure (Anderson 2004). There were however issues that were similar to Cheeseman (2006), but on this occasions all the fence line comparisons were within each of the 5 farms being studied. The issue then became the lack of commitment of the land manager to the conventional set stocking and because it had become abundantly clear to each of them that TCG was preferable, their commitment to maintaining set stocked paddocks faded over the 15 years of the experiment. The result was that ram and bull paddocks, paddocks near the house kept for the "killers" and others such as laneways had to be used. The problem was exacerbated because the overall grazing rate in these specialist paddocks was in general much lower then the TCG paddocks. Nevertheless, soil attributes were able to be discerned and concomitant increases in pasture perenniality came with carefully managed rotational grazing (Anderson 2004). The original project was unable to be completed and the benefits TCG brings in increased SC to considerable depth (anecdotally) have yet to be quantified.

The effects of pasture cropping (PC) have also been examined in detail by recent studies (James 2009; Warden, 2009) with on going research to examine the overall economics of the different systems. The general conclusion seems to back up widely held beliefs in the farmer and researcher communities that constant year round ground cover (living perennial plants and organic matter), absolutely minimal or zero tillage, short sharp grazing episodes with very long rest periods (ie ungrazed), and minimising inputs (fertiliser, a variety of biocides) all lead to not only increasing levels of soil C, better soil physical attributes, and more balanced soil nutrient levels in better buffered soil chemistry, but to increased profitability and reduced financial risk. The problem came in this research when those carrying out the research began to realise that although all land holders practised PC, PC itself was expressed in many forms, in fact in more forms that there were land holders, because two managers had two variants each operating on their properties. The researchers went on to note anecdotally, that a general reduction in levels of stress in the landholder community was observable, for those who are using these alternative systems and that may have also affected their results.

The techniques for measuring SC are currently many and varied and it appears that so are the results obtained by the variety of methods currently in use. There has been a glaring need to try and standardise these measurements because the politicians are currently quoting the lack of a consistent measurement technique as a very strong reason for not including agriculture in a C trading scheme. A recent study by McRorie (2009) has compared the Weil, Walkley-Black and Blair methods across a range of soils, and across a range of land management systems, while also comparing systems of representative sampling, problems of soil heterogeneity, sampling across and down slope positions and sample preparation techniques (McRorie 2009). Issues arose when data from otherwise reliable sources turned out not to be, and the experimental design had less soil types and systems covered and the reduction in what was to be variation led to a need for more intricate and complex statistical analysis. Nevertheless the Weil method turned out to be the most reliable, safe to use technique, and at the least cost for measuring SC, but sampling protocols, and sample preparation and sub-sampling protocols have to be used to get representative data to make decisions. In addition, the greatest disadvantage is that the Weil technique is marginally less able to be used to differentiate between land management systems, despite being very highly and significantly correlated with Walkley-Black (McRorie 2009).

Conclusion

Our conclusions after the last decade of work are that even greater care has to be used when working in the real world, with land managers operating real world businesses. On the other hand, research in these situations is rewarding and alternative agricultural and grazing systems do in general lead to better outcomes in a truly triple bottom line analysis. Grazing systems need to include rotations with long ungrazed spells; cropping should be carried out with continuous ground cover, and with zero tillage. Systems that combine the use of animals in a symbiotic relationship with cropping, and by doing so reduce or eliminate chemical inputs, have multiple benefits. They produce benefits on all of the (triple) bottom lines: better and more reliable profitability, with less land degradation and measurable rehabilitation of soils, and while the social factors are still being investigated anecdotal evidence to date suggests that communities will benefit too.

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Influence of land use change and water management on soil properties and water quality from combined geomodelling and geochemical modelling

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Abstract
Multisecular land use change and difficulties in water management lead to changes in soil properties, but the slow kinetics, the spatio-temporal variability of soils, land use and water quality makes these changes difficult to decipher. The method used here combines geomodelling (gOcad geomodel) and geochemical modelling (PHREEQC) to integrate multiple sources of information. The gOcad geomodel has long been recognized as an appropriate tool to reconstruction of 3D architectures in fundamental geology and geology engineering for minerals and oil prospection. It is used here to interpolate soil data, land use data and outputs of the geochemical model. The groundwater quality is the input of PHREEQC geochemical model. The output of the geochemical model is the saturation index (SI) of water with respect to calcite. By using classical data such as soil, hydrogeological and land use maps (Figures 1, 2), soil profiles, piezometric water analyses in time and combining the tools gOcad and PHREEQC (Figure 3) on a demonstration area in the Crau's plain, it is shown that: i) multisecular irrigation modifies soil properties by a recarbonatation process which counterbalances the general geochemical tendency towards decarbonatation; this results in the presence of active limestone in the topsoil; ii) water shortage for irrigation during dry years results in smaller SI (smaller oversaturation or slight undersaturation) in grasslands, and larger SI in land used for orchards, where groundwater was used for irrigation; iii) the presence in the subsoil of a "poudingue" of pedogenetic origin (pebbles cemented by a calcrete), thick, but fractured and discontinuous, does not prevent the land use to impact directly the water quality of the underground water.

Key Words
Soil, water, geomodelling, geochemistry, land use

Introduction
Land use change driven by demographic pressure and urbanization lead to change in soil properties. Climate change may result in larger interannual variations of temperature and rainfall, and shortage in water supply for irrigation. Soil and water resources are needed for food production, water supply and their degradation can impair the possibilities of cities to generate wealth and the economic resources for financing local development and managing public resources. Degradation of soils and waters thus impair the capacity of cities to develop and satisfy the needs of populations who live in. Data on soils, land use and water quality are generally available but dispersed, heterogeneous and incomplete. The aim of this paper is to show the interest to combine the geomodeller gOcad (Mallet 2002; Bile *et al.*2009) and the geochemical model PHREEQC (Parkhurst and Appelo 1999) to integrate multiple sources of information on the soil, land use and groundwater quality. More specifically, in many regions such as Mediterranean, soils have been irrigated since several centuries, and land use and soil properties are well known. This is the case of Crau's plain: this region is pedologically homogeneous, and one of the few remaining steppes of Europe, and soil and water resources are threatened by urban spreading, land use change and climate change. Parts of it have been irrigated since the XVIth century, with water from an alpine river that brings carbonate to the soil system, while other parts remained in the original land use (nomadism, sheep breeding). All region has been subjected to recent climate change and drought, so this study tries to assess both the influence of land use changes and water management on soil and groundwater quality, through the consideration of calcium carbonate geochemistry, which affects soil pH, soil buffer capacity, soil biota and hence all biogeochemical cycles.

Methods
Demonstration area, pedological and hydrogeological context
The Crau's plain results from the continental deposits of the Durance river before its stream way changed during the Alpille's mountains orogenesis and its capture by the Rhone river about 12,000 yrs BP. Crau in provençal language defines a field covered by stones. Today only 10,000 ha of this original steppe remain in the centre of

the plain, named “dry Crau” (Figure 1, 2). In the North, the “wet Crau” or “green Crau” is characterized by grassland irrigated since the XVIth century. The pebbles of Durance come from the Alps and are either of limestone or of siliceous nature, accumulated on large thickness, and are cemented by pedogenic precipitation of calcite (calcrete locally named “taparas”) between 40 and 60 cm depth, forming the “poudingue” horizon. The upper horizons of soil were decarbonated during the warmer and more humid period that succeeded to the Last Glaciation Maximum (18,000 yrs BP). The deposits moved progressively from the NW to the SE before the capture of Durance, hence the age of the parent material changes but not its nature (Figure 1). Soils are red fersiallitic soils in all the Crau’s plain. The thickness of the “poudingue” ranges from some meters on the upland zones to 50 m in the old stream ways of Durance river with a small slope oriented NE-SW. The Crau’s plain has a large underground water table, which flows from the NE to E and from NE to SW. There is no more direct connection between the groundwater and the Durance river, but water pumped upstream in the Durance river constitute 70% of the input of this water table via the agricultural irrigation. In the North, from E to W, grasslands predominate. In the SW of the area, the groundwater itself is used for irrigation of orchards. Just out of the limits of the groundwater (Figure 1, blue), in the South, there are natural wetlands and rice crops irrigated with water from the Rhône river. This Crau’s plain of 20,000 ha supplies drinking water to 250,000 inhabitants and water to the large industries located in the south of the territory. This area is submitted to diverse pressures, all in relationship with the spreading of the cities: (i) urban and industrial pressures concentrated in the South in relationship with the Fos industrial zone; (ii) spreading of urbanization from the districts of Saint-Martin de Crau, Miramas, Salon-de-Provence and Arles; (iii) increase of pressures on the underground water table: uptake, urban sludge spreading, diffuse pollution; (iv) increase of greenhouse fruit production.

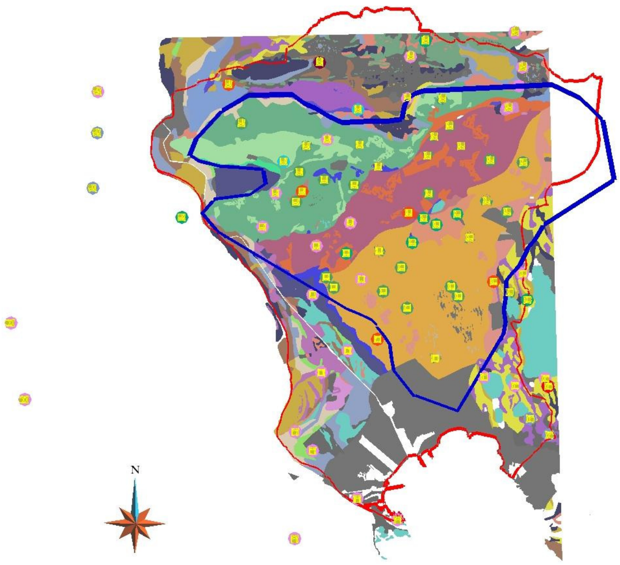

Figure 1. Soil map, soil profiles (points), underground water table limits (in blue) and limits of the study area (in red).

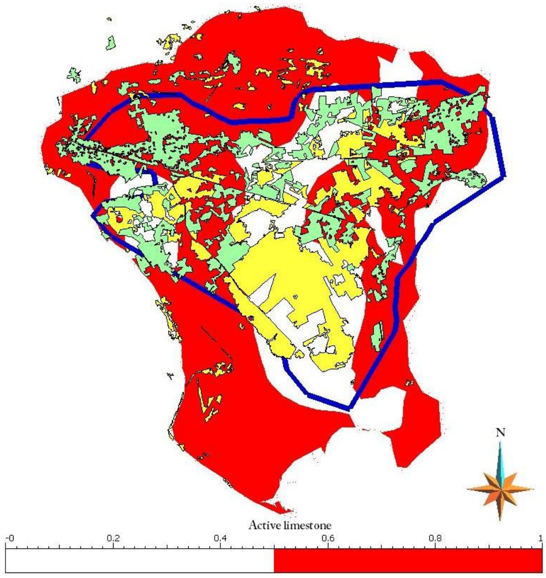

Figure 2. Land use: irrigated grassland, (in green), steppe (in yellow), soil containing active limestone (in red), or without active limestone (white).

Methods

Soil and parent material heterogeneities in these conditions have been mainly taken into account via Geographical Information Systems (GIS). But the main problem is that GIS split information in different strata, and natural objects, either soils or geological formations disappear. A priori knowledge on the objects, and different constraints, either as geometrical or physical, cannot be introduced in GIS. Geological Object Computed Aided Design (gOcad) instead is a geomodeller, designed for petroleum and mining applications in Nancy, on the basis of Discrete Smooth Interpolation by Mallet (2002). Constraints due to the nature of geological objects, such as a sedimentary basin, strata, faults, folds, ores etc. can be incorporated in the model. Soil data were obtained from the soil map (Bouteyre and Duclos 1994) considering not only limits of soil units but the exact location of soil profiles, succession of horizons and soil analyses (Figure 1). Among these latter, active limestone analysed in the upper two horizons was selected. It was transformed into a Boolean variable, as the presence or absence of active limestone is more important from a geochemical point of view than the content of active limestone: its presence indicates that irrigation water or rain water will rapidly react with calcium carbonate, irrespective of its content, before reaching the subsoil and groundwater.

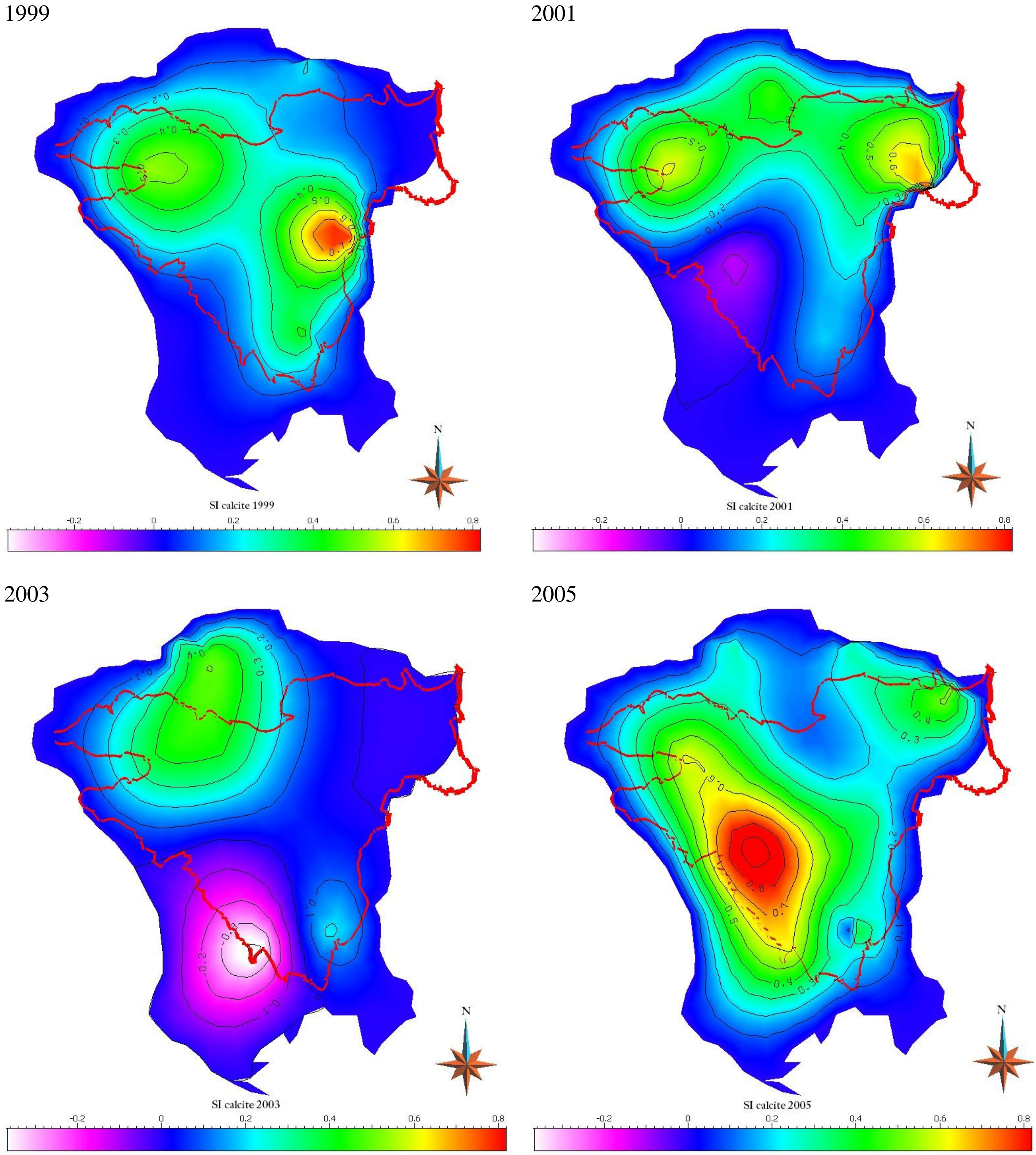

Figure 3. Computation by PHREEQC of the Saturation Index of calcite in underground water and spatial representation with gOcad at different dates.

Active limestone presence was spatialised using gOcad (Figure 2). Land use was obtained from the regional database of the Centre Régional de l'information géographique (CRIGE) and superimposed on active limestone in gOcad.. Water quality of the groundwater table was derived from the database Ades ("Analyse des eaux souterraines") of BRGM. This database gathers analyses of groundwater from 50 – 70 piezometers. The data were selected at the autumnal season, which is at the end of summer, the period when both concentration of water by evaporation is the largest and when irrigation stops. The contrast is then maximum between irrigated areas and "natural" steppic areas at this time. Rainfall and irrigation resources were normal in 1999 and 2001, while in 2003 and 2005 there were severe shortages of water supply due to unfavourable climatic conditions (warm temperature, low rainfall both locally and in the Alps), that generated social conflicts for water. Concentrations of inorganic elements, pH, alkalinity, temperature of aqueous solutions obtained from this database were used as inputs for geochemical modelling using PHREEQC (Parkhurst and Appelo 1999).

PHREEQC model was chosen because the calcium carbonate system was particularly refined in this model, both for thermodynamic modelling and kinetics. Saturation indexes (SI) with respect to calcite were thus locally obtained, providing for activity coefficients, ion pair formation and temperature dependence of equilibrium constants and of the solubility product of calcite. Due to the low ionic strength of solutions, Debye-Hückel extended law was used and not Pitzer equations. These data cannot be directly compared at different times and locations, as the database is not complete, but SI computed by PHREEQC were used as inputs to gOcad to obtain a spatial interpolation of SI that can then be used to evaluate the variations of SI with time and space (Figure 3).

Results and Discussion

Comparison of land use (Figure 2) and SI for calcite (Figure 3) shows that in "normal years" (1999 and 2001), irrigated areas are spatially correlated with large oversaturations of groundwater with respect to calcite, while the original non irrigated area corresponds to slight oversaturation or undersaturation with respect to calcite. This is consistent with the prevailing present geochemical conditions in the North Mediterranean: calcite tends to dissolve. The input of irrigation water from the Durance, whose basin is rich in limestones, and its concentration by evapotranspiration counterbalances this natural tendency and tends to favour calcium carbonate precipitation in the topsoil. This is confirmed independently by the spatial correlation between irrigated areas in the North of the study area and the presence of active limestone (Figure 2) and is in agreement with field observations by Duclos (Bouteyre and Duclos 1994) concluding to the "recarbonatation" of topsoil in irrigated soils. In years when water shortage was such that irrigation was stopped during the summer (2003 and at a lesser degree 2005), SI for calcite are smaller in the North. In 2003, just outside the study area, very small SI correspond to input of irrigation water from the Rhône river for rice crop. In 2005, the large SI in the SW of the study area corresponds to irrigation of orchards by pumping in the groundwater: when shortage of irrigation water from the Durance river occurs, farmers can afford to let grassland suffer, but not orchards.

Conclusion

Combining geomodelling and geochemical model is a powerful tool to integrate soil, land use and geochemical data, and to discuss spatio-temporal correlations. In temperate oceanic climate, there exists a general geochemical tendency towards calcium carbonate dissolution, calcium desaturation of the exchange complex and acidification. In Mediterranean climate, this tendency is smaller, but still exists if evaporation of waters with a positive alkalinity does not compensate it. This is the case in the North of Mediterranean. In the study area, multisecular irrigation with water from the Durance river has counterbalanced this tendency, which explains the presence of active limestone in the topsoil. This of course affects soil pH, buffer capacity, soil biota and all biogeochemical cycles. Another result of this study is that land use, via irrigation, and climatic variations (water shortage) directly affect the geochemistry of subjacent groundwater, despite the presence of a calcrete, which does not constitute a continuous geochemical barrier. This can be ascribed to the fracturation of this calcrete, which is a result of active tectonics in the region.

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Mapping and monitoring issues of a forest soil network in Southern Belgium

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Abstract
Soil monitoring has become a rising concern during last decade in Europe. A Forest Soil Survey (FSS) is being implemented in Southern Belgium as a component of a forest observation and monitoring programme. An analysis of the monitoring network has been performed at mid-term of the first investigation stage in order to assess current soil conditions and the temporal evolution that can be detected in the future. The fertility status and major and trace elements in forest soils have been investigated at regional scale through uni- and multivariate statistical analyses of the 410 soil samples of the network. A performance analysis of the network has been realized regardings the capacity to detect evolution of soil parameters. The first results show a moderate to strong variability according to the variable considered. High levels of variability were attributed to the presence of carbonated parent material in a distribution largely dominated by detritic terrigeneous rocks. The total contents in forest soils are mainly driven by pedo-geochemical background. The FSS allowed detailed mapping because there are clear convergences between spatial distributions of most of the elements and lithology or small natural regions. The levels of minimum detectable differences (MDD) that can be expected seem only compatible with the monitoring of soil acidification and changes in carbon stocks in the long-term. Future prospects should focus on the improvement of the MDD assessment.

Key Words
Forest soil, natural background mapping, soil monitoring.

Introduction
Soil protection should become a major environmental concern in Europe as the importance of soils in human activities and ecosystem functionning has recently been highlighted by the Soil Thematic Strategy for the Protection of Soil. Soil monitoring networks may be used as instruments (i) for the identification of risk areas according to major soil threats, (ii) for early detection of evolution, and (iii) to evaluate the effectiveness of soil protection measures (Van-Camp *et al.* 2004). The Walloon Forest Inventory is a region-wide programme dedicated to the monitoring of the forest condition in southern Belgium with a density of one observation every 50 ha. 10 000 observation plots are thus distributed over the 530 000 ha of forest land, along a regular 1.0 x 0.5 km rectangular grid. A Forest Soil Survey (FSS) is also being implemented from a selection of one tenth of these points. The monitoring network constitutes therefore the most detailed survey dedicated to assessment of forest soil conditions in southern Belgium. The objective of this paper is to assess the situation at mid-term of the first investigation stage regardings both mapping and monitoring potentialities of the network.

Material and methods
Field work and laboratory analyses

The 10 000 forest observation plots are located at the intersection of a regular 1.0 x 0.5 km grid, of which about ten percent should be soil-monitored. According to situations encountered on the field, 80 to 90 plots are sampled every year, which theorically supposes a time-frequency of 10 years for monitoring considerations. Up to now, five field campaigns have been completed and 410 soil samples have been analyzed. After field characterization of biophysical environment, soil was sampled by pooling twenty 20 cm-deep cores taken at the perimeter of three concentric circles (3 m, 9 m, and 15 m radius). The organic layers were removed before sampling. The samples were air-dried immediately after sampling and then sieved at 2 mm. A portion was ground to 200 µm for total analysis. The following variables were measured in the laboratory: total organic carbon (TOC), total nitrogen (NT), pH_{water}, pH_{KCl}, exchangeable acidity (Ac_{EXC}) and aluminium (Al_{EXC}), cationic exchangeable capacity (CEC), NH4Cl-extractible (EX) cations (Ca, Mg, K, Mn, Fe, Zn), total (T), mineral (MIN), and extractable (EX) P, and aqua-regia extractible (T) concentrations of Ca, Mg, K, Al, Fe, Mn, Cd, Co, Cr, Cu, Ni, Pb, and Zn.

Data processing and statistical methods
Statistical analysis were performed in order to (i) resume the data and identify the presence of possible subsets and outliers within the general population, (ii) identify and organize hierarchy within the driving factors of the soil properties, especially the lithological determinism, (iii) identify sites where risks of unfavourable forest development would need further investigations, and (iv) produce regional maps of the forest condition status. Multivariate Principal Component Analyses (PCA) were performed to summarize information and identify factors driving soil properties. The spatial structures of the variables were studied and compared to the geographical distribution patterns of various natural spatial entities (lithology, soil groups, ecological territories).

In order to predict the level of change that can theoretically be detected between two surveys, the power analysis or derived methods have frequently been used. The power (1-β) of a test is defined as the probability to reject null hypothesis when it is false. According to Dagnelie (1975), if α and β are small enough, respectively ≤ 0,1 and ≤ 0,5, the following relationship may be written:

$$N = \frac{2 * (u_{1-\alpha/2} + u_{1-\beta})^2 * \sigma^2}{\delta^2} \quad (1)$$

where σ^2 is the variance of the population and δ the mean differences, or the Minimum Detectable Difference (MDD) if the other parameters are fixed.

Results

What is the representativeness of the network according to soil spatial variability?
Forested areas represent 530 000 ha out of 1 690 300 but are unevenly distributed across Wallonia, depending on climatic, topographical, and lithological factors. Forest soils are mainly Cambisols, according to WRB classification. The other soil types which can be found under forest are very weakly represented (Luvisols, Regosols, and Gleysols), or even not at all (Podzols). The soils are mainly well drained and stony, as a consequence of both relief and lithology. For synthesis purpose (Figure 1), soil groups were classified according to texture (S: sand; L: loamy-sand; A: silty; E: clay; G: loamy and stony), nature of stones (fi: slate; r: shale and sandstone; f: shale; p: micaceous sandstone; k: limestone), intensity of natural drainage (1: favourable; 2: deficient). Other soils are only differenciated into peat soils, other stone charge, and other soils (complexes and artificial soils). In the FSS, stony soils represent 85% of samples. The nature of the stone charge is dominated (35%) by mixes of shales and sandstones. Unmixed charges of shale or sandstone parent materials are however very frequent too, respectively 26% and 19%. The soil thickness was classified into five classes by slices of 20 cm. Thirty percent of FSS soils are shallow (< 40 cm) while 35% are deep (> 80 cm). Seven classes of humus types have been identified from calcic mull to peat. Most frequent types are moder (35%), mull-moder (30%) and mull (16%). The most discriminating soil morphological properties in the FSS are the nature and abundance of the stone charge, the thickness of loose material, and the humus type.

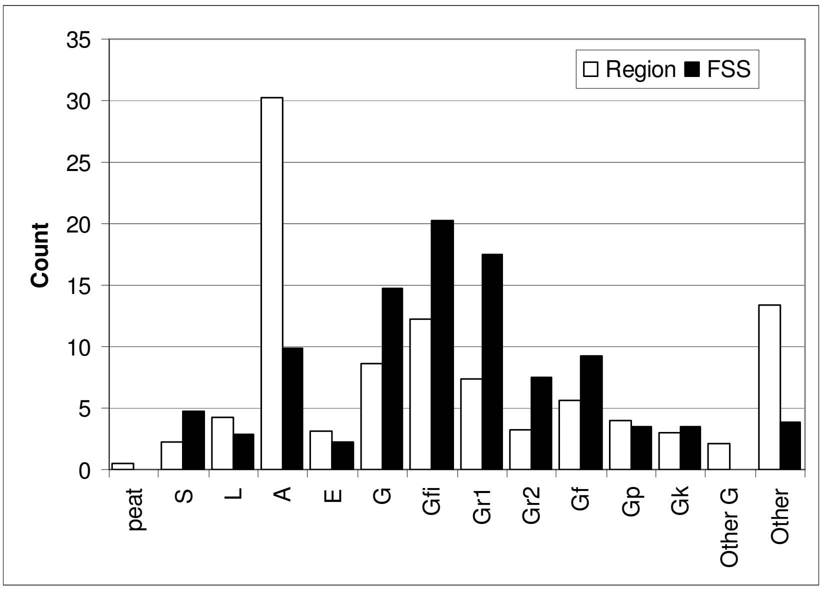

Figure 1. Frequency distribution of main soil groups in Walloon region and in the Forest Soil Survey.

What are driving factors of soil properties
Statistical indicators were calculated for each variable (data not shown). Most distributions are right-tailed skewed, which is common for soil properties (Webster, 2001). Skewness is particularly high for Pb_T (13.45), Cd_T (7.52), Ca_T (6.83), Zn_{EX} (6.65), Mg_{EX} (5.82), P_{MIN} (4.79), P_{EX} (4.53). To the exception of P, assymetry seems mainly due to the occurence of soils developped on two contrasted types of lithology, that is the detritic rocks composed basically by quartz on the one hand, and carbonated rocks on the other hand. The classical pedological characteristics (organic status, pH, CEC) present generally less variation than total chemical reserves and particularly than exchangeable pools.

A Principal Components analysis (PCA) was performed on the data in order to visualize the relationships between the variates and to identify the main factors driving their variability in the FSS. Results are presented in Figure 2. The 34 variables analyzed present high correlation coefficients and almost 70% of the variability is taken into account by the first four components of the ACP. The first axis is driven by the variables linked to presence of calcareous material). The second axis is influenced by most of total elements for which the distribution in soils formed on detritic rocks is linked to the abundance of quartz which plays as dilutant phase. The third axis expresses the variations of organic content and CEC which are closely linked. The fourth axis bears variability of P, which appears as an element with very specific behaviour as it can not easily be linked with other soil properties. It should be noticed that the exchangeable elements, especially Zn, are the variable for which the variations are the least taken into account by the four axis mostly because the behaviour of these elements in soil surface are sensitive to various environmental conditions which are specific for each element and hence scatters the weight of exchangeable pools on several different axis.

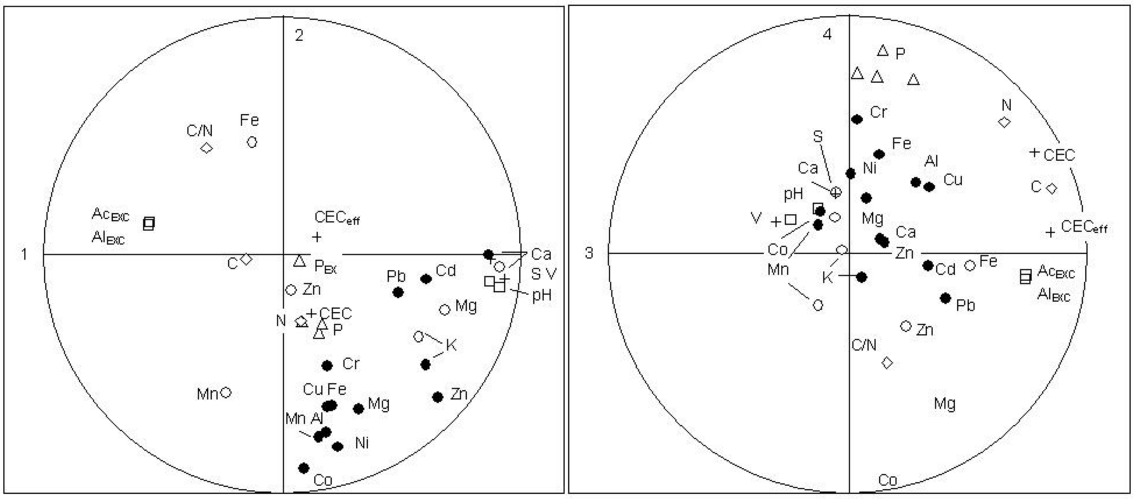

Figure 2. Scatter plots of correlations between forest soil properties and the first two principal components (left) or third and fourth (right). The symbols used for plotting soil variables are: ◊ for organic status, □ for acidity status, + for CEC group, ○ for NH_4Cl extractable elements, ● for total elements, and ∆ for P forms.

What is the network sensitivity to detect evolution?
MDD have been calculated according to Dagnelie (1975) for every variate (table not shown). The calculations were made considering $\alpha = 0.05$, $\beta = 0.10$, and assuming that 90 samples would be collected each year. The results for untransformed data can be directly linked to the variability found in the distributions. The higher variance, the larger relative confidence interval and MDD. Given the high number of samples, the maximum relative error on the estimation of the mean is between 2 and 5% for variables presenting the lowest variations and can amount to 15-30% for Ca contents. The best measurable differences to expect concern pH, C:N, CEC, exchangeable acidity, and organic content, which are variables that one wouldn't expect to see changing as fast as exchangeable pools of elements. Results for transformed data clearly stress the importance of having distributions approaching normality, even when the size of the sample is large. Calculated MDD appear higher than those found by other authors (Brejda *et al.* 2000; Goidts and van Wesemael 2007) but studies concern different environments and are therefore barely comparable. Moreover, we chose to fix a rather high level of power for the test in order to reduce the risk of missing effective evolutions.
Monitoring soil acidification or changes in carbon stocks within the FSS seems therefore promising. For the other variables, sample can be stratified into subsets with physical meaning, such as lithology or relief.

Identifying local trends could be also of more interest than assessment of a trend for regional mean. Provided that their number of samples be high enough, MDD may be calculated for geographical subsets.

What about mapping the soil properties?
At the regional scale, some long-range spatial structures could be identified. These result mainly from the geological structure of the Walloon region, where the lithological zonation is rather clearly marked. A subsequent goal of the stratifications is to allow the regional mapping through the attribution of distinct values to the polygons of existing soil, lithological, or ecological maps. As the number of samples in the various groups is largely unbalanced, non-parametric tests were preferred to parametric and Kruskall-Wallis tests were performed in order to identify groups with different medians (data not shown). Relationships were found between soil properties, mostly the acido-basic status and the variables linked to the nature of the parent material, and main soil types, or ecological territories. The soil map (texture and nature of stone charge) appears relevant at that scale too but does need a generalization process and fails however to differenciate the intra-type soil spatial variability. Some techniques of data interpolation combining qualitative and quantitative informations were investigated (results not shown). The density of observations in FSS allows regional mapping and identification of local specificities.

Conclusion

Forest soils in Wallonia are mainly Cambisols developed from sedimentary rocks. However, some diversity can be found among the soil series within short distances. The first results therefore show a large extreme-based variability and moderate variation coefficients. The exchangeable cations and carbonate–sensitive variates present the highest variabilities. Frequency distributions are often largely skewed. As a result, the prediction of statistical parameters suffers from a lack of precision and most data need transformation to approach normal distributions. Assuming that the variance of the sample was properly known, minimum detectable differences were assessed at regional scale for low levels of α and β error risks, and for annual evaluations. The resulting MDD appear therefore rather high for some variables, especially the exchangeable pools and the total contents, compared to the speed of soil transformations resulting from natural processes. However, smaller MDD should be obtained by (i) allowing to loose test power, (ii) raising the size of the sample through consideration of longer periods (e.g. 4 or 5 years), and (iii) by reducing the variance of the sample. For this particular point, the stratification of the survey according to qualitative attributes seems the most appropriate solution. Stratification constitutes furthermore a way for geographical differenciation of the expected MDD. On the other hand, scattering the sample into homogeneous subsets reduces the size of the comparison set and might lead to a loss of precision for monitoring considerations if the decrease of the variance is not marked enough. Moreover, it shouldn't be forgotten that the bigger the sample the more reliable estimation of the variance (Webster 2001). The sampling intensity of the FSS allowed rather fine geographical differentiation for every variable, which constitutes its main contribution to the knowledge of soil conditions in Wallonia. Sound references for pedo-geochemical content are now available and should allow easier identification of contaminations from both natural and human origins.

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Measuring soil organic carbon stocks – issues and considerations

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Abstract
Increasing soil organic carbon (SOC) stocks has been widely discussed as a short- to mid-term implementable solution to the problem of rising atmospheric greenhouse gas (GHG) concentrations, with the technical mitigation potential of soil carbon sequestration around 5 $GtCO_2eq$/yr by 2030. However, as the spatial variability of SOC is high, coefficients of variation in SOC stocks can rise sevenfold when scaling up from point sample to landscape scales, resulting in high uncertainties in calculations of SOC stocks. This hinders the ability to accurately measure changes in stocks at scales relevant to emissions trading schemes (ETSs). Further, the depth distribution of SOC has not been thoroughly investigated, resulting in possible underestimations of carbon sequestration due to inadequate sampling depth. Research is needed to eliminate these uncertainties in SOC stocks and enable soil organic carbon to be incorporated into ETSs. Reliable and low-cost methodologies for measuring SOC are also required, so analysis costs do not outweigh carbon credits. When assessing SOC stocks, sampling and analysis must be carefully planned to avoid bias and ensure accurate calculations.

Key Words
Soil organic carbon, sampling, spatial variability, vertical distribution, analysis.

Introduction
In 2005, approximately 10 – 12% of anthropogenic GHGs originated from agricultural activities, mainly as methane and nitrous oxide emissions from livestock and rice farming, with the net flux of carbon dioxide from soils estimated at only 0.04 Gt/year (Smith *et al.* 2007). However, soils store approximately 2344 Pg (1 Pg = 10^{15} g = 1Mt) of organic carbon worldwide - over three times the atmospheric carbon content - (Jobbagy and Jackson 2000) and manipulation of SOC stocks through land-use and management changes has been widely discussed as part of a solution to rising atmospheric levels of CO_2 (Lal 1997; Lorenz *et al.* 2007). The 2007 IPCC report estimated that around 5 $GtCO_2$-eq/yr could be sequestered in soils by 2030 (Smith *et al.* 2007). In order to provide governments and other interest groups with the foundations for effective policy making with regard to ETSs, soil scientists are faced with the challenge of identifying and quantifying the fluxes of soil GHGs. In particular, reliable methodologies and protocols for monitoring SOC stocks must be developed, which ideally are easily implementable and not costly. Baseline measurements of natural systems and current stocks must be made to ensure quantification of anthropogenic induced changes and avoid bias in ETSs. Further, the influence of different management practices and land-uses on SOC dynamics must be understood, as well as the effects of other variables, such as soil type, soil texture, climate, topography and vegetation. Finally, understanding SOC dynamics requires defining appropriate time-scales for monitoring changes and also definitions of efficacy and permanence of sequestration.

Sampling issues
Due to the heterogeneity of SOC distribution, the number of samples required to accurately assess SOC stocks at scales suitable for carbon trading is high. Scaling up of SOC stocks from point sample to landscape unit is problematic and caution should be made that any calculations are based on reliable data. Goidts *et al.* (2009) found coefficients of variation increased from 5 % to up to 35 % in SOC stocks when scaling up from sample to landscape scales in Belgian soils. In his review of global soil carbon storage, Schlesinger (1977) noted coefficients of variation of up to 87 %. Altering management practices in strongly degraded soils to increase carbon stocks by 0.01% per year has been proposed (Lal 1997), but such changes would be undetectable due to the heterogeneity of SOC distribution and limitations of analysis techniques. For carbon accounting purposes, a minimum detectable difference must be defined (mean ± one standard deviation, coefficient of variation within a land-unit?), and SOC changes must lie above it. This requires high resolution data to predict SOC stocks at appropriate scales within the required accuracy so only significant changes are accounted for. In turn, the appropriate scales require definition: which land-units should we investigate - field, farm, catchment? When designing sampling campaigns, taking into account the factors influencing SOC distribution, such as soil type, land-use, climate, and vegetation will help to optimise sampling depths and numbers, ensuring that samples

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accurately reflect the distribution of SOC at the site.

In addition to the problems arising from insufficient sample numbers, inadequate sampling procedures can produce a bias in data, leading to incorrect estimations of SOC stocks (Harrison *et al.* 2003). For instance, sampling with cores can lead to underestimation of the coarse soil fractions (> 2mm) due to the inability of corers to sample larger rocks. Gaudinski *et al.* (2000) found that CO_2 flux calculations are very sensitive to estimations of rock content. As SOC content is determined on sieved samples (< 2 mm), disregarding rock content in rocky soils will lead to overestimation of SOC stocks. For example, if rocks account for 30% of the volume of a sample but SOC concentrations and bulk density are determined on sieved samples, then calculating stocks without accounting for rock volume will lead to an overestimation of SOC stocks by 3/7, or over 40%. In soils with low rock content this effect may be negligible, but in soils with high (variability of) rock content, comparability between sampling events will be limited.

Temporal issues

SOC content varies not only spatially but also temporally. For example, comparing samples taken in July one year with samples in January 5 years later is unlikely to provide accurate information on SOC dynamics. So that carbon credits can be generated and traded in ETSs, a definition of and time-scale for assessing permanency of changes in SOC stocks is required. For instance, is permanency reaching steady-state conditions under new management practices? How long do SOC stocks have to remain constant for steady-state conditions to be declared? When are carbon credits generated – after reaching steady state or after a defined increase in carbon stocks? Chan and Hulugalle (1999) detected significant differences in SOC content three years after conversion of agricultural practices, indicating that assessment every few years may be necessary to evaluate SOC dynamics. However, Gaudinski *et al.* (2000) found continuing carbon sequestration over 100 years after reforestation of abandoned agricultural land, indicating that permanency may only be identifiable in the long-term which will complicate ETSs with credit trading on daily markets.

Measuring soil organic carbon

Accurate measurement of SOC content is costly and time-consuming. Determining SOC via wet oxidation and titration (Walkley and Black 1934) produces toxic dichromate waste, and incomplete oxidation can lead to underestimation of carbon stocks. Elemental analysis is assumed to deliver the most accurate results but equipment is costly, representativity is questionable due to the minute amount of sample used in the analysis, and any inorganic carbon present must be separately measured. Loss-on-ignition is low-cost and its reduced accuracy may be the trade-off for lowering of analysis costs and the ability to process more and larger samples, increasing representativity and satisfying the need for high resolution of SOC data.

Recently, infrared spectroscopy (IRS) has been applied to measure numerous soil properties including OC content and composition in bulk soils and soil fractions (Ellerbrock *et al.* 1999; Janik *et al.* 2007; Viscarra Rossel *et al.* 2006). IRS has the advantage of being fast, inexpensive, non-destructive, and requiring little to no sample pre-treatment. However, calibrations with traditional methods are required and accuracy is often limited to local data sets. IR analysis has also been successfully used to measure SOC *in situ*, which appears promising as a reliable, low-cost method for assessing SOC stocks on the field scale (Stevens *et al.* 2008), thus helping to resolve the problem of inadequate data resolution. Before this technique finds widespread application, issues with sampling methodology must be resolved. These include: accessibility to the soil, which must be free from vegetation for reliable measurements; depth of sampling; measurement of bulk density concurrent with SOC concentration *in situ*; and calibration of data sets.

Vertical distribution of SOC and sample depths

The vertical distribution of SOC is controlled by various factors. In reviewing over 2700 soil profiles worldwide, Jobaggy and Jackson (2000) concluded that vegetation and climate were associated with the relative vertical distribution of SOC, but climate and clay content were more important in determining the absolute amount of SOC stored. In surface layers, climate is the dominant control on SOC content, which is negatively correlated with temperature and positively correlated with rainfall. With increasing depth, clay content becomes the dominant control. Therefore, in sandy soils in arid zones sampling to half a metre may prove sufficient, as SOC concentrations in deeper layers may be below the detection limit, whereas in clay rich soils under high rainfall sampling to bedrock may be necessary to accurately assess SOC stocks. Furthermore, pedogenesis can lead to translocation of organic matter (e.g. during podzolisation), so soil type must also be considered when deciding upon appropriate sampling depths. In summary, different sampling depths may be required dependant upon climate, soil texture and soil type, which in turn creates issues when comparing land-use effects in soils with different physico-chemical identities.

Despite the fact that around 70 % of SOC is located below 30 cm (Batjes 1996), studies of SOC usually investigate topsoils (e.g. Dick 1983; Skjemstad *et al.* 2008). Jobaggy and Jackson (2000) were unable to directly calculate SOC stored below 1 m due to a scarcity of data, instead extrapolating from model distributions computed with data in upper layers. The rationale for investigating the topsoils seems to be that these are the active soil layers with the highest turnover and any changes in SOC stocks will occur here. For example, tilling of soils leads to SOC depletion associated with decreased aggregate stability (Chan *et al.* 2002) and a loss of CO_2 (Reicosky *et al.* 2005). Ease of, and the time and costs associated with, sampling deeper soil layers may also play a role in determining sampling-depth in many studies. However, ignoring deeper stored carbon seems risky when changing stocks could mean taxes or income to land-users. Chan and Hulugalle (1999) reported increases in SOC content 60 cm below the surface of irrigated Vertisols after conversion from conventional to minimum tillage, showing that deeply stored carbon can be dynamic. Defining appropriate sampling depths is essential for accounting purposes and it has been proposed that a depth of at least 1 m is required to assess SOC stocks (e.g. Young *et al.* 2005); Lorenz and Lal (2005) suggest that even this may be insufficient to accurately assess stocks. In any case, many more data are required before we can confidently measure changes in SOC stocks. Initial studies in an area should aim to characterise SOC distribution to bedrock with consequent sampling depths based upon previous data.

Current research needs

Soils are four-dimensional systems, changing across landscapes, with depth and through time. For ETS purposes, high-resolution, low-cost SOC data are required. In particular investigations of SOC variability and distribution at different scales, depths and through time, as influenced by internal and external factors such as soil texture, land-use and climate, are required, allowing confident assessment of SOC stocks and hence generation of carbon credits. Methodologies allowing rapid, easy and low-cost measurements of SOC stocks, such as *in situ* IR-measurements, must be developed so that measurement costs do not outweigh carbon credit payments.

Our research is currently focussed upon investigating the relationship between soil fine mineral content, the vertical distribution of SOC and SOC turnover in Wombeyan Caves, NSW. The karst bedrock above the cave system enables rapid transmission of water to the underground system, providing the possibility of linking SOC activity with dripwater chemistry. Speleothems are well-known environmental and climate archives, and this investigation will enable an initial analysis of the relationship between SOM turnover and speleothem C signals, linking past soil and climate processes.

Conclusions

For soil carbon sequestration to be incorporated into ETSs, SOC stocks and their dynamics must be reliably measured and closely monitored. Currently, insufficient data are available to confidently account for changing stocks, in particular heterogeneity of distribution in landscapes and deeply stored SOC. Closer attention must be paid to sampling procedures and calculations of stocks, with particular regard to the number of samples required to significantly assess stocks at scales required for accounting purposes, vertical distribution of SOC and the rock content of soils. Lastly, time-scales for assessing SOC dynamics and defining permanency are required, so carbon trading credits can be generated.

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Carbon sequestration in kiwifruit soils of New Zealand

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Abstract
Soil is a major component which sequesters huge amounts of carbon. In different locations and over time soil may exhibit a complex degree of variability in organic carbon content. Changes in soil organic carbon (SOC) levels have not been monitored for kiwifruit grown under different management practices. To evaluate the sequestration of organic carbon in Andisols used for kiwifruit production we analysed three management systems from three growing regions in the Bay of Plenty of New Zealand. Replicated soil samples were collected from three kiwifruit orchards per region. Within each orchard samples were taken from the grass alleyways, the wheel tracks and plant rows in three depths for two consecutive years. For comparison, soil was also collected from nearby pastoral and arable land as paired samples. In kiwifruit orchards of two regions, soil under organic management sequestered more carbon than under conventional management while the opposite was the case in the third region. We recorded higher SOC concentrations in the wheel tracks followed by the alleyway and then the plant row. In the top 0.15 m SOC significantly decreased over 5 cm increments. Kiwifruit orchards in the Katikati region sequestered more SOC than pastoral land and the reverse was true for the Te Puke orchard region. SOC stocks in kiwifruit orchards were significantly higher than under arable land use in all three regions. Our results indicated that soils of kiwifruit orchard can be a good carbon sink.

Key Words
Andisol, carbon footprints, climate change, southern hemisphere

Introduction
Increased greenhouse gases in the atmosphere are becoming a world wide concern. Scientists predict that current relative annual increase in tropospheric CO_2 is almost 0.5% and future levels are projected to double during the next century due to human activities. Resolving this problem requires decreasing CO_2 from the troposphere by either reducing or avoiding emissions, or increasing the amount of carbon sink activity. Depending on management and environment, soil organic carbon can either be a source or sink for carbon. It is estimated that about 40 x 10^3 to 80 x 10^3 million t C can be sequestered in soils over next 50 to 100 years through sustainable management. The magnitude of CO_2 emission from agricultural and deforestation activities is estimated at about 1.6 x 10^3 million t C y^{-1} and SOC sequestration potential could offset about 15% of global CO_2 emission (Lal 2007). A widespread loss of soil carbon has been reported under some dairy and dry stock pastoral land uses in New Zealand (Schipper *et al.* 2007). Australian research has shown that SOC declines after the conversion of land to arable cropping and the rate of decline was between 19 and 45% in major soils of cereal belt of southern Queensland after 20-70 years of cropping (Dalal and Mayer 1986, Skjemstad *et al.* 2001). On the other hand, conversion of cropland to pasture or forest is likely to lead to an increase in soil carbon (Vesterdal *et al.,* 2002). It is therefore predictable that agricultural management systems as well as climate and soil inherent properties determine whether soils will be a net carbon sink or source. Kiwifruit of the genus *Actinidia* (Actinidiaceae) is very widespread in both the northern hemisphere and southern hemisphere. In the southern hemisphere, New Zealand is the largest kiwifruit producer. The total acreage of kiwifruit planted in New Zealand now exceeds 12,177 ha with different management practices and species. New Zealand climate, soils and agricultural management practices are different from than those of other countries, and will have an important impact on soil carbon stocks in New Zealand compared with other regions. To our knowledge there is a scarcity of detailed research on carbon sequestration in kiwifruit systems. Investigating carbon sequestration also requires the ability to understand the spatio-temporal distribution of organic carbon in the soil system. We hypothesized that soil properties of kiwifruit orchard alter due to management practices over time, which leads to changes in soil organic carbon levels. The aim of the present study was to determine the effects of different management practices on carbon sequestration of Andisol used for kiwifruit production in the Bay of Plenty in New Zealand.

Materials and methods
In this study three management practices (organic, biological and conventional) were selected from three agro-ecological zones (Katikati: 37°36S 175°56E; Tauranga: 37°43S 176°06E and Te Puke: 37°47S 176°23E), located in the Bay of Plenty, North Island of New Zealand. The Katikati sites are approximately 19 km north-west of

the Tauranga sites, with the Te Puke sites 28 km to the east of the Tauranga sites. Katikati sites ranged from 15 to 20 m, Tauranga sites ranged from 20 to 48 m and Te Puke sites ranged from 7 to 12 m above sea level. The biological, conventional and organic sites are all within a kilometre of each other in each region. Sites were also included from nearby pastoral and arable land to use as pair samples. Soils of all experimental sites are classified as Allophanic Orthic Pumice soils (Vitrads/Vitricryands Andisol, USDA; Mollic Andosol, FAO) formed predominantly from ryolitic tephra between ~ 4000 and 40,000 years ago during the region's geographic history of periodic volcanic eruptions. Three sampling plots (bays) of 500 cm x 400 cm were randomly selected within each site for soil collection at 0-5cm, 5-10cm and 10-15cm depth. In kiwifruit orchards, soil samples were collected from between the plants along the row (plant row); in the middle of the sward between the rows (grass alleyway) and from the area that machinery travels along between the rows (wheel tracks) using Daiki Soil Sampler with 100cc core (Daiki Rika Kogyo Co., Ltd., Japan) in August 2008 and 2009. The plant row is often sprayed with an herbicide to control weeds, and also some banded application of fertiliser is used. This region of the orchard is also not compacted by any machinery operations. The wheel tracks are compacted by vehicle traffic, and the alleyway has a certain cover of various plant species that may affect the soil properties. Three phase distribution of soil and different forms of soil water were measured from undisturbed soil samples and particle size distribution and chemical properties were measured from disturbed and sieved (<2mm) soil sample. Soil properties were measured following standard methods (data not shown). All experimental sites were sandy loam soils. Soil organic carbon was measured using three recognised methods: wet chemistry (Walkley and Black 1934), dry chemistry (TruSpec® CHN Determinators, LECO Corp., St Joseph, MI, USA) and loss-on-ignition (Kalra and Maynard 1994) for the 81 soil samples, collected from different kiwifruit orchards at various depths including a wide range of soil types. Based on the results, a regional regression equation was developed to estimate SOC from loss-on-ignition (LOI) with optimum heating temperature and duration, as LOI is popular as a rapid, easy and inexpensive method. Carbon storage was estimated as: Carbon stock (t ha^{-1}) per soil layer = carbon (%) x bulk density (Mg m^{-3}) x layer depth (m) x 10, 000 (m^2 ha^{-1}). The relative proportion of carbon in the kiwifruit orchard and pastoral land to the 10-15cm deep sample from the arable land was calculated. All statistical analyses were performed by JMP 4.0 (SAS Institute, Cary, North Carolina, USA).

Results and discussion

Overall effect of management practice, depth, position, and regions on soil organic carbon was significant (MANOVA results are not shown). Various researchers have noted that soil organic carbon is influenced by management, climate, soil mineral composition, soil biota, and position in the landscape. Due to the interactions that occur between these factors it is difficult to determine the absolute importance of any single factor on soil organic carbon. However, management is arguably the most important, followed by climate (Lal 2007, Baldock and Skjemstad, 1999) and soil type. In Japanese apple orchard production systems on Andisol Rahman and Sugiyama (2008) observed that sampling time of year had a more predominant effect on soil organic carbon than management. In the present study, results revealed that the region is the most dominant factor in soil organic carbon sequestration. The minimum, maximum, mean, standard deviation, coefficient of variation, skewness and kurtosis for soil organic carbon kiwifruit orchard for depths, positions, managements and regions are depicted in Table 1.

Table 1. Comparisons of summary statistics for soil organic carbon in kiwifruit orchard of Bay of Plenty.

	Management[1] (n = 162)			Depth, cm (n = 162)			Position[2] (n = 162)			Zone[3] (n = 81)		
	Org	Bio	Con	0-5	5-10	10-15	AW	WT	PR	KK	TR	TP
Minimum, %	1.59	1.60	1.79	2.01	2.02	1.65	1.89	1.59	1.65	2.47	1.59	1.65
Maximum, %	7.39	8.72	7.52	8.72	7.15	7.51	6.97	7.52	7.51	8.72	7.31	7.61
Mean, %	3.96	4.11	3.93	4.82	3.82	3.36	3.98	4.29	3.72	5.14	3.66	3.19
SD	1.35	1.49	1.28	1.33	1.14	1.23	1.38	1.42	1.27	1.10	1.18	0.99
CV, %	34.1	36.2	32.7	27.7	29.8	36.5	34.7	33.0	34.2	21.5	32.3	31.0
Skewness	0.43	0.44	0.70	0.26	0.56	0.88	0.51	0.50	0.51	0.53	0.62	1.07
Kurtosis	-0.50	-0.38	0.40	-0.34	-0.30	0.77	-0.40	-0.09	-0.28	-0.14	0.15	2.02

[1]Org: Organic; Bio: Biological; Con: Conventional; AW[2]: Alleyway; WT: Wheel track; PR: Plant row; KK[3]: Katikati; TR: Tauranga; TP: Te Puke.

The present study showed a considerably larger variability at a smaller scale (CV: 21.5 to 36.2 %) which is comparable to the results of Goderya (1998). Biological and/or organic kiwifruit management systems stored more SOC than other systems. The highest SOC was recorded in the 0-5 cm layer, and decreased linearly with increased depth. Organic carbon content in soil depends on several factors, amongst which microbial community plays a significant role. In Andisol used for apple cultivation, recent research showed that organic carbon content positively correlated with fungi population, but negatively correlated with bacteria and

actinomycetes (Rahman and Sugiyama 208). The highest SOC was found under wheel track, followed by alleyway then plant row. Previous research in kiwifruit orchards has shown that more litter accumulation is associated with higher numbers of earthworms in wheel tracks than alleyway or plant row (M H Rahman *et al.* unpublished data 2008). Such accumulations in association with earthworm activities are most likely to contribute more SOC in wheel track. Baldock and Skejemstad (1999) pointed out that the activity of soil decomposers and fauna may be important for soil organic carbon.
The values of total organic carbon stocks in soils for different management systems averaged over two years are presented in Table 2. Soil organic carbon stocks varied significantly across regions with the highest in

Table 2. Organic carbon stock (t ha^{-1}) in soils (average for two years) with different management practices of Bay of Plenty.

Soil management	Depth, cm	Katikati	Tauranga	Te Puke	Average
Organic kiwifruit	0-5	24.41 (1.80)[1]	20.42 (1.94)	18.16 (3.27)	21.00 (2.09)
	5-10	19.12 (1.41)	14.01 (1.32)	14.51 (2.62)	15.88 (1.58)
	10-15	16.63 (1.23)	11.61 (1.10)	11.32 (2.04)	13.19 (1.31)
	LSD[2]	1.62	1.13	1.61	*1.57*
Biological kiwifruit	0-5	24.92 (1.84)	21.79 (2.07)	18.34 (3.21)	21.68 (2.16)
	5-10	19.17 (1.42)	15.81 (1.49)	13.13 (2.32)	16.04 (1.60)
	10-15	17.36 (1.28)	12.92 (1.22)	9.46 (1.68)	13.25 (1.32)
	LSD	1.31	1.11	1.14	*1.70*
Conventional kiwifruit	0-5	22.83 (1.69)	22.72 (2.16)	16.05 (2.88)	20.53 (2.04)
	5-10	17.15 (1.27)	16.26 (1.53)	12.71 (2.22)	15.37 (1.53)
	10-15	14.66 (1.08)	12.81 (1.20)	10.69 (1.87)	12.72 (1.26)
	LSD	1.21	1.03	1.12	*1.57*
Pastoral	0-5	21.95 (1.62)	20.84 (1.98)	20.11 (3.66)	20.97 (2.09)
	5-10	19.12 (1.41)	16.31 (1.54)	17.85 (3.21)	17.76 (1.77)
	10-15	15.12 (1.12)	12.68 (1.19)	7.65 (1.31)	11.82 (1.17)
	LSD	1.18	1.18	1.29	*1.83*
Arable	0-5	16.76 (1.24)	16.18 (1.54)	13.09 (2.35)	15.34 (1.53)
	5-10	16.01 (1.18)	13.59 (1.29)	11.30 (2.01)	13.63 (1.36)
	10-15	13.53 (1.00)	10.63 (1.00)	5.91 (1.00)	10.02 (1.00)
	LSD	1.10	1.05	1.17	*1.07*
Organic kiwifruit	0-15	60.15 (1.30)	46.04 (1.14)	43.99 (1.45)	50.06 (1.28)
Biological kiwifruit	0-15	61.45 (1.33)	50.52 (1.25)	40.93 (1.35)	50.96 (1.31)
Conventional kiwifruit	0-15	54.64 (1.18)	51.79 (1.28)	39.45 (1.30)	48.62 (1.24)
Pastoral	0-15	56.19 (1.21)	49.82 (1.23)	45.61 (1.51)	50.54 (1.30)
Arable	0-15	46.30 (1.00)	40.40 (1.00)	30.31 (1.00)	39.00 (1.00)
	LSD	3.05	2.52	2.27	*2.44*
Kiwifruit	0-15	58.75 (1.27)	49.45 (1.22)	41.46 (1.37)	49.89 (1.28)
Pastoral	0-15	56.19 (1.21)	49.82 (1.23)	45.61 (1.51)	50.54 (1.29)
Arable	0-15	46.30 (1.00)	40.40 (1.00)	30.31 (1.00)	39.00 (1.00)
	LSD	2.78	2.90	3.39	*2.56*

[1]Data within parenthesis are relative proportion in the kiwifruit orchard and arable land to 10-15 cm deep.
[2]LSD: Least significant difference at $p < 0.05$ among the values within columns.

Katikati and the lowest recorded in Te Puke. This may reflect climatic differences of the areas studied, the time that a given site has been under the present land use relative to the initial soil carbon content or a systematic difference in orchard management. The relative proportion of SOC was significantly higher in kiwifruit orchards than that of arable land (Table 2). Our results indicate that kiwifruit soil stored 49.9 t SOC ha^{-1} in top 0-15 cm layer, which is ~5 times higher than those of cotton soils in northern New South Wales of Australia (Knowles and Singh, 2003). Additionally, kiwifruit tend to have a deeper rooting system than shallow rooted plants and may sequester higher SOC at depth in kiwifruit orchards than in pastoral or arable systems. This study found that organic carbon storage capacity of kiwifruit orchard soils was ~1.3 times higher than arable soil at 0-15 cm layer. The SOC stocks in kiwifruit orchards averaged across the three regions and different management systems showed a decrease of about 40% in the top 15 cm of the soil. We intend to analyze the SOC stocks below a depth of 15 cm in kiwifruit orchards in the future.

In kiwifruit, there may be some more potential for increasing carbon stocks by adopting new land use and soil amelioration including bioenergy herbaceous perennial grasses, recycling organic products such as biochar and introducing soil microfauna. However the information on these aspects with special reference to carbon budgets is lacking in kiwifruit orchards. The information is also lacking related to climatic conditions and biological activity versus carbon sequestration in kiwifruit management systems which need to be elucidated.

Conclusions and considerations

This is a baseline study on carbon stocks in kiwifruit soils in the Bay of Plenty of New Zealand. Our preliminary study showed that organic management leads to significantly higher soil carbon storage than conventional kiwifruit management in two out of three regions. To maintain potentially high SOC in soils, higher C:N ratio (more lignin and less carbohydrate and protein) and/or lower N:C ratio plant or plant material should be established in the sward or introduced as compost or mulch. The following should to be considered in future research on carbon storage in kiwifruit orchards to mitigate and adapt to climate change: (i) soil process and profile (ii) plant species and management practices (iii) landscape and climate (iv) surveillance site and baseline benchmarks (vi) regional and temporal scale (vii) related and effective soil sampling technique and (viii) cost-effective and rapid SOC estimation method which will help to develop and disseminate guidelines for kiwifruit growers economically and environmentally sustainable carbon storage. Since undertaking this work, PlusGroup is now managing the Sustainable Farming Fund funded Carbon in Orchard Soils Team, investigating the role of different kiwifruit management systems on SOC; and the effect of SOC on kiwifruit production, carbon sequestration and carbon footprints.

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Potential carbon sequestration of Lombardy soils (Italy)

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Abstract
Lombardy soils have a high potential capacity to incorporate considerable amount of carbon, if carbon (C) saving land management is adopted. However, the certification of the changes of soil organic carbon (SOC) stock over the time is needed on the basis of transparent and cost effective methodologies. To this purpose, the soil sampling protocol recently proposed by the European Commission's Directorate General Joint Research Centre (JRC) of Ispra has been tested comparing three different land management occurring on the same soil type. Results showed a huge potential SOC sequestration rate ranging from 3,5 and 4,2 tC/ha/year and suggested the soil involvement in land-based C management practice can be actually feasible, even if the influence of soil mass on estimates of carbon storage should be in particular studied in more detail.

Key Words
Sampling protocol, uncertainty, reproducibility, CO_2 storage

Introduction
SOC stored in the upper 30 cm of the Lombardy soils is about 130 millions of tons. Nevertheless, this pool is varying according to bioclimatic conditions, soil types and land use (Brenna *et al.,* 2004). Due to historical cultivation the SOC is low in particular on the Po Plain, where cropland shows a mean content of 57 t/ha with level below 30-40 t/ha in some areas of the western and southern parts of Lombardy (Figure 1).

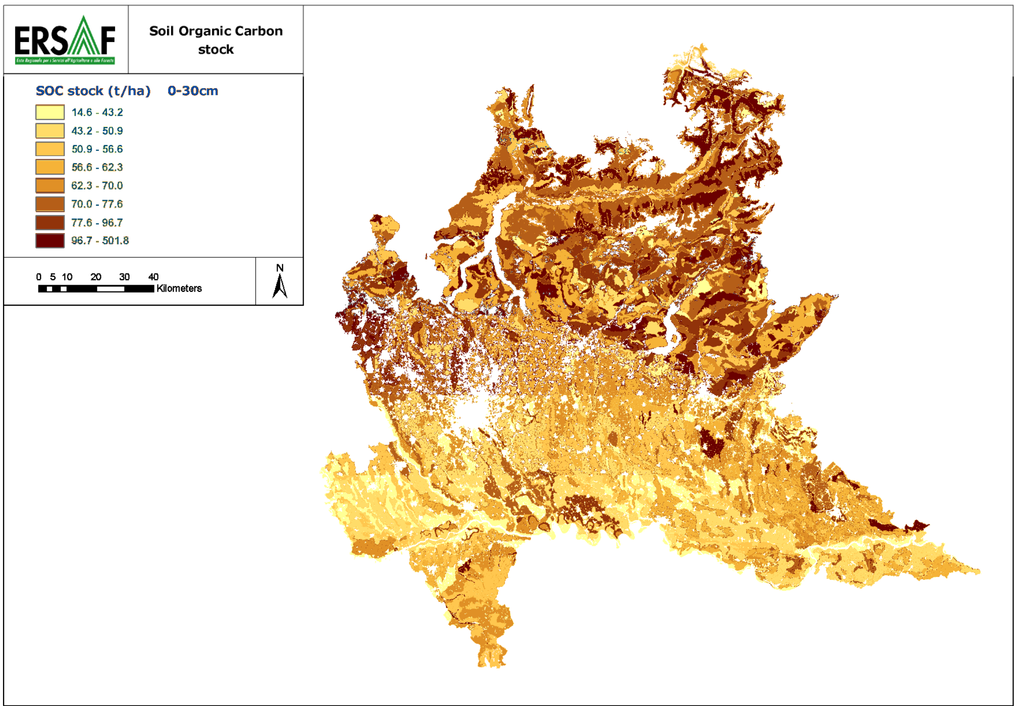

Figure 1. Soil organic carbon stock in Lombardy.

This amount of C is close to the content of the inertial fraction. Therefore these soils show a considerable capacity to re-gain a large amount of C under C saving management regime, e.g.reduced tillage and reduction of soil disturbance leading to decrease of mineralization of the crop residues, increase of organic input into soil, control of soil erosion, etc. In other words, C sequestration could become a big challenge as well as a relevant opportunity for agriculture of the region.
In fact increasing SOC by 0,1% (for example from 2 to 2,1%) in the ploughed layers of croplands (nearly 900.000 ha in Lombardy), could make the regional soil C stock to grow by over 3 million of tons, that are equivalent to a CO_2 storage of 10 million of tons.

Hence, the certification of the changes of SOC stock over the time is needed on the basis of common, simple, transparent and cost effective methodologies.
To this purpose, a soil sampling protocol, referred to a new method named "Area-Frame Randomised Soil Sampling"(AFRSS), has been recently proposed by the European Commission's Directorate General Joint Research Centre (JRC) of Ispra (Stolbovoy *et al*. 2005 and 2007).
The protocol follows the general requirements of the International Standard (ISO 2002) and is consistent with the principles of the IPCC Good Practice Guidance (IPCC 2003).
However, to bring any new method in practice does request considerable validation efforts; this validation is essential to set up boundary conditions for the method and to adjust the latter to a practical field survey.
In this study the protocol has been tested comparing three different land management occurring on the same soil type in one of the ERSAF (Regional Agency for Agriculture and Forests) experimental farms.
The test has been carried out in the frame of a wider research programme aimed at developing a regional soil C monitoring system and identifying suitable indicators and methodologies to upgrade information about the Lombardy SOC stock change and the SOC potential sequestration.

Methods
The validation test has been achieved in a farm located in the south-eastern part of Lombardy on the so called "Main Level of the Po Plain" (Late Pleistocene). The site is characterized by a nearly level surface (slope 0,2%), mean annual rainfall is about 650-700 mm (Figure 2).

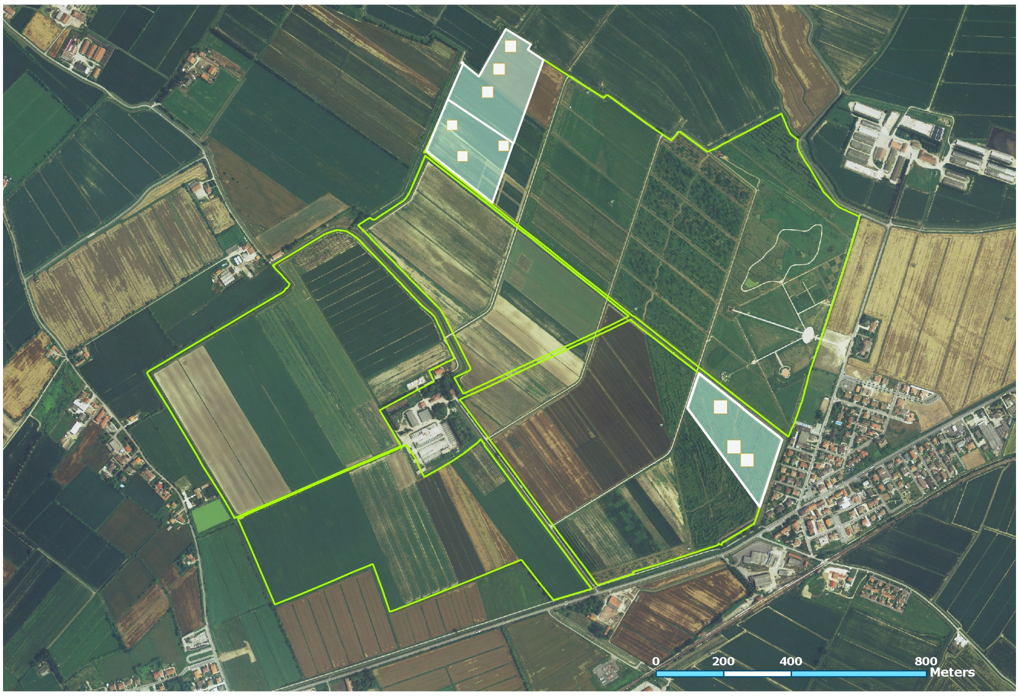

Figure 2. Aerial photograph of the ERSAF farm and localization of the testing plots, bounded by white lines.

Three testing plots standing for as many different types of land management have been identified. All of them have been cultivated in the pass with cereals (maize and winter cereals) for a long time. Currently the first plot is still cultivated with cereals (test plot identified as "cropland"), the second plot is occupied by a Short Rotation Forestry ("SRF") since 2004. The third plot is used as a forest plantation planted in the 2003 ("young forest"). The plantation includes "slow-growing" trees (oaks, hornbeams, ...), so that soil has been mainly covered by herbaceous vegetation till now.
The testing plots occur on the same soil type, classified as Endogley-Hypercalcic Calcisol (WRB 2006); the soil is developed on calcareous silt – fine sandy fluvial sediments, is moderately well drained, alkaline with a $CaCO_3$ content in the topsoil ranging from 4,1 to 26,8%, whereas a relatively shallow water table is around a depth of 100 cm. Rock fragments are lack in the soil profile and on the surface.
According to the AFRSS method, a template based on a grid with 100 cells resulted from 'modified random sampling' with a distance threshold has been considered and adapted to the testing plots (Figure 3).
Then three cells ("sampling sites") have been selected within each plot. Within the cells a "cross-sampling scheme" has been adopted, so that 9 sub-samples have been collected and mixed in a single composite sample for the SOC laboratory analysis. Also samples for the determination of bulk density have been taken.

The target of the AFRSS method is the estimate of the changes in SOC stock and its standard error, but the assessment of the reproducibility (R) of the sampling method, that is site-specific and corresponds to the minimum detectable SOC stock change, is also proposed.

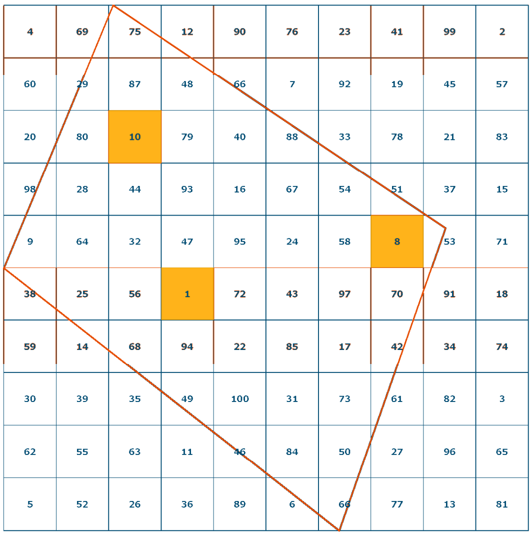

Figure 3. Adaptation of the template to the "cropland" plot

Therefore a second sampling has been performed with the same criteria on other three cells shifted for 5 m far from the first ones, to assess the reproducibility of the sampling result, that in practice simulates the error of the average coming from the mistake of the second sampling sites positioning, due to the inherent variability of soil characteristics over short distances, which is not tackled by the method.
Following the IPCC guidelines (IPCC 2003) suggesting one standard soil depth (0-30cm) for all soil types in different land cover classes, for the computation of SOC stock a soil thickness of 30 cm has been considered. For the determination of bulk density, undisturbed samples with a minimum volume of 100 cm^3 cylinder have been taken. Soil samples have been taken by auguring and composite samples analyzed in the laboratory for the SOC content with Walkley-Black method.

Results

Laboratory analysis show low SOC content, ranging from 10,90 g/kg ("cropland") to 11,89 g/kg ("SRF") and 13,79 g/kg ("young forest") on average. Bulk density is lower for "cropland" (1,17 g/cm^3) than for "SRF" (1,46 g/cm^3) and "young forest" (1,43 g/cm^3).
These measured data lead to a mean soil C density (SCD) of 3,81 kgC/m^2 for "cropland", 5,22 kgC/m^2 for "SRF" and 5,92 kgC/m^2 for "young forest".
The test provided a coefficient of variation (CV) around 6-8% for all the plots and reproducibility (R) values close to 5% for "cropland" and "SRF" and higher (9%) for "young forest" (Table 1).

PLOT	n samples	Mean SCD (Kg/m2)	Stand. Dev	CV (%)	R (%)
Cropland	6	3,81	0,33	8,75	5,77
SRF	6	5,22	0,34	6,47	4,48
Young forest	6	5,92	0,46	7,76	9,14

Table 1. Mean soil carbon density (SCD), standard deviation, coefficient of variability and reproducibility (R) pointed out by the test

The test results show a huge difference in SCD between "cropland" and the land management of the other tested plots. Clearly, the computation of the changes in SOC stock and the detection of the uncertainty are not applicable for a one time sampling. Nevertheless, assuming the same starting SOC content when all the plots were cultivated and no change in the cropland over the time, these differences, of about 21 tC/ha between "cropland" and "young forest" and 14,1 tC/ha between "cropland" and "SRF", would lead to a very impressive potential soil carbon sequestration rate ranging from 3,5 and 4,2 tC/ha/year, that should be attributed only to change in land use, being the soil the same in all the plots.

Conclusion

Obviously, the evaluation of uncertainty is crucial for any estimates and as a general opinion (Batjes 1996), SOC stock variability is in fact large, leading to doubts for the implementation of SOC management procedures. However, this assumption is provisional and the results of the validation test described in this study can contribute to the discussion concerning the uncertainty of SOC detection and the feasibility to develop an effective method to certify the change of SOC.

As already reported in previous studies (Stolbovoy *et al.*2007), also in this test all the soils show a similar pattern with a standard deviation of SOC stock less in the sampling sites having high SOC content. This finding suggests the uncertainty of SOC change expects to be less where the SOC enrichment occurs and should be considered a favorable argument to support soil involvement in the C sequestration practice.

Furthermore the validation test highlighted the difference in bulk density of the soil plots accounts for 40% and for 67% of the total observed change of SCD. The tested methodology basically involves calculating SOC stock in a given soil volume, as the product of bulk density, depth, and OC concentration. However, when soils are being compared on an equivalent soil volume basis, only if the average bulk density in all of the plots does not change much the evaluation of the SOC stocks will not be strongly affected; but, because overall variation in bulk density is common when land use systems are modified, or when tillage practices change (eg. till and no-till), it could induce some mis-interpretation of experimental results due to a computation of the SOC amount referred to different soil masses.

Recent publications (Ellert and Bettany 1995, Gifford and Roderick. 2003) indicate a serious and persistent lack of awareness about the influence of soil mass on estimates of nutrient storage.

Instead, according to these researchers, the SOC computation on the basis of an “equivalent soil mass” method could be very accurate, drastically reducing the standard error and allowing finding also small statistically detectable differences in soil organic carbon content.

Moreover, the application of the “equivalent soil mass” method would not require bulk density measurements, that are really time consuming and often providing practical sampling difficulties.

Thus, as a further step the test of a sampling approach based on the “equivalent soil mass” method to verify if it can lead to a sound reduction in the uncertainty of the results and an improvement on the SOC change estimation as well, taking in account the cost effectiveness, has been already planned.

However the study results encourage the soil involvement in land-based C management practice is actually feasible and uncertainty of the SOC change verification in mineral soils should not be considered as a constraint for its certification.

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Potential effects of different land uses on phosphorous loss over the slope in Hungary

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Abstract
Water erosion is a natural process and occurs on almost every open-air field. In, close-to-natural conditions soil degradation and soil formation reaches its climax, reflecting the environmental factors of a certain area. When we start agricultural production, forest and pasture or meadow management on an area, the threat of accelerated soil erosion occurs, thus the rate of soil degradation will exceed the rate of soil formation. In our study we have chosen slopes with pairs of contrasting land use (e.g. arable land with forest or arable land with meadow or meadow with forest, etc.) where the slope length and angle are similar under the different land use types. For methods we chose the methodology of the Hungarian Soil Information Monitoring System and took soil samples from the upper and from the lower third of the slopes in order to compare the soil properties on these slops. We performed laboratory measurements of basic soil parameters (pH [H_2O and KCl], SOM, P_2O_5, K_2O, $CaCO_3$). A good example of the results is with phosphorus because this is one of the best indicator for analysing the effect of water erosion as it is connected with the soil particles, so it is washed towards the lower slope together with soil aggregates where water erosion occurs. According to our measurements, the amount of the P_2O_5 is usually bigger at the lower third of the slope, and in those cases we found 2.6–680.3% more on the lower slope third. In general, the measurements provide help for farmers to reduce nutrient loss (save fertilizer), hold the nutrient at the right place and thus provide crops with the necessary amounts of nutrients to reach better yields, and this way they save the purity of surface water and use the environment in a considerate manner.

Key Words
Erosion, nutrient loss, slope sections, different crops.

Introduction
Soil erosion considered as serious problem on agricultural fields mostly in the humid tropics (Babalola *et al.* 2007). The average precipitation of Hungary in the hilly areas is between 600 and 800 mm/year. Even in these circumstances we can find high amount of soil and nutrient loss, severe erosion causing gullying and rills (Pottyondy *et al.* 2007; Bádonyi 2006; Jakab 2006; Várallyay 2007). The structure of crop rotations do not favour soil protection (Faucette *et al.* 2007), contains big number of medium or low soil protection crop (Barczi *et al.* 1999; Centeri 2002; Szilassi *et al.* 2006). Tillage practices cause large soil losses. Soil and nutrient loss, runoff and sediment yield calculations (Jakab and Szalai 2005) are important in protecting our valuable arable lands. Examination of soil parameters are essential to teach farmers better management practices in order to save nutrients, soils, money, time and to protect the environment (Jordan *et al.* 2005). Soil and nutrient loss are calculated in erosion models all over the world (Evelpidou 2006; Gournellos *et al.* 2004), especially in connection with cultivation (Hedin *et al.* 1995; Davidson 1969). Reduced soil fertility and subsequent reduction in plant growth lead to reduced canopy and soil cover, worse plant conditions and possible increase of weed species. In our work we show differences of sediment quality based on shallow drillings and deep soil profile descriptions for two remote areas.

Methods
Examination of the slope thirds followed the methodology of the Hungarian Soil Protection Monitoring Manual (Marth and Karkalik 2004). Samples for pedological survey were taken from the top 20 cm layer and basic laboratory analysis were done: pH (H_2O and KCl), $CaCO_3$ % (Scheibler method), humus % (Tyurin's method), P_2O_5 (mg/kg) and K_2O (mg/kg) (Buzás 1988). 41 slope section pairs are introduced in this paper, focusing on the phosphorous loss of the examined areas.

Results

It is not possible to include all basic soil descriptive data with the results of laboratory analyses we show the results of the Nemesgulács area as an example (Table 1.). The results of phosphorous measurements are in Table 2.

Table 1. Results of the laboratory analyses of soil samples, Nemesgulács, Hungary.

Sample site	Description	pH (H_2O)	pH (KCl)	$CaCO_3$ (%)	SOM (%)	AL-P_2O_5 (mg/kg)	AL-K_2O (mg/kg)
Nemesgulács	Horse pasture, upper slope third	7.81	7.39	14.51	4.25	164.5	214.9
	Horse pasture, lower slope third	7.65	7.23	1.32	4.74	374.5	441.8

Table 2. Values of phosphorous content for the upper and lower third of the slopes.

No.	Site name	Land use/crop	Upper Slope Third (P_2O_5) (mg/kg)	Lower Slope Third (P_2O_5) (mg/kg)
1.	Alsószuha	arable land (2004)	32.41	90.07
2.	Alsószuha	arable land (2006)	25.5	72.8
3.	Alsószuha	meadow (since 1990) (2004)	28.7	20.9
4.	Alsószuha	meadow (since 1963) (2004)	66.6	19.6
5.	Alsószuha	meadow (since 1990) (2006)	15.6	38.8
6.	Alsószuha	meadow (since 1963) (2006)	38.8	25.5
7.	Csopak	grey cattle pasture	501.6	1178.8
8.	Galgahévíz	arable land (2004)	1523.5	1322.0
9.	Galgahévíz	arable land (2006)	819.9	1652.8
10.	Galgahévíz, Sósi Creek	alfalfa	98.8	119.6
11.	Galgahévíz, Sósi Creek	deciduous forest I.	215.4	205.5
12.	Galgahévíz, Sósi Creek	deciduous forest II.	47.4	110.7
13.	Galgahévíz, Sósi Creek	arable land	160.1	169.9
14.	Gömörszőlős	arable land (2004)	140.8	166.4
15.	Gömörszőlős	arable land (2006)	88.4	141.0
16.	Gömörszőlős	meadow (2004)	110.1	181.6
17.	Gömörszőlős	meadow (2006)	128.2	163.5
18.	Maglód	black fallow	86.4	175.7
19.	Nagymező	horse pasture	108.7	56.6
20.	Nagymező	pasture, trampled	79.1	59.3
21.	Nagymező	control	53	36.8
22.	Somogybabod	alfalfa II	116.46	273.65
23.	Somogybabod	maize	16.25	39.14
24.	Somogybabod	black fallow II	8.77	29.94
25.	Somogybabod	winter wheat	8.37	45.41
26.	Somogybabod	black locust forest (2)	17.02	132.81
27.	Somogybabod	triticale (2)	277.09	303.1
28.	Somogybabod	alfalfa I (2)	88.99	99.63
29.	Somogybabod	alfalfa II (2)	155.27	247.93
30.	Somogybabod	maize (2)	261.09	267.94
31.	Somogybabod	black fallow I (2)	121.01	164.13
32.	Somogybabod	maize II (2)	76.84	130.12
33.	Somogybabod	black fallow II (2)	57.37	136.87
34.	Somogybabod	winter wheat (2)	48.6	101.97
35.	Tihany	horse pasture	163.5	185.4
36.	Pilismarót	winter wheat I.	46.4	93.9
37.	Pilismarót	winter wheat II.	86.9	100.8
38.	Pilismarót	alfalfa I.	72.1	59.3
39.	Pilismarót	alfalfa II.	63.2	34.6
40.	Pilismarót	maize	95.8	22.7
41.	Nemesgulács	horse pasture	164.5	374.5

The high amount of $CaCO_3$ content (Table 1.) proves that the examined soil was formed on loess parent material. Table 1 provides information on erosion processes concerning $CaCO_3$ content and other parameters. $CaCO_3$ content is approximately 12 times higher on the upper slope third because erosion took away the upper layers and tillage brought the $CaCO_3$ rich parent material closer to the soil surface. We can state that K_2O content is a good indicator of erosion, underlying our zero hypothesis of erosion causing nutrient runoff because

K_2O content at the lower slope thirds is doubled compared with the upper slope third.
Statistical analyses showed significant differences between the values measured on the lower and upper third of the slopes (T-probe, p=0.027) without distinguishing whether the amount was higher on the upper or lower slope thirds. According to our understanding about erosion and runoff processes, we assumed that there should be more phosphorous at the lower third of the slope. In case of samples No. 3., 4., 6., 8., 19., 20., 21., 38., 39. and 40. The phosphorous content was bigger on the upper third of the slope. There is an explanation for this because these are the areas where there is either extensive farming and/or soil protecting plants with smaller amount or no fertilizer used so the chance for severe soil and nutrient runoff is not as high as on intensive arable lands. The next step was – since we wanted to prove that there is significant difference between the slope thirds for the cases where there was more phosphorous on the lower slope third – we made an analysis with only those data where there were more phosphorous at the lower third of the slope. Statistical analyses showed significant differences with even less possibility of error (T probe, p=0.0097).

Conclusion

Soil nutrient loss can be proven by a simplified method as in our case. Based on the analyses of 41 slope third pairs we can state that the differences in phosphorous content of the upper and lower slope sections are significant. This simplified method can be used to prove the connection between soil water erosion and nutrient loss and not only its presence but its extent. These way farmers can choose the proper method of soil management and fertilization.

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Root zone soil moisture content in a Vertosol is accurately and conveniently measured by electromagnetic induction measurements with an EM38.

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Abstract
It is sometimes preferable to measure soil moisture content without destructive sampling or equipment installed in the field (as required for TDR probes and neutron probes). Electromagnetic induction (EMI) instruments are lightweight and portable, and measure apparent soil electrical conductivity (ECa), which is affected by moisture content. The EM38 is an EMI meter that is usually used for mapping salinity. As well as salinity and soil moisture, EMI meters respond to factors including clay content, soil temperature and magnetic minerals. Assessing the importance of these factors, and negating them where possible, would be a significant step toward using EMI for routine soil moisture measurement. This study shows that EMI measurements can accurately predict soil moisture content at a range of depths.

Key Words
EM38, soil moisture, electro-magnetic induction.

Introduction
Soil moisture measurement is a key to studies of the water balance. One of many methods of measuring soil water is to induce electrical currents in the soil and measure subsequent electromagnetic emissions (current lags voltage by 90º in an inductor). EMI meters for soil have been used since the 1960s (e.g. Howell 1967), often for metal detection, archaeological surveys and salinity mapping. The EM38 (Geonics Ltd., Ontario, Canada) is a contemporary EMI meter that is easy to use, lightweight, and can be used to rapidly measure many locations without the need for in-field installations or destructive sampling. It displays units of the apparent electrical conductivity (ECa, mS/m).

Soil moisture content affects the use of the EM38 for measuring soil spatial variability to the extent that Corwin and Lesch (2005) suggest "It is important to remember that if the water content of the soil drops too low (e.g. <0.10 cm^3 cm^{-3}), then the EM signal readings can become seriously dampened. In most practical applications, reliable EM signal data will be obtained when the soil is at or near field capacity. Surveying dry areas should be avoided."

ECa may be considered a function of the ECa of the soil solids (ECs, mS/m) and the soil solution (ECw, mS/m)), the soil moisture content (W, cm^3/cm^3) and a tortuosity coefficient (T) (Cook and Williams 1998, equation 1). Therefore, a curvilinear relationship might be expected between ECa and W, with an intercept of ECs.

$$\text{ECa (mS/m)} = \text{ECs} + \text{ECw.W.T} \qquad (1)$$

ECs is affected by the concentration of conductive, magnetic and dielectric materials in the soil. Buried metal, including metallic minerals, have extremely high ECa. Some soil magnetic materials possess magnetic viscosity, a transient magnetism that affects ECa. The minerals include magnetite and maghaemite. Magnetic viscosity depends mainly on the abundance, grain size distribution, and oxidation state of iron, titanium and other elements. Dielectric materials including water, organic matter and clay minerals affect ECa, but not EC, by transmitting alternating current (but not direct current). ECa also increases with temperature, by approximately 1.9% per degree C (Corwin and Lesch 2003).

If temperature is relatively constant, or accounted for, and the change in salinity is not great, a site-specific relationship between ECa and water content can be found (Akbar *et al.* 2005; Huth and Poulton 2007). The aim of this study was to measure soil moisture content and ECa in a black Vertosol (Isbell 1998). Key measures of success were the ease and quality of calibration, calibration quality at a range of soil depths, and ease of use.

Methods

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Suitability of the EM38 for measuring ECa
For EMI meters to measure a signal that is proportional to the apparent conductivity of the adjacent soil, the induction number (IN) must be less than 1, and preferably less than 0.1. IN is the unitless ratio of distance (m) between the coils to the depth of conduction (m). For the soil in question and an EM38, IN is very low. It is between 0.01 to 0.10, depending on the exact electrical conductivity and magnetic permeability of the soil.

Site and measurement information
ECa (mS/m) measurements were made with an EM38 and volumetric soil moisture measurements (W, g/cm^3) were made concerning a black Vertosol (Isbell 1998) near Pampas, 35 km southwest of Toowoomba. Duplicate cores were taken for gravimetric moisture content and a single core for bulk density, in 0.2 m increments. Measurements were made at a wide range of soil moisture contents, including moist soil (near the drained upper limit) that had been bare fallowed and much drier soil under nearby woodland vegetation. The EM38 was used in vertical and horizontal modes at the soil surface and 0.1 m and 0.4 m above the soil surface. Raising the EM38 reduces the depth of soil moisture measurement (Rhoades and Corwin 1981; Cook and Walker 1992). Table 1 shows the depths considered for these combinations of modes and heights above the ground. Two EM38 meters were used for some samples to check for meter-to-meter differences.

Table 1. Depth in the soil of the nominal depth of sounding (m) in two modes and at three heights above the ground. The nominal depth of sounding includes 90% of the depth response of the EM38 at 0m.

	Height of measurement (m)		
Mode	0	0.1	0.4
Vertical	1.5	1.4	1.1
Horizontal	0.75	0.65	0.35

Linear regression was used to predict soil moisture content (mm) to a range of soil depths (m) from ECa measurements (mS/m) (Table 1).

Effects of temperature
Measurements at a nearby site showed that subsoil temperatures were near their annual maxima (24 to 26 C, depending on depth) and relatively constant. Due to this, and wishing to consider the common scenario where data for soil temperature are not available, the ECa data were not corrected for soil temperature.

Results
Differences between meters
The two meters gave similar relationships between ECa readings and soil moisture content (Figure 1). It was concluded that they are sufficiently similar for both instruments to be calibrated together.

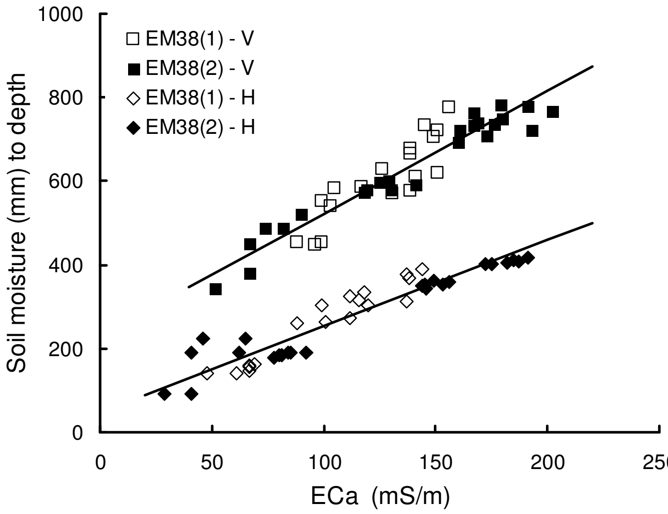

Figure 1. The calibration data from two EM38 meters (1 and 2) used in vertical (V) and horizontal (H) modes.

Relationships between ECa and soil moisture content
Figure 2 shows the relationship between ECa and soil moisture content for the EM38 used in the vertical mode.

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The regression data are shown in Table 2.

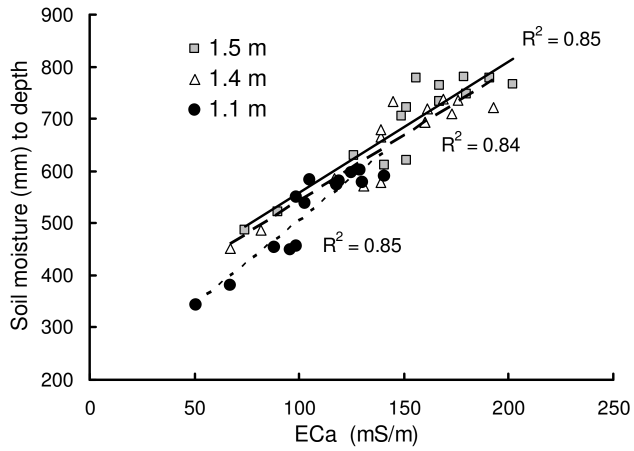

Figure 2. Soil moisture (mm) and ECa (mS/m) measured in vertical mode to 3 depths.

Figure 3 shows the relationship between ECa and soil moisture content for the EM38 used in the horizontal mode. The R^2 values are greater than for the vertical mode (see also Table 2).

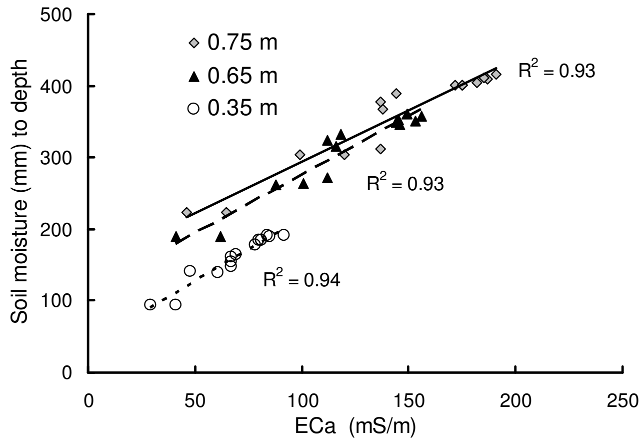

Figure 3. Soil moisture (mm) and ECa (mS/m) measured in horizontal mode to 3 depths.

Table 2. Regression coefficients and parameters for Y=a + bX, where Y is the soil moisture content (mm) and X is the ECa (mS/m), as measured by an EM38. The coefficients are not independent. SE is the standard error.

Depth (m)	a (mm)	b (mm/mS/m)	R^2	SE (mm)
0.35	38	1.77	0.94	9
0.65	113	1.63	0.93	16
0.75	152	1.42	0.93	19
1.1	183	3.22	0.85	33
1.4	292	2.50	0.84	38
1.5	307	2.52	0.84	39

Conclusions

Our principal finding is that the factors that might potentially blur the relationship between ECa and soil moisture content had only minor effects. These factors include air temperature, soil temperature and spatial variability in conductivity due to clay, salt and magnetic mineral content. The meter response appears linear over a wide range of soil moisture values, as verified by the high R^2 values in Figures 2 and 3. The response in vertical mode of approximately 3 mm of soil moisture per mS/m, and the small standard errors indicate that the EM38 discriminates moisture differences as small as a few mm, and is highly accurate when calibrated. To predict within 10 mm of the true mean of soil moisture content, approximately 5 measurements are required in vertical mode, and approximately 15 in horizontal mode. The absolute accuracy is sufficient for the EM38 to be used in research studies, where it is comparable to, or better than methods based on neutron scattering and time domain reflection.

Raising the EM38 to sample to a range of depths was more effective than we had hoped. Because the EM38 has a non-linear response to depth, it seemed likely that raising EM38 would reduce the quality of the calibrations for shallower measurements. However, good correlation was obtained for both orientations and all heights (Figures 2 and 3). In higher positions, there may have been slightly reduced sensitivity (increased the slope of the relationship, Figures 2 and 3). This was expected, because a smaller volume of soil is sampled as the height of the EM38 increases. Nevertheless, at 0.4 m height in horizontal mode, where the measured soil volume was the least, the relationship between moisture content and ECa has only a small error (standard error = 9 mm). We note that measuring surface soil moisture is often difficult or avoided with a neutron probe (as shields and extra calibrations are required), and soil coring for moisture and bulk density are labour-intensive. The EM38 appears to have considerable advantages over these technologies for repeated measurements of surface soil moisture.

In horizontal mode, the intercept of the calibration curves varied with depth (Figure 3). As sampling depth decreases, less soil and therefore less ECs is measured. The likely intercepts of the ECa data suggest that ECs for each depth is negative for this soil, (Figures 2 and 3). Negative ECs can be explained by dielectric and magnetic conduction in the soil. Current precedes voltage by 90º through a dielectric, which is the opposite (180º difference) of an inductor. Dielectric conduction therefore reduces induction. Magnetic minerals may also negate induction by altering the time-response of currents in soil.

The EM38 was easy and quick to use. Our rate of data collection was typically 400 ECa measurements per hour across a large field. Sampling in a smaller area would be faster.

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Soil carbon variability of a grassy woodland ecosystem in south-eastern Australia: Implications for sampling

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Abstract
A detailed examination was conducted on two paired-sites (grassy woodland and pasture) in central NSW, Australia, to assess the efficacy of different soil sampling regimes in the context of general ecosystem carbon (C) balance. Specifically, soil organic carbon (SOC) and bulk density (BD) were measured in order to calculate carbon density (CD), used for C accounting purposes. To achieve the same levels of accuracy in estimating the mean for SOC, substantially more samples were required in the highly variable woodland surface soils, whereas sub-surface numbers were more comparable between woodland versus pasture. Woodland soil BD was generally more variable than pasture, this being reflected in the need for larger sample numbers. The SOC variability was in part related to arbitrary demarcation between the soil component and other input sources of C such as the litter and CWD layers. This emphasized the importance of delineation between C pools in an effort to reduce measurement errors. This paper examines some of the issues related to these C pools and the problems of obtaining accurate C estimations.

Key words
Biomass, soil organic carbon; bulk density; spatial variability; paired-site.

Introduction
In recent years, increased attention has been given to the assessment of soil carbon stocks in relation to national scale carbon emissions accounting (e.g. AGO 2005) and for soil condition monitoring (McKenzie *et al.* 2002). In their assessment, McKenzie and Dixon (2006) highlighted the current lack of appropriate soils information and the need to address this problem by obtaining improved data for modelling.
Furthermore, appropriate delineation of the carbon pools (in both biomass and soil) is essential for accurate carbon accounting, as variability of each component is dependent upon this categorisation. Incorporation of surface dead biomass (e.g. litter and CWD) into the soil carbon component is logical as they provide the carbon inputs (Baldock and Nelson 1998). However there are few data available on these, and most soil carbon accounts deal only with that carbon found within the mineral soil (i.e. obtained via laboratory analysis).
The objectives of this study were to examine SOC variability by spatially sampling within the framework of current protocols (McKenzie *et al.* 2000) and to assess the efficacy of given approaches. Focus was placed on two paired-sites consisting of a grassy woodland and adjacent pasture in central NSW, Australia.

Materials and methods
The two paired-sites (1 and 2) in central NSW, consisted of grassy woodlands that were directly adjacent to cleared land now under pasture. Site 1 was located near Orange (S33^{0}21.439'/ E148^{0}54.624', Red Chromosol, 910 m a.s.l. with rainfall of 900 mm/year). The dominant vegetation was *Eucalyptus macrorhyncha, E. bridgesenia*, and *E. melliodora*. An adjacent pasture site was cleared of vegetation in the early 1960s, cropped for a few seasons, before being used for lightly grazed pasture until the present. Site 2 was at Canowindra (S330^{0}32.998'/ E1480^{0}40.343', Red Chromosol, 340 m a.s.l. with rainfall of 677 mm/year). Here the dominant vegetation was *E. Albens* and *Callitris glaucophylla,* with an adjacent cleared site dominated by the grass *Themeda triandra*. The grassy woodlands of both sites have experienced minimal disturbance from fire and stock grazing for many decades and both sites demonstrated highly diverse ground flora.
The soil sampling protocols of the Australian Greenhouse Office (McKenzie *et al.* 2000) recommend a paired-site approach using 25 × 25 m quadrats in both sites, from which a minimum of four samples are extracted. This framework was followed using at least 5 randomly located sampling points (15 and 5 for sites 1 and 2, respectively). In determining soil BD, an excavation method using the measurement technique to determine the volume of the excavated hole (0.20 × 0.20 × 0.05 m depth) was used (adapted from Blake and Hartge, 1986). Incremental sample depths of 0.05 m were taken to a total depth of 0.40 m. Coarse fragments (stones, roots and charcoal >2mm) were separated from all of these BD samples, dried at 40^{0}C and weighed. SOC analysis was predicted using a pre-calibrated mid-infra red spectrometer. Sample numbers (n) were calculated using $n = t^2 \times s^2 / d^2$, where, t = Student's t at 95%, s^2 = variance and d = specified acceptable error.

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Soil carbon density (CD) was calculated using the equivalent mass approach (Murphy *et al.* 2003). Litter amounts were estimated using 25 × 0.25 m^2 quadrats systematically placed within each of the the main 25 × 25 m sampling areas. All coarse woody debris (CWD) >2.5 cm in diameter, was directly measured within the woodland quadrats. Above and below ground tree and shrub biomass were assessed using the guidelines of Snowdon *et al.* (2002). Below ground tree and pasture root biomass amounts were estimated using a root: shoot ratio of 0.35 and 1.58, respectively (IPCC 2003, Table 3A.1.8, P. 3.168).

Results and Discussion

The C estimates contained within the ecosystem component parts of the two paired-sites 1 and 2 were estimated and summarized in Table 1.

Table 1. Summary of ecosystem carbon (t/ha) of Site 1 (nr. Orange) and Site 2 (Canowindra), central NSW, Australia.

	Trees	Shrubs	Wood site grasses & pasture grass	CWD	All litter classes	Charcoal pieces >2 mm (0-30 cm)	SOC i.e. CD (0-30 cm)	Sub-total ecosystem C	Total ecosystem C
	Site 1: wood								
Above ground	106.418	0.045	1.195	9.979	22.796	-	-	140.433	237.8
Below ground	22.792	0.014	0.121	-	-	3.76	73.87	97.557	
	Site 1: pasture								
Above ground	-	-	1.213	-	-	-	-	1.213	51.9
Below ground	-	-	1.324	-	-	3.884	45.49	50.698	
	Site 2: wood								
Above ground	74.319	0.033	0.906	1.573	24.14	-	-	100.971	176.4
Below ground	24.731	0.01	0.064	-	-	4.357	46.27	75.432	
	Site 2: pasture								
Above ground	-	-	2.915	-	-	-	-	2.915	39.8
Below ground	-	-	3.183	-	-	1.209	32.45	36.842	

Some generalisations were possible from Table 1:

- Total ecosystem C amount found in the *E. Macrorhyncha mid-high open-forest* (Hnatiuk *et al.* 2006) was 237.8 t/ha. There had been a total loss of 185.9 t/ha (78.2%) after this ecosystem was converted to pasture (~45 years). This figure also represented an approximation of the C sequestration potential if reafforestation to the same original baseline vegetation was to take place.
- When similar estimates were made for Site 2, the *E. albens sparse mid-high woodland* (Hnatiuk *et al.* 2006) had a total C content of 176.4 t/ha, with a post-clearance C loss of 136.6 t/ha (77.4%) on conversion to pasture.
- Above ground C therefore accounted for 59.0% and 57.2% of the total ecosystem C for Sites 1 and 2, respectively. Corresponding below ground C pools therefore accounted for 41.0% and 42.8%, respectively, of the total.
- A large proportion of the total ecosystem C occurred in the tree biomass of the uncleared sites (44.7% and 42.1% for Sites 1 and 2, respectively). About 75% of the above ground total C was accounted for by the above ground tree C, for both woodlands of Sites 1 and 2. Since this component represents proportionally large C pools, even relatively small differences between C estimate methodologies (e.g. through choice of allometric equation to determine biomass) can be potentially substantial, especially when extrapolated.
- 31.0% and 26.2% of the total woodland ecosystem C was in the 0-30 cm soil depth for Sites 1 and 2. A larger proportion of the pasture ecosystem C was in the same depth of soil, that is, 87.6% and 81.6%, respectively.
- A larger proportion of SOC (0-30 cm) was lost in Site 2 than in Site 1 due to land clearance (38.4% and 29.9%, respectively), probably due to the higher annual mean temperatures at Site 2.
- The mean litter amount for Site 1, when twigs (<2.5 cm diameter) were included, was 22.8 ± 4.21 t/ha (where n = 25). However when ten separate groups, each with only 3 replicate samples were assessed, as recommended in the given protocols (McKenzie *et al.* 2000), the mean litter estimates had a range from 16.8 ± 7.07 t/ha to 32.0 ± 26.7 t/ha. Similar high variability was obtained for Site 2. The large range and wide variability of the 3 replicate samples sets was a reflection of that variability associated with twig inclusion and the problems of determining where the demarcation between components should be, in this

case between twigs and CWD. A number of CWD definitions based on a range of CWD diameters, have been reported (Woldendorp and Kennan 2005) making comparisons difficult.

- CWD only accounted for 4.2% and 0.9% of the total ecosystem C for the woodlands at Sites 1 and 2, whereas litter accounted for 9.6% and 13.7%, respectively. CWD estimates using the method described by McKenzie *et al.* (2000) compared with actual total measured amounts (Table 1), underestimated CWD by about 40% and 3% for Sites 1 and 2, respectively. Such uncertainty in CWD estimates, related to high variability of CWD, was also demonstrated by Woldendorp and Kennan (2005).
- Charcoal pieces (>2 mm) in the 0-30 cm soil depth accounted for 1.6%, 7.5%, 2.5% and 3.0% of the total ecosystem C for the Site 1 wood, Site 1 pasture, Site 2 wood and Site 2 pasture, respectively. Whilst charcoal can often constitute between 15 to 20% of the total SOC in the top 20 cm of soil, values can also vary between 0 to 70% for the same depth (Skjemstad and Baldock 2006).

Differences in SOC between the contrasting land uses were apparent only in the surface 0.30 m (Figure 1). For the depth increments close to the surface, SOC amounts in the wood site were approximately double those in the pasture. With subsequent depth increments, amounts became more comparable until a depth of about 0.30-0.35 m after which no significant differences between wood and pasture occurred. This indicates the depth of disturbance through land use change and subsequent management and reinforces the IPCC (1997) recommendation of 0.30 m as a sampling depth for C accounting purposes.

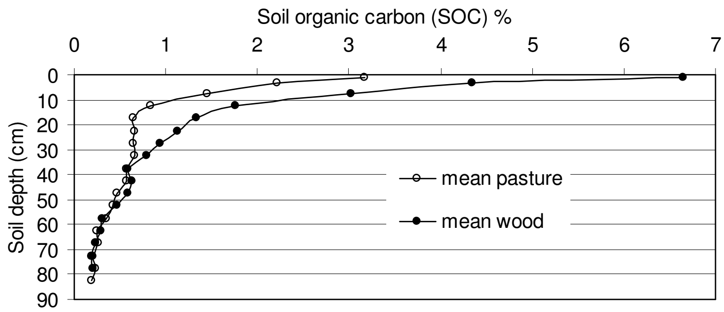

Figure 1. Change in SOC with soil depth for wood and pasture at Site 1.

An assessment of sample number requirements (*n*) for SOC and BD (Figure 2) was made for each soil depth increment. Only the Site 1 results are shown: the wood and pasture of Site 2 showed similar trends. Figure 2 is now discussed in relation to variability.

There is a gradual, but substantial, decrease in SOC *n* requirement from surface to subsurface in the woodland (Figure 2a). In addition, considerably more samples were needed in the surface soil of the woodland, compared to the pasture, at the same error levels. However, for both sites below 0.15 m, SOC *n* requirements were similar for the same levels of error (*d*) which corresponded to those similar amounts of SOC with depth shown in Figure 1. Whilst fewer SOC samples were needed from the surface horizons of the pasture soil, there was a larger *n* for the pasture at 0.25-0.30 and 0.30-0.35 m (CV% of 31.2 and 25.5 respectively, compared with 13.5 and 14.4 for woodland, respectively). This was likely due to the influence of localised concentrations of charcoal found in the pasture, probably due to the method of land clearance i.e. in situ burning of tree stumps/woodpiles. This charcoal inclusion also increased BD variability at these depths (Figure 2d). At the 0.35-0.40 m depth increment is the lowest *n* requirement for both the wood and pasture sites. This probably reflects a general limit of influence from management practices in the disturbed pasture and reduced bioturbation in the undisturbed woodland site as also indicated in Figure 1.

More BD samples would be needed in the woodland than in the pasture to give the same levels of accuracy, as indicated by the *d* values (Figures 2c & 2d). In the woodland soil, fewer BD samples would be required for surface horizons compared to subsurface horizons, with a wider spread of *n* requirements with depth. This was reflected in the corresponding CV% values showing a general increase with depth from 8.8 (0.05-0.10 m) to 15.9 (0.35-0.40 m). This was probably due to an interactive effect of coarse fragments and bioturbation. For the pasture, BD sample number requirements were less variable with depth (Figure 2d), reflected in a narrow CV% range of 4.0-6.7 for all depth increments. This uniformity was probably related to soil slumping due to disturbance, stock compaction, with less bioturbation than occurs in the woodland site.

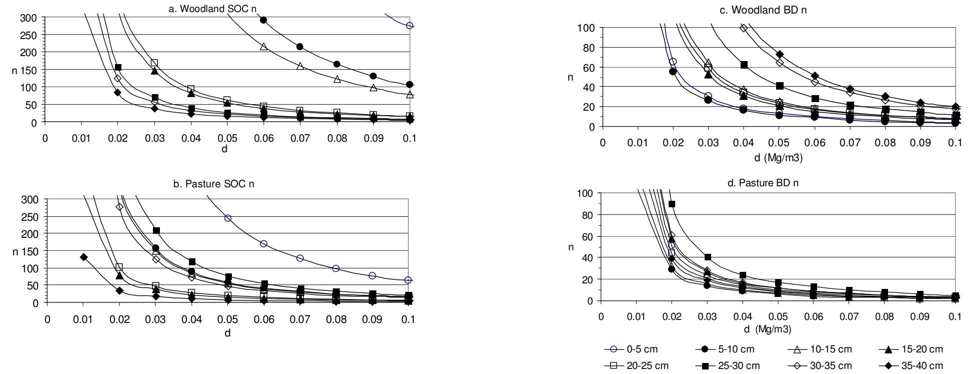

Figure 2. SOC and BD (excavation method) sample number requirements (*n*) in relation to specified acceptable error (*d*) for the woodland and pasture sites at the 95% probability level. Site 1 results presented only: Site 2 showed similar trends.

Conclusion

In estimating SOC amounts, in particular for soil monitoring purposes, sampling intensity has to be sensitive enough to detect long-term change in soil properties. This is dependent upon soil variability, influenced to a large extent by management type and history. While differences in sampling requirements do occur to depth in the soil, it is recommended that at least a minimum of 10 samples be obtained per sampling unit (usually 25 m × 25 m), especially in wooded situations where soil variability is greatest.

This paper has identified a number of errors that may confound accurate quantification of C amounts (both biomass and soil carbon density) for a site. These include natural variations and as well as sampling, measurement and calculation errors. Each one of these sources of variability, if not handled correctly, has the potential to undermine C accounting efforts especially when extrapolated to regional or national scale.

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Soil compaction survey in Estonia

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Abstract
The aim of the study was to detect the current conditions of Estonian soils in case of compaction and to test the indicators recommended in the result of the Environmental Assessment of Soil for Monitoring (ENVASSO) project final report in 2008. The soil compaction survey on Estonian fields was conducted in the fall of 2008 in a collaboration of Estonian University of Life Sciences and Agricultural Research Centre. Fifteen annual crop fields and 3 grasslands were under survey on different soils. Current study results for 4 study areas are presented as for those areas the data are for the last survey (1987) available. The parameters measured were soil bulk density, total and air filled porosity at 6 kPa suction, soil texture and content of organic carbon. Samples were collected from 5 randomly selected places from each field from 5-10 cm and 20-25 cm depth in two replications. One sampling place was selected at the field edge or tyre track. In three places under investigation the bulk density measured in 2008 was higher than in 1987 by 0.15 to 0.39 g/cm^3 and in one area remained the same at both depths. In all investigated places soil bulk density exceeded 1.55 g/cm^3, which is over the suggested limit of those soils. Soils total porosity was around 40%, but air filled porosity exceeded 10% only in two places in the top 5-10 cm soil. In field edges or tyre tracks statistically significant lower aeration porosity and higher bulk density were measured compared to the rest of field. However, even if the soil bulk density values in some places were over the recommended limit, the aeration porosity showed the good condition of these soils.

Key Words
Soil compaction, bulk density, aeration porosity, survey.

Introduction
Soil is definitely one of the most important environmental components but also one of the most underestimated, abused and ill-treated resource on Earth. Systematic soil surveys were started in the 1950s in most countries against the background of an urgent need for increased agricultural productivity. In Estonia, systematic large-scale soil mapping was initiated in 1949 and from 1954 special survey was carried out under supervision of the Ministry of Agriculture (Reintam *et al*. 2003). However, from EU 25 countries only 9 lave some direct legislation for soil protection (Thematic Strategy 2006) and in other countries no direct legislation for soil protections are implemented, including Estonia. In Estonia systematic survey of agricultural area started in year 1983 and extended to the year 1994. During that survey the main attention was paid to macronutrients in soils. In the beginning of that period also the special survey of soil compaction was carried out, during which the bulk density in the plough layer and under it were measured. Soil survey was restarted by order of Ministry of Environment by Agricultural Research Centre in 2001. No soil compaction studies were included in this survey. As it is now more than 20 years from the last soil compaction survey, an application was presented to the Ministry of Agriculture in cooperation with the Agricultural Research Centre and Estonian University of Life Sciences. In spring of 2008 funding was received and the work could start in the fall of this year. The aim of the current study is to present some results of the soil compaction study carried out in 2008 in Estonia. Innovative during that survey was that first in time that air filled porosity as an indicator was included in to the survey.

Methods
The soil compaction survey on Estonian fields was conducted in the fall of 2008 in collaboration of Estonian University of Life Sciences (EULS) and Agricultural Research Centre (ARC) and funded by Estonian Ministry of Agriculture. Fifteen annual crop fields and 3 grasslands were under survey on different soils. In current study results of 4 study areas are presented as for those areas the data of last survey (published in 1987) are available. The places selected for this study were Kiislimõisa and Tuuleveski on sandy clay loam soil, Laiuse and Avispea on sandy clay soil. By World Reference Base for Soil Resources WRB (2006) classification of the soils of experimental areas are the follow: Kiislimõisa and Laiuse – Stagnosol, Tuuleveski and Avispea – Calcaric Phaeozem.
The parameters measured were soil bulk density, total porosity, air filled porosity at 6 kPa suction, soil texture

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and content of organic carbon. Organic carbon measurement was needed for calculation of soil porosity.

Field measurements
Samples were collected from 5 randomly selected places from each field from 5-10 cm and 20-25 cm depths in two replications. One sampling place was selected from the field edge or tyre track. Sampling was done by 100 cm^3 steel cylinders in September and October after harvest and before ploughing. Special soil samples were taken to measure soil texture and organic carbon content. Sampling was done by ARC.

Laboratory measurements
Soil texture, bulk density and porosity analyses were carried out at in the laboratories of the Department of Soil Science and Agrochemistry of EULS and organic carbon measurements in laboratories of ARC.
To find out the air filled porosity, the soil samples taken from the field were weighed and saturated with water. After saturation the soils with the cylinders were placed on ecoTech plastic suction plates at 6 kPa suction for approximately 10 days (to equilibrium water content). After that, to find out the water content, porosity and dry bulk density, the soil samples taken from the apparatus were weighed and dried at 105°C to constant weight and weighed again. After that the water content, total porosity, air filled porosity and dry bulk density were calculated. Samples for the determination of particle size were treated with sodium pyrophosphate to break down aggregates. Sands were sieved and fractions finer than 0.063 mm were determined by pipette analysis (van Reeuwijk 2002). Air–dried soil samples were sieved through a 2-mm sieve and used to determine soil carbon (Corg) (Vorobyova 1998).

Statistical analysis
The statistical analyses were used the software Statistica 8.0 and analysis of variance (ANOVA) was implemented to find out the least significant differences (LSD) at $P<0.05$.

Results
Results of the survey on four fields in Estonia indicated that soil bulk density at a depth of 5-10 cm has been increased the most being in the Avispea experimental area, where the bulk density ranges to 1.65 g/cm^3 in 2008 compared to 1.35 g/cm^3 in1987 (Figure 1a). In general terms the bulk density is the lowest on Tuuleveski experimental area (about 1.45 g/cm^3 in 2008). Bulk density has remained on the same level on Laiuse experimental area. At a depth of 20-25 cm the soil bulk density has significantly increased on Tuuleveski experimental area (Figure 1b) compared to 1987. The highest values of bulk density were measured on the Avispea experimental area. Bulk density has remained at the same level on the Kiislimõisa experimental area. On the Laiuse experimental area the bulk density had decreased by 2008 (1.54 g/cm^3) compared to earlier measurements (1.67 g/cm^3).

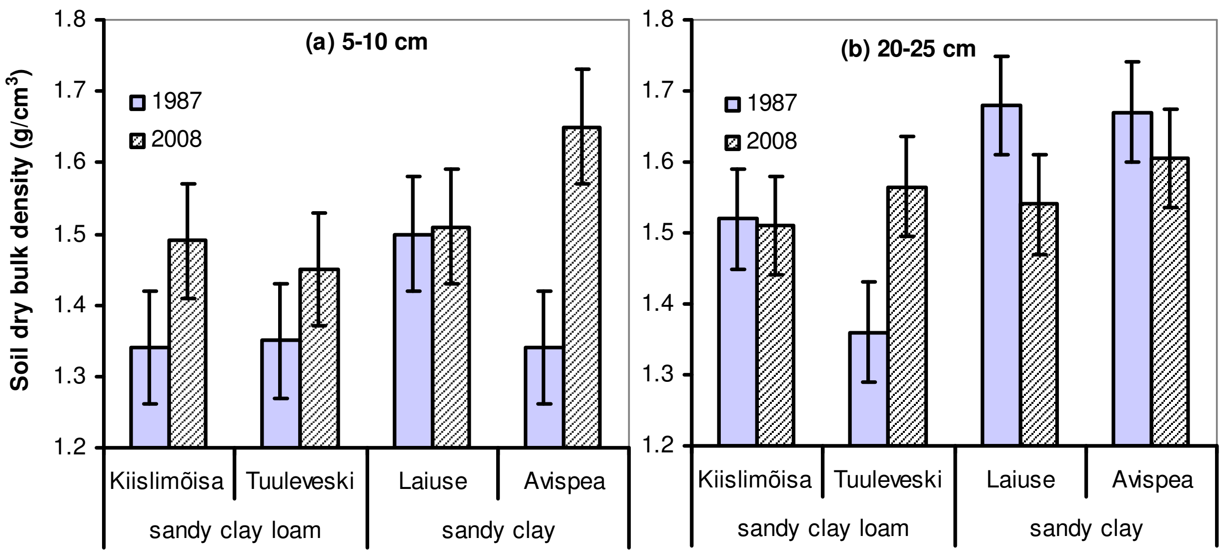

Figure 1. Soil dry bulk density of four survey sites in 1987 and 2008 in 5-10 cm (a) and 20-25 cm (b) depth in Estonia. Vertical bars denote least significant difference at *P*<0.05.

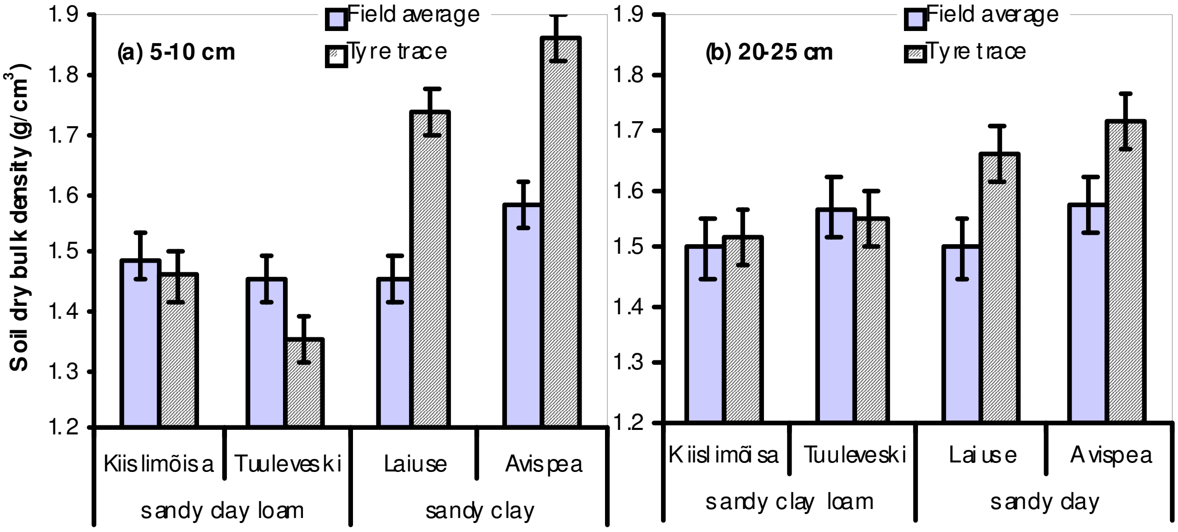

Figure 2. Soil dry bulk density of four survey sites as average of study field and in tyre track in 5-10 cm (a) and 20-25 cm (b) depth in Estonia in year 2008. Vertical bars denote least significant difference at *P*<0.05.

Highest values of soil dry bulk density were measured on sandy clay loam soil on Laiuse and Avispea in tyre tracks in 5-10 cm depth where it exceeded 1.7 g/cm^3 (Figure 2a). No significant differences were detected in soil of Kiislimõisa and Tuuleveski. Under the plough layer the bulk density was higher than in the plough layer (Figure 2b). As for topsoil also in deeper soil significantly higher bulk density values exist in tyre tracks for Laiuse and Avispea. On Tuuleveski experimental area bulk density at field edge was lower than the average of the field.

The aeration porosity in wheel tracks at a depth of 5-10 cm was lower in two experimental sites– in Laiuse and Avispea (Figure 3a). In other two places there was more air filled pores in track line. But probably it was not the real track line. In wheel tracks at a depth of 20-25 cm the aeration porosity was also higher or on a same level with the average of variables (Figure 3b). Generally the aeration porosity was less than 10% therefore the soil is suffering shortness of air. Lowest aeration porosity was measured at Avispea in tracks at 5–10 cm depth, where it was significantly lower from the average value of the field. In deeper soil no statistically significant differences between study sites were detected. The highest moisture content was measured on Laiuse 2 experimental area (about 33%) and the lowest on Avispea experimental area (about 25%). In general the moisture content at depth was slightly lower but on the Avispea experimental area it was higher than at a depth 5-10 cm.

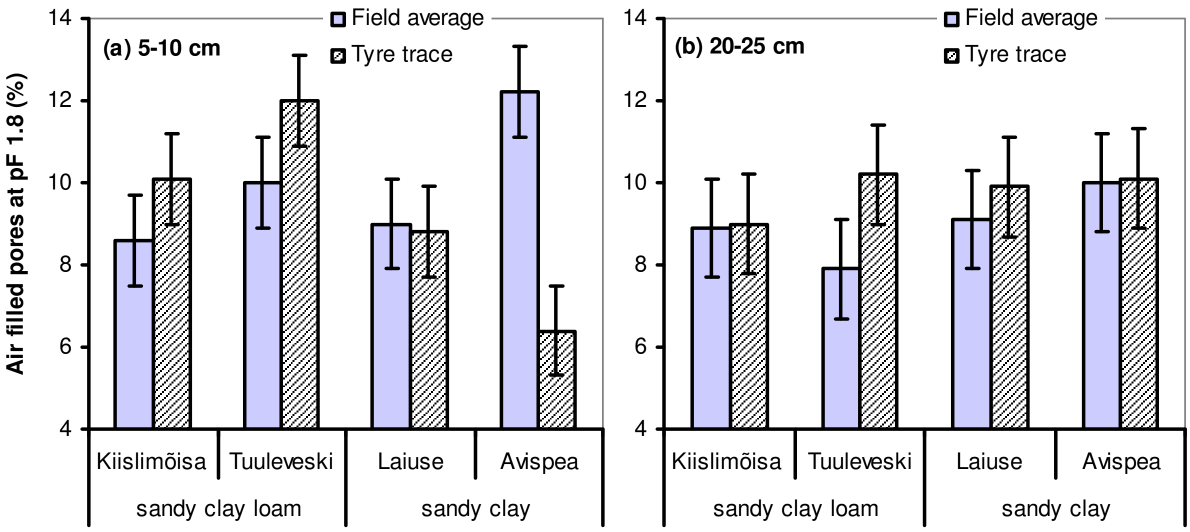

Figure 3. Soil air filled pores at pF 1.8 (6 kPa) of four survey sites in 2008 in 5-10 cm (a) and 20-25 cm (b) depth in Estonia. Vertical bars denote least significant difference at *P*<0.05.

In soil of all investigated sites the bulk density exceeded recommended values for plant growing. For most of the cultivated plants the suitable soil bulk density is 1.0–1.35 g/cm^3 and acceptable limit is 1.5 g/cm^3 (Lehtveer and Nugis 1987). In our earlier studies the plant growth limiting bulk density in sandy loam soil was 1.6 g/cm^3 (Reintam *et al.* 2008). In studies of Daddow and Warrington (1983) (in Gray 2002) the plant growth limiting value of dry bulk density for sandy clay soils is between 1.55 and 1.65 g/cm^3 and for sandy clay loam soils between 1.55 and 1.75 g/cm^3. Those critical values were exceeded only in track tracks in the soils of investigated areas. However, average aeration porosity remained below 10% in soils of investigated areas. Some researchers found that if aeration porosity is below 10%, there is not enough air in the soil, 10–25% is moderately aerated and over 25% is a well aerated soil (Conlin and Driessche 2000). In the conclusions of ENVASSO project they found that 10% is required for a satisfactory medium for plant growth and at least 5% air filled pores in subsoil are needed in well structured soil at 5kPa suction (ENVASSO 2008). Soil aeration in Kiislimõisa, Tuuleveski and Laiuse in plough layer was, but in the subsoil the aeration was over 5%.

Conclusion

Soil compaction in Estonia is a problem. Bulk density and porosity is in a poor state, so it would be necessary to continue the investigation of soil compaction and thus to identify the best methods to improve the situation of our soils. To evaluate the soil conditions, the air filled porosity at 6 kPa indicates the conditions better than just measurement of soil bulk density.

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Soil survey and inventory of dynamic soil properties in the U.S.A.

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Abstract
The U.S. National Cooperative Soil Survey (NCSS) is implementing a program to evaluate dynamic soil properties (DSP) and their response to changes in land use and management for all lands in nation as a new standard protocol. The objective of this paper is to present an overview of this protocol and its design. The design for the DSP inventory is a comparison study that substitutes space for time to infer change. This inventory relies on the assumption that properties of a native soil are sufficiently uniform across the landscape that changes due to land use and management can be detected by data collected from replicate plots in stable land use and management systems. A minimum set of DSP's were selected by analysis of rankings from a panel of experts. Properties selected for sampling were those that have known relationships to soil and ecosystem processes and functions, are relatively insensitive to daily or seasonal fluctuations in environmental conditions, are easy to measure accurately and precisely by different people, and are relatively low cost. The evaluation identified eight properties, organic C, pH, electrical conductivity, bulk density, soil structure, wet aggregate stability, total N, and soil stability.

Key words
Soil function, land use, agricultural management, organic carbon, soil quality, comparison studies

Introduction
Information about how soils change and human impacts on soil function is crucial to sustainable soil management. During the last few decades, the focus of user's needs for soil survey information expanded from erosion control, productivity, land use planning, and infrastructure development to include soil, water and environmental quality, ecosystem sustainability, food-security, biofuel production and soil response to climate change. The need to address these and similar issues demands development of additional knowledge of the state of the soil resource, conditions that can be achieved under specific land uses and management systems, and procedures to predict response to management as an aid to policy makers, land managers, producers, and others who make decisions that protect soil function.

In response to heightened demand for information on soil quality and function, the U.S. NCSS has begun a new standard program to evaluate and inventory disturbance-sensitive DSP for all lands in the U.S. The objective of the DSP inventory is to understand land use and management effects on DSP and soil function for all U.S. soils and to provide tools for land managers and others to better design and implement management systems to maintain and enhance soil quality and ecosystem services. The inventory will focus on soil properties that 1) may change over time scales of decades to centuries, 2) are important for interpreting soil function, 3) reflect management effects as well as soil and vegetation dynamics, 4) identify positive and negative critical management thresholds, and 5) can be documented with measurements recorded at one point in time. The objective of this paper is to present an overview of the protocol developed by the NCSS to produce a scientifically defensible and statistically valid inventory of DSP associated with land use and management.

DSP Protocols
The design selected for the DSP inventory is a comparison study that substitutes space for time to infer change. The basis for the comparisons among land use and management systems is that the variability in an inherent soil property is less than the change in the property induced by differences in land use and management. Thus, differences that are observed in properties of a specific soil under different land use or management conditions reflect changes that can be attributed to the effects of land use or management.

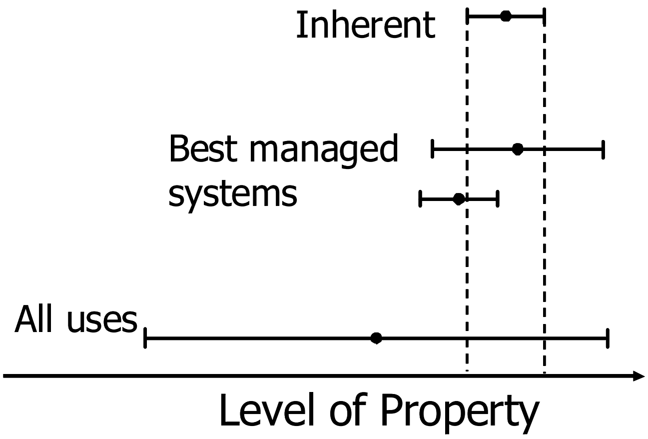

Figure 1. Diagram representing levels of a soil property observed in a non-degraded inherent condition, under best management, and under a range of management conditions.

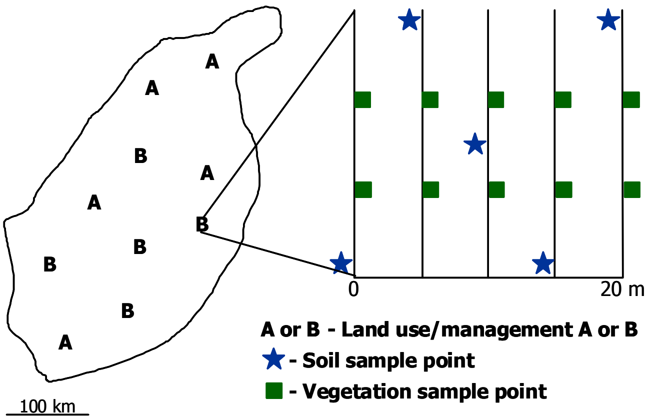

Figure 2. Plot layout for DSP data collection and sampling. The left portion of the figure depicts the geographic extent of Soil X. A and B represent two managements being evaluated for Soil X. The enlargement (right portion of the figure) depicts the sampling layout within each plot for grassland and cropland.

The use of comparison studies to evaluate DSP meets NCSS needs because of their efficiency and simplicity, both of which are necessary for a large nationwide inventory. Space for time comparison studies overcome many of the constraints associated with long-term monitoring including land use and management changes that may occur on private land over the monitoring period. It is recognized that comparison studies may fail to differentiate natural variability from that resulting from differences in land use and management history (Pickett 1989), that the underlying assumption that all locations were initially equal is difficult, if not impossible, to verify, and that the "management legacy" is difficult to assess since the complete history of any site is seldom known (Foster *et al.* 2003). These limitations can be minimized by restricting sampling to a well-defined soil and replicating data collection at multiple spatial scales. Use of a conceptual model that describes expected changes in properties with deviation from the native condition will also help to separate land use and management effects from natural variability and will permit identification of critical management thresholds useful to land resource managers.

The NCSS DSP inventory focuses on land use and management systems that have been in place for sufficient time, such that the functionally important properties are at relative steady-state. It also focuses on understanding properties under two conditions; 1) a reference management system which is either non-degraded native conditions or where such conditions no longer exist, a well-managed alternative system and 2) an attainable condition that represents the soil under best management (Figure 1). The requirement for steady-state conditions is necessary to gain meaningful data from a one-time evaluation of soil conditions related to a specific disturbance regime and to relate management systems to optimal soil functional levels. The steady-state restriction, however, limits information gained concerning rates of change which is better obtained through long-term monitoring (Richter *et al.* 2007). The DSP effort will yield information on how the soil functions under differing land use and management and how the soil can be expected to function if best management is imposed over its aerial extent.

A hypothetical distribution of plots and the standard plot layout for collecting statistically valid data within operational conditions necessary for a nationwide effort are illustrated in Figure 2. Two levels of stratification are imposed in site selection; all plots occur on a single soil (confirmed in the field) and the land use and/or management has been in place for sufficient time to achieve steady state (derived from local knowledge). An additional requirement is that, if possible, plots sites extend over the full extent of the soil's occurrence in order to capture variability due to environmental gradients. In the absence of variance data to develop more precise sampling requirements, a minimum of five replicate plots and five systematically located soil replicates per plot are sampled (Figure 2). The plot design also accommodates collection of functionally important plant community features including foliar cover, species composition, site indices, and productivity data. The numbers of plots per soil and plot replicates will be evaluated and modified as data are collected.

Properties evaluated

Although soil function cannot be measured directly, certain soil properties that reflect complex processes and functions can be measured and used to represent soil functions (Karlen and Stott 1994). For the NCSS DSP program, a functionally important difference in DSP is considered to be a difference sufficiently large to reflect or cause an important difference in soil or ecosystem function. Minor differences in management systems are expected to translate to minor differences in DSP. Thus, procedures have been or are being developed to group land use and management systems into categories expected to cause functionally important differences in soil properties.

For a nationwide inventory of DSP, it is not possible to evaluate all properties that may change in response to differences in land use and management. Those evaluated must fit within the constraints of the large project including availability of laboratory resources, the requirement that the data and samples must be collected in a single visit to the site, and the expertise and availability of soil scientists collecting the data and samples. Properties that fit within these constraints and that satisfy the objectives of the effort were selected with a multi-step process that employed expert knowledge. The steps in the process included 1) developing criteria for rating soil properties, 2) selecting properties reported to change with disturbance across a wide range of soils, 3) having experts rank the properties in order of importance and according to the rating criteria, 4) prioritizing the properties statistically, and 5) reviewing the rankings for potential bias due to evaluators area of expertise (Wills *et al.* in review).

The criteria used to select and rank properties are as follows (Wills *et al.* in review):
1. The property is sensitive to disturbances or management.
2. The property clearly reflects the status of processes that are important for a number of soil or ecosystem services and functions or is a key indicator of one particularly important service or function. The relationships between the property and the processes, functions or services it reflects have been quantified for a wide range of soils.
3. The property is relatively insensitive to daily or seasonal fluctuations in environmental conditions of moisture, temperature, and light, or such fluctuations are well-understood and can be quantitatively predicted.
4. The property is easy to measure accurately and at the necessary level of precision by different people or by the same person at different times.
5. The cost, including field and laboratory expenses and time, necessary to obtain the required number of measurements is low.

A rating tool that included 19 properties meeting criteria 1 (Figure 3) was developed and sent to 41 scientists with expertise in soil survey (25), soil biology (7), agronomy (6), forestry, range science, and hydrology (1 each). Each expert ranked the properties in order of importance and rated each property relative to its fit to each of the ranking criteria 2 thru 5. From these data, a weighted average score was calculated for each property (Figure 3), and the eight highest ranked properties; organic C, pH, electrical conductivity, bulk density, soil structure, wet aggregate stability, total N, and soil stability; were selected as the minimum dataset for each DSP project. In addition to these eight properties, additional properties that are easily measured, e.g. plant available P, or are potentially diagnostic of soil degradation or aggregation, e.g. active C, will be evaluated in selected projects and may be added to the minimum dataset. Local soil scientists and cooperators involved in data collection are encouraged to collect additional data at the sites for evaluation of locally important questions.

Conclusions
The U.S. NCSS is initiating a standard protocol to produce a nationwide inventory of dynamic soil properties subject to change in response to changes in land use, agricultural and silvicultural management, and other disturbances. The inventory is based on comparison of major management systems in optimal condition, each reflecting a specific disturbance regime, through substitution of space for time. Resources available preclude evaluation of all soils, land uses, and management systems that occur in the U.S. Thus, initial efforts target Benchmark and other important soils and broad land use and management groups to build a dataset that can be used to represent other similar soils. These data and that collected over the longer term will provide the information needed to encourage and implement wise land management practices and policies to sustain important soil functions. The full protocol the NCSS DSP program is available in Tugel *et al.* 2008.

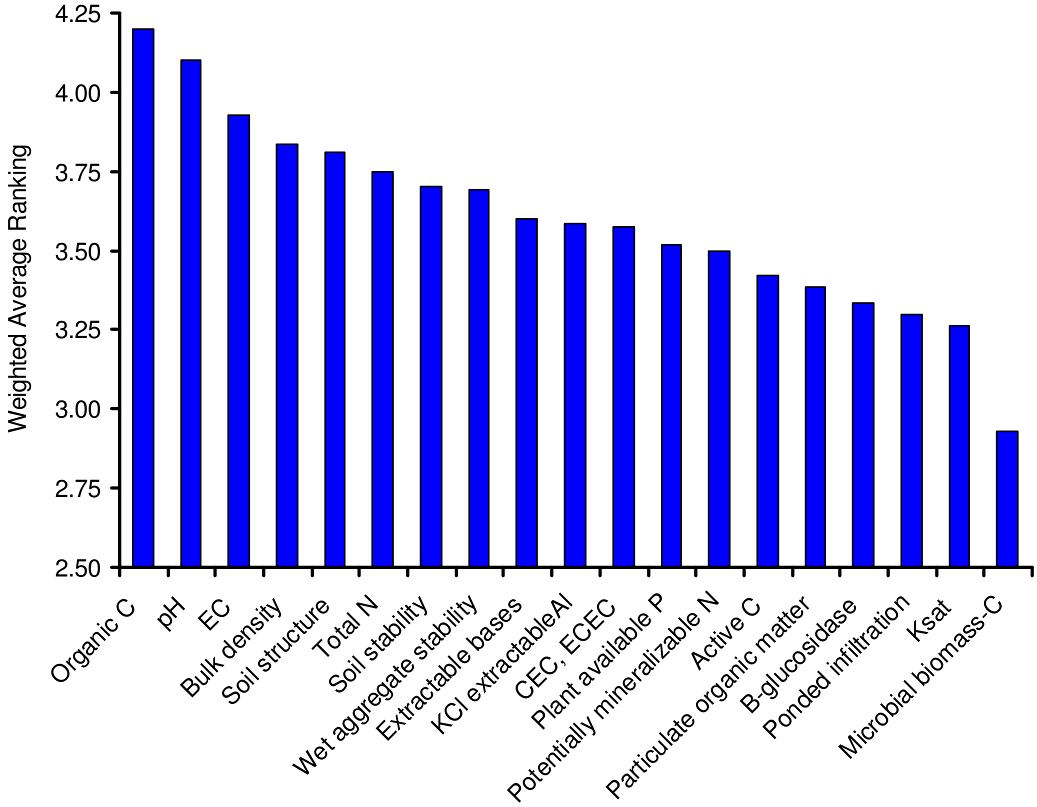

Figure 3. Weighted average ranking for properties considered in DSP.

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Soils in space and evaluation of their changes in time: experience in studying the soils in two regions of Russia.

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Abstract
Under consideration are results obtained in an evaluation of soil changes that have taken place in the second half of the 20th century in two regions of Russia. The regions are characterized by different natural conditions, soil cover and effects of human activities. The soils have been thoroughly studied in space at several hierarchical levels using traditional and digital mapping techniques, thus reflecting vertical differentiation of the soil profile, randomized variety within the soil area, spatial regularities of changes in soil combinations.

Keywords
Solonetz complex, chernozem, short-term evolution, soil in space and time, monitoring.

Introduction
The current stage of soil science development has risen to a qualitatively new level in evaluating or monitoring evolution of soils and geosystems for the first hundred years owing to detailed studies carried out by trained specialists in the 20th century.

Such an approach (retrospective monitoring) allows study of processes, and rate of soil changes. At the same time, the study of present soil changes in time for the period have accumulated vast experimental data obtained during previous decades, depending on natural peculiarities of soils and environmental conditions as well as available information presented by the other authors.

The objective of this paper is to demonstrate the necessity of studying soil distribution in space for solving the tasks relating to the study of present tendencies for soil changes over time.
This study has been carried out in two regions of Russia characterized by different natural conditions, soil cover and human interventions: (1) soils of the light-chestnut solonetz complex within the Pre-Caspian lowland and (2) agro-forest landscapes predominated by chernozems in Kamennaya Steppe.

Objects and methods
The solonetz complex in the Pre-Caspian lowland has been studied at the territory of Dzhanybek experimental station located in 30 km north of Elton Lake in the Volga and Ural interfluve on frontier between Russia and Kazakhstan. Since the time of its foundation in 1949 the complex investigations have being conducted with the aim at compiling maps of microrelief (Mozeson 1955), vegetation and soils (Kamenetskaya *et al.* 1955). The soils have been studied in detail (Rode, Polskiy 1961) including water and salt regimes in them (Rode, Polskiy 1967; Maximyuk 1966). The solonetz complex consists of solonetzes and earth hummocks formed by animals under wormwood-cypress vegetation in convex, light-chestnut solonetzic and non-solonetzic soils under matricary-esparto grass associations on microslopes and meadow-chestnut soils under herbaceous vegetation with concave relief (Rode, Polskiy 1961). The parent material is represented by loess-like loams of marine origin (the early Khvalynian transgression of the Caspian Sea). In the 1950s the groundwater table was at a depth of 6-7 m (Rode, Polskiy 1961), since the 1980s it is 3.5-5 m deep. This territory is used for extensive grazing.

Investigation methods of the solonetz complex
The territory under consideration was surveyed in detail according to a grid of 1-2 m and additional special points (extremum, folds) by theodolite, level and metal measuring tape (50 and 100 m) with the aim at identifying a test area which has been studied earlier and overlaying the obtained factual picture of microrelief with the maps published in the 1950s. Changes in microrelief for 50 years were measured twice in 1950 and 2004 by algebraic subtraction of digital relief models in the grid 0.5 m. Independent from each other the soil and geobotanical surveys at a scale of 1: 200 were carried out in some key plots. The data obtained in 2000 observation points were comprehensively analyzed to gain a spatial overview of changes in the thickness of different soil horizons with the kriging approach. The salt content in soils to the groundwater depth was

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determined in a 1:5 water extract, a potentiometric method was employed to measure the activity of sodium ions in paste with 40% of moisture.

The main results obtained to study changes in the solonetz complex in the Pre-Caspian lowland in the second half of the 20th century.
1. Changes taken place in virgin solonetz complex for 50 years reveal a combination of all-round evolution of its different components on positions closely located in space. Such trends are rather peculiar in dependence on the microrelief type. The half a century of changes in the solonetz complex are mainly induced by the groundwater rise from 6-7 to 4.5-5 m.
2. General pattern of relative spatial position and configuration of the most contrasting microrelief elements (convex and concave) remains almost unchanged, as is evidenced by field observations and overlaying of former and newly obtained cartographic materials.
The surface of the virgin solonetz complex shows some changes in microrelief of watershed and flat types. Stable unchanged positions of microrelief occupy about 50% of the total area, thus forming a tracery net-like frame with mosaic inclusion of some areas, which became lower to 3-20 cm (30-33%) and elevated up to 3-25 cm (14-18%). This is the action of several mechanisms responsible for changing the microrelief, which and functioned simultaneously and/or in consecutive order.
It was possible to observe uneven surface sinking to 10-20 cm in microrelief of radial-convex type (80-90% of the total area) probably due to rising groundwater table and capillary fringe resulted in moistening of dry salt horizons in the aeration zone.
3. As distinct from the widely used opinion about homogeneity of the solonetz complex and homogenous inter-convex watersheds at the given territory (Rode, Polskiy 1961) the obtained results serve as evidence of its heterogeneity and dependence on microrelief type.
Statistical indices of soil horizon depth in the main components of this solonetz complex reveal no changes for the half of century.
4. It is evident that the surface carbonate light-chestnut (solonetzic and non-solonetzic) soils and crust solonetzes without features of mixing the soil material by living organisms should be considered as newly formed components of this soil complex. These soils have a convex surface with the fine cracks due to accumulation of calcium carbonate in the initially carbonate-free topsoil.
5. At present, it is feasible to find any soil within the virgin solonetz complex at different spatial positions and under different plant associations with a probability of 0.3-0.8.
6. One third of inversions are connected with changes in microrelief that have taken place over 50 years, while two thirds of inversions are confined to stable unchanged areas, thus testifying their mobility during this period. At the territory under study the rate of change in microrelief seems higher than for soil morphology but it is comparable with the changing rate of the salt state in soils.
7. In view of what has been said one should assume that the virgin solonetz complex in the Northern Pre-Caspian lowland correspond to a *nonstationary, fluctuent regime of evolutional development*. The only possible reason is periodical cycles of groundwater fluctuation (duration from a few ten years to 1-1.5 century). Moreover, the solonetz complex is invariant according to the relative ratio between the main soil and vegetation components as well as its principal configuration. By this reason, a part of the territorial position is occupied by components corresponding to a quasi-stationary regime of functioning or close to it. In the other part of Territorial position the same components are found at different stages of their development. In total, we observe a mosaic picture of spatial arrangement of quasi-stationary soil components and those developed through different trends, which relaxes the close connection between soil, relief and vegetation.

<u>The agro-forest landscape of Kamennaya Steppe</u> is situated in central part of the East-European plain in the interfluve between Bityug and Khoper tributaries of the Don river within the transitional zone from Kalatch upland to Oka-Don lowland in the southeast of Voronezh region. This is one of three test areas taken for research by a special expedition under V.V. Dokuchaev's guidance in 1892. In the XX century this test area was used as a polygon for elaborating and approving general strategy of agricultural management under conditions of frequent droughts in the chernozem zone of Russia.
Since 1892 there have existed meteorological and since 1933 hydrogeological stations. Repeated soil investigations have been conducted at this territory (Glinka *et al.* 1894; Tumin 1930; Pershina *et al*. 1947; Antipov-Karataev *et al*. 1963; Aderikhin *et al*. 1984; Khitrov 2009). The vegetation (Maltsev 1923), hydrogeology, the groundwater regime (Basov, Grishchenko 1963) and the landscapes (Milkov *et al.* 1971, 1992) have been also studied in detail. The initial steppe landscape was found to be transformed into an agro-forest one as affected by human interventions. At present, the major part of this landscape is under crop

surrounded by protective forest belts. The surface runoff was completely regulated and transformed into an underground one. The soils derive from loess-like clays as the parent material.
The soil cover is represented by typical, ordinary, leached and zooturbated chernozems in flat watersheds and sloping areas as well as by solonetzic chernozems, chernozem-like meadow solonetzes and meadow-chernozem soils in depressions.

Investigation methods in Kamennaya Steppe.
43 key plots including above 950 observation points helped compile a map of the soil cover structure in Kamennaya Steppe as a GIS-project. The relief in 6 key plots from 0.2 to 6 ha in size has been surveyed at a scale of 1:1000. Some key plots were specially confined to that place, where soil profiles were analytically studied by other specialists 50 years ago. They embraced not only the soil profile described in detail earlier but also adjacent areas. Digital maps have been compiled to show the microrelief, the soil cover, changes in the depth of soil horizons and analytical characteristics of soils taking into account the soil profile form. The main objectives of this study were aimed to identify spatial regularities in distribution of soil properties and to select factual objects for evaluating the changes in properties of the same soil over time.

Some conclusions of this study
1. The soil cover of flat watersheds in Kamennaya Steppe is represented by a soil combination consisting of three components of chernozems including typical (40-70%), zooturbated (20-50%) and leached (10-15%) ones; it promotes the spatial regularities of changes in the depth of soil horizons and should be certainly taken into consideration for evaluating soil changes in time.
2. It was established that after 50 years the bulk density of the middle part of arable layer (10-20 cm) became higher from 1.0 to 1.2 g/cm^3 as well as a part of the humus horizon at a depth of 35-50 cm disturbed not by tillage but by animals. This is explained by repeated use of heavy agricultural machines in the 1960-1980s.
3. For 50 years a tendency to decreasing organic carbon content in the arable horizon (0-22 cm) from 5.2 to 4.6% was statistically substantiated.
4. After discontinuance of annual plowing 55 years ago the morphological configuration of hydromorphic chernozemic solonetz revealed its regeneration within the former arable layer, thus preserving physical-chemical features of the solonetz process (EC_{se} < 2-4 dS/m and ESP > 10%) for the period of observations since 1952.
5. Application of gypsum and manure exerted a short-term (not more than 5-7 years) positive effect. 50 years later it was impossible to find any morphological and analytical indication of this effect.
6. Due to mulching of the solonetz surface by materials taken from the humus horizon of chernozem the regeneration of morphological solonetz properties seemed to be slower at least for 30-50 years. However, under hydromorphic conditions the mulching layer acquires the physical-chemical and morphological properties of solonetz soils.
7. Monitoring of the soil state as based upon data obtained at different times within an area exceeding some square kilometers doesn't allow conclusions about soil change in time. The difference in statistical distribution is conditioned, on the one hand, by the diverse soil cover and combination of soil-forming factors over the large territory under study, on the other hand, by the arrangement of observation points to solve definite tasks in every period of observation as well as by methods of statistic and criteria for specific objectives and subjective data. Small test areas are required to carry out a detailed survey at different periods of time and to obtain information about the diversity, regularities of the soil cover organization and statistical indices of properties of soils in the test area.

Conclusion
When evaluating soil changes over time it seems reasonable to use information about the distribution of soil properties at several hierarchical levels:
- differentiation of soil properties throughout the vertical profile in every genetic horizon (but not in layers with a fixed depth);
- randomised variety of soil properties within the rather homogeneous soil cover or their trend changes according to the interaction the studied soil with adjacent soils;
- qualitative and quantitative difference of properties in various soils as components of any soil combination;
- different relationships between soil components in identical (as well as different ones) soil combinations and the possible quantitative difference in the same property of soils of the same type in different soil combinations.

Acknowledgements
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Spectral effects of biochar on NIR and mid-IR spectra of soil/char mixtures

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Abstract
The objective was to investigate the effect of adding biochar to soil on the spectra of mixed biochar and soil in the near- (NIR, 10,000 to 1000 cm^{-1}) and mid- infrared (mid-IR, 4000 to 400 cm^{-1}). Biochar produced from the Ingá (*Ingá edulis* Mart.) tree at 3 temperatures (400, 500 and 600^oC) was added to soil by mixing or grinding together and at 0 to 6% by wt. Results indicated that the addition of biochar to soil results in spectral distortions which tend to distort/mask, the signature of the original soil. These effects are very evident when applying spectral subtraction to the resulting spectra, e.g., the spectra of biochar and soil are not additive, i.e., (A + B) <> (A+B). Significant distortions in the NIR were seen between 5000 and 4000 cm^{-1} where significant information on soil C is found and could have profound effects on attempts to determine total soil C (TSC) by NIR in biochar amended soils. Similar distortions were found in the mid-IR, but primarily in the silica spectral region, and while making spectral interpretation more difficult, may have less effect on the ability to develop calibrations for TSC in biochar amended soils, but further research will be needed in both spectral regions to answer this question.

Key Words
Biochar, Mid-Infrared, Mid-IR, Near-Infrared, NIR, soil

Introduction
Biochar is the C rich product produced when biomass is heated in the absence of oxygen (pyrolysis). The liquids produced, often called bio-oils and along with the gaseous products can be burned or used as feedstock for bio-based fuels or chemicals (Biochar, 2009). Much of the interest in biochar has come from research on the *terra preta* or dark earths found in the Amazon Basin (Lehmann and Joseph, 2009). The production and use of biochar is being highly touted as a way to sequester C in soils and to improve soil quality and productivity (Laird, 2008; Bracmort, 2009), but as discussed at the recent N. Amer. Biochar Conf. (http://cees.colorado.edu/northamericanbiochar.html, August 2009), there are still questions to be resolved. An investigation into the effect of biochar on C mineralization in soil, using near-infrared (NIR) and mid-infrared (mid-IR) spectroscopy showed spectra to exhibit some very unexpected properties. The objective of this investigation was to investigate the spectral properties of mixtures of biochar and soils.

Methods
Biochar production Biochar was made from the Inga tree species (*Ingá edulis* Mart.) at two temperatures (400 and 500 ^0C). Ingá biomass from a 7-yr old tree was collected from secondary forest growth at the National Institute of Amazonian Res. (INPA) Tropical Fruit Culture Expe. Station. Woody material was chosen so that similar diameter samples (other biochars not discussed) were placed in the pyrolysis chamber, in place of selecting material from similar locations on the trees. In the case of Ingá, the branches were used and biochar made from fresh material in the pyrolysis furnace at the Cellulose and Charcoal Lab., Forest Prod. Div., INPA, Manaus, BR. Slow pyrolysis was accomplished in a furnace of refractory brick with a 20L capacity, with samples brought to temperature over 2h. After reaching temperature the furnace was turned off and allowed to cool. After combustion, samples were ground to pass a 2mm sieve. The biochar yield, as well as yield of associated products that characterize the feedstock and biochar is provided in Table 1.
Spectroscopy Spectra were obtained in the near- (NIR, 10,000 to 4000 cm^{-1}, quartz beamsplitter, lq. N_2 cooled InSb detector, sulfur background) and mid-infrared (mid-IR, 4000 to 400 cm^{-1}, KBr beamsplitter, Peltier cooled DTGS detector, KBr background) ranges using diffuse reflectance on non-KBr diluted samples (Digilab FTS7000 Fourier transform spectrometer, Varian, Inc., Lake Forest, CA, equipped with a Pike Autodiff autosampler, Pike Technologies, Watertown, WI). Samples consisted of biochar mixed with soil (Millhopper sand, pH 5.2 or pure sand) by gentle swirling or by grinding the previously mixed materials together in a mortar and pestle. Some samples were also stored in a desiccator over water for various periods. Spectra were processed using GRAMS AI Ver. 8 (Thermo Fisher Scientific, Waltham, MA) and Panorama Ver. 1.2 LabCognition Gmdh & Comp., Cologne, Germany) software. Spectra of multiple sub-samples were often taken

and averaged, but no differences were found between replicate spectra indicating homogenous samples.

Results

Table 1. Yield of biochar, wood tar and vinegar and C-mineralization results for Inga biochar.

	Sub-products of Pyrolysis					C-Mineralization Results		
	Wood, Dry	Biochar	Biochar	Wood	Wood	Net C Released mg/g		
Temp.	Wt (Kg)	(kg)	(%)	Tar (ml)	Vinegar (L)	Max.	Day	Day 162
400	13.3	3.4	25.6	0.1	9.3	4.93	35	-47.41
500	13.6	3.4	25	0.1	9.3	1.83	21	-55.68

As shown in Table 1, the yield of char and other products can vary significantly with temperature. Significant differences between different biomass sources also existed, but are not part of this discussion. Results from experiments examining the amount of C mineralized as CO_2 over a period of 162 days also indicate that compositional changes vary with temperature and not necessarily in a linear fashion.
In Figure 1, the NIR spectra of Inga 500 ^{0}C biochar, soil, and a mixture of 6% biochar by wt. in soil are shown. As can be seen, the addition of biochar appears to mask the spectral signature of the soil even when only mixed by swirling. As shown in Figure 2, spectral subtraction of the biochar from the biochar-soil mixture does not produce the spectrum of that of the original soil as would be expected. As all materials were dry powders, the effects must be physical such as coating of the soil particles by biochar. This was still not expected as the biochar is present at only 6% of the concentration of the soil and NIR radiation can penetrate at least a mm deep in such samples. Initial thoughts that the biochar might also be causing dehydration effects were shown not to be so as demonstrated for samples stored over water for up to 11 days which showed only a modest change in the water bands (Figure 3). As shown in Figure 4, grinding the char and soil together after mixing only intensifies the effect supporting the concept of a physical coating of soil particles by biochar which NIR radiation does not seem to penetrate well.

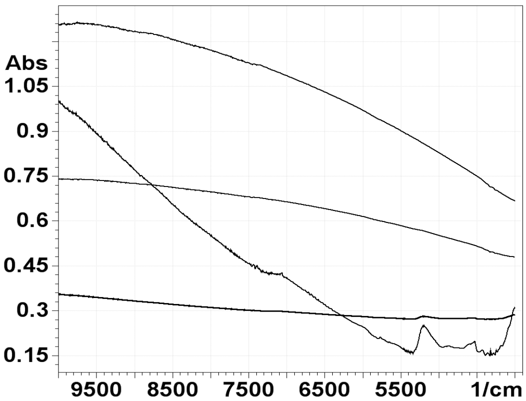

Figure 1. Near-infrared spectra of (Top to bottom) Inga 500 C biochar (IG500), soil + 6% IG500, soil, and normalized soil spectrum.

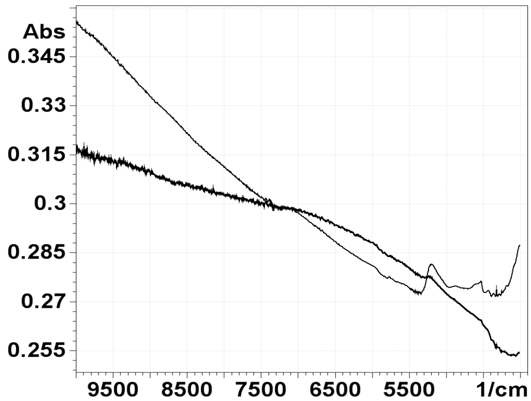

Figure 2. Near-infrared spectrum of soil (Top) and spectrum resulting from subtraction of spectrum of Inga 500 C biochar (IG500) from spectrum of 6% IG500 in soil (Bottom).

As demonstrated in Figures. 6-8, similar spectral distortions were found in the mid-IR spectral region but primarily in the 2300 to 1700 spectral region where silica strongly absorbs (Figures. 5 and 6). Interestingly, the same results were not as apparently in work using sand and biochar (Results not presented). Again the distortions are readily apparent if spectral subtraction is attempted (Figure 7), but grinding does not seem to have any additional significant effects over just simply mixing (Figure 8). This is the opposite of what was seen in the NIR where grinding the mixed materials together intensified the effects. Thus overall, the effects in the mid-IR of adding biochar to soils appears to be less than in the NIR despite the fact that NIR radiation penetrates much deeper than mid-IR radiation and thus any particle coating effects should be much greater in the mid-IR. As silica does not absorb in the NIR, the effects seen in the mid-IR are not expected in the NIR, but the effects on the C-H and N-H regions seen in the NIR between 5000 and 4000 cm^{-1} might well have been expected in the mid-IR. These results overall seem to support a physical spectral effect, which nevertheless could have significant effects on calibration development especially for remote sensing using visible (Not examined here) or NIR spectra.

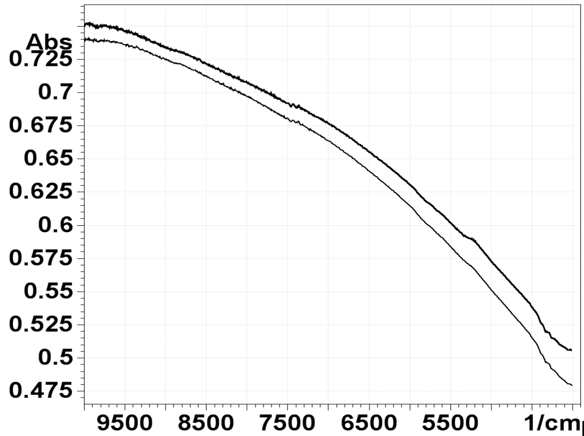

Figure 3. Spectra of 6% Inga 500 biochar in soil (Bottom) and after storage for 11 days over water (Top).

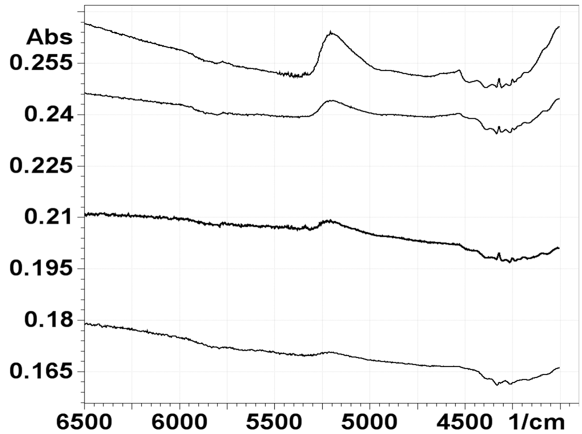

Figure 4. Spectra of (Top to bottom): a. non-ground, and b, ground soil, c. from spectral sub. of Inga 400 C biochar spec. from spec. of 6% IG400 mixed with soil, and d. ground with soil.

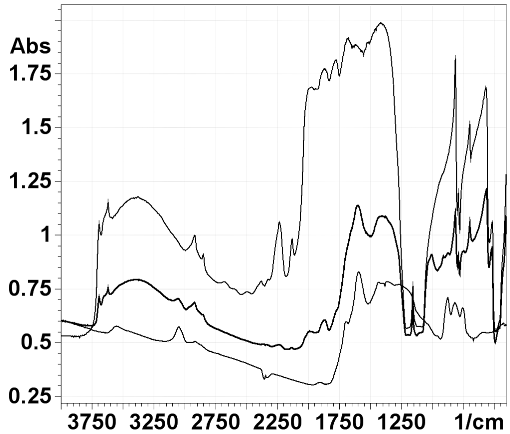

Figure 5. Mid-infrared spectra of soil, soil + 6% Inga 500 biochar (IG500) and IG500 (Top to bottom).

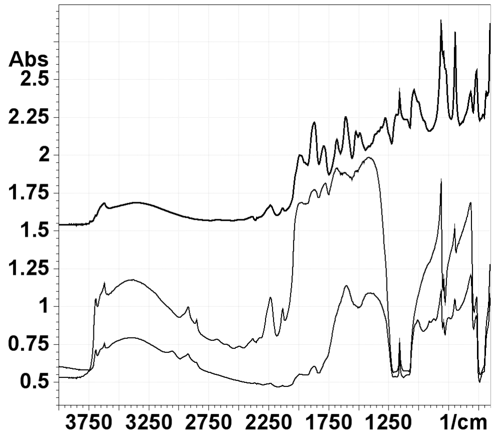

Figure 6. Mid-infrared spectra of (Top to bottom): 50% silica in KBr (Shifted + 1.5 A), of soil, soil + 6% Inga 500 biochar (IG500).

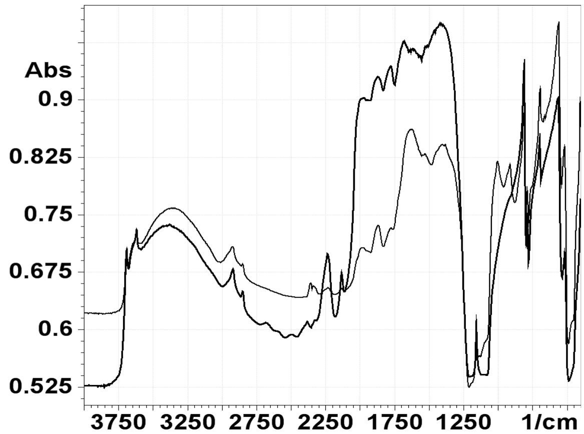

Figure 7. Normalized mid-infrared spectra of soil and from spectral sub. of Inga 400 C biochar spec. from spec. of 6% IG500 mixed with soil.

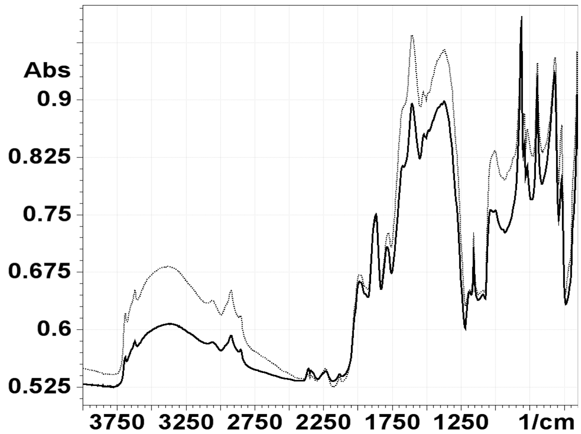

Figure 8. Spectra of 6% IG400 mixed with soil (Dotted line) and ground with soil (Solid line).

Conclusions

Results on biochar amended soils showed that the addition of biochar to soil results in spectral distortions which tend to mask, and or distort, the spectral signature of the original soil carbon. These effects are particularly evident when applying spectral subtraction to the resulting spectra in that the combined spectra of biochar and soil are not additive, e.g. spectrum of (mixed char + soil) – spectrum of char <> spectrum of soil, as should be the case if there were no physical or chemical interactions. These effects are particular evident in the NIR region between 5000 and 4000 cm^{-1} where significant information on soil carbon is found and could have profound effects on attempts to determine total soil carbon by NIR in biochar amended soils. Similar distortions were found in the mid-IR, but primarily in the region dominated by silica, and while making spectral interpretation potentially more difficult, may have less effect on the ability to develop calibrations for soil carbon in biochar amended soils, but further research will be needed in both spectral regions to answer this question. Interestingly, results using sand in place of soil with mid-IR spectra did not show the same effects. The difference between the NIR and mid-IR results are curious as one would expect less rather than greater effects in the NIR due to the greater ability of NIR radiation to penetrate deeper into the samples. As silica does not absorb in the NIR, the effects seen in the mid-IR are not expected in the NIR, but the effects on the C-H and N-H regions seen in the NIR between 5000 and 4000 cm^{-1} might well have been expected in the mid-IR. These results overall seem to support a physical spectral effect, which nevertheless could have significant effects on calibration development especially for remote sensing using visible (Not examined here) or NIR spectra.

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Sustaining soil carbon and nitrogen pools for future cereal production

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Sustainability of the agricultural system is the main concern for food security in the context of a rapidly growing population. The significance of soil organic matter accumulation for sustaining agriculture has long been recognized. In general, an increase in the organic matter content of agricultural soil improves soil quality, crop growth, and system sustainability, and reduces pressure on forest, thereby reducing green house gas emissions to the atmosphere.

Under a given climate, cropping system and management, the soil C and N tend to maintain an equilibrium. Following cultivation related changes, the soil moves gradually toward new C and N values and tries to establish a new equilibrium (Figure 1). The rate of increase or decrease depends on diverse factors such as climate, cropping system, soil characteristics specially texture, N fertilization, organic manure addition, extent of soil tillage, and soil moisture. Depending on the gains and losses of C and N, the functioning of agro-ecosystem can be considered not sustainable, sustainable or highly sustainable. The C and N pools and their turnover rate are different in the agro-ecosystem and natural ecosystem. The loss of C and N resulting from cultivation can have serious implications for chemical, physical, and microbial fertility of soil. The lack of accurate information on effects of continuous cultivation including fertilization specially in tropics has long been considered to be a major knowledge gap. Recently, some systematic efforts have been made to assess the interaction of soil and fertilizer N of soil N and C content and dynamics, and the impact of synthetic fertilizer on depleting soil N reserves is argued (Mulvaney *et al*., 2009).

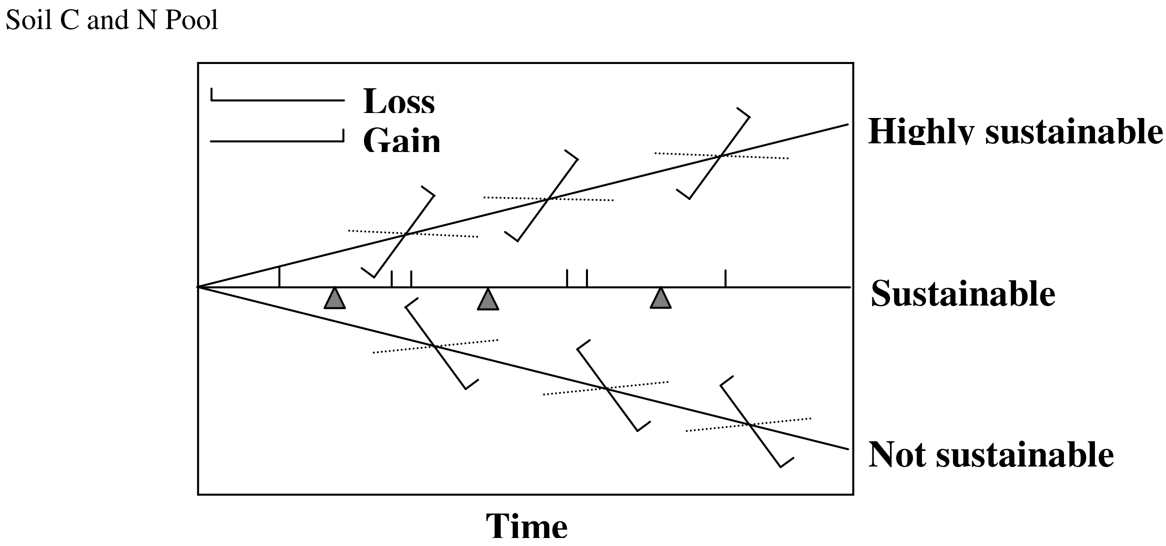

Figure 1. Soil C and N equilibrium concept

Diversified agro-climatic regions of South Asia from subtropical plains to warm-temperate provide opportunities to produce a wide range of agricultural commodities. The most common cropping systems are rice-rice, rice-wheat, rice-maize and maize-wheat. The demand for cereals is estimated to be 50 to 70% higher by 2050 to feed 9.3 billion people (Wood *et al*., 2004). However, cereal based cropping system in South Asia are faced with stagnation in crop productivity. The cultivation of marginal lands together with agricultural intensification on existing land have resulted in a decline in productivity (Ladha *et al*., 2003). Diverse factors including gradual decline in soil C have been attributed to the negative trends in agricultural productivity.

This paper examines the (a) effect of continuous cultivation and fertilization in relation to soil crop management practices in major cereal based cropping systems on selected soil quality parameters specially soil C and N content, and (b) diverse ways to sustain inherent soil C and N levels. From a large number of long-term experiments and comparison of soils from farmers fields with those of uncultivated fields in Asia, we found widespread decline in soil C and N content. However, declines in total soil C and N tend to be more pronounced and the base levels are lower in crop rotations where soils either remain aerobic (uplands) or goes through cycles of aerobic-anaerobic conditions (upland-lowland). Soils under crop rotations such as rice-rice (lowland-lowland) where anaerobic conditions remain during most of the growing period maintain high base level of soil C and N and often do not show a decline in C and n content. In fixed plot long-term experiments, crop rotations with continuous application of chemical N fertilizer tend to have more loss of soil C, a trend similar to long-term plots without fertilizer-N input. However, plots which received organic amendment with or without N fertilizer tend to show gains of soil C. The ways and means to sustain soil C and N would include: a) minimizing soil disturbance, b) avoiding cycles of soil flooding/drying, c) avoiding dry fallow, d) using quality residue, e) replenishing soil nutrients, f) applying plant need based N, and g) subsurface application of fertilizer N.

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Temporal changes in topsoil qualities of dairy pasture and maize cropping sites in the Bay of Plenty Region, New Zealand

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Abstract

Temporal changes in topsoil qualities of dairy pasture and maize cropping sites were monitored periodically over a ten-year period. Results indicate that for both sites, many of the topsoil quality parameters were being maintained and these are within the provisional target values set by Landcare Research New Zealand for production and/or environmental criterion. However, the steady increase in the levels of anaerobically mineralisable N and Olsen P in dairy sites is a concern. High values of anaerobically mineralisable N could potentially lead to increased nitrate leaching while high values of Olsen P could lead to P-rich sediments polluting water bodies.

Key Words

Soil quality, soil health, soil quality monitoring.

Introduction

Environment Bay of Plenty (the Bay of Plenty Regional Council) has been collecting soil quality or soil health data since the late 1990's when it participated in the 500 Soils Project involving the various regional councils of New Zealand (Sparling and Schipper 2004). A total of more than 70 soil quality sites have been progressively established under various land uses. The sites were categorised by land use which include: cropping (maize), dairy, sheep and beef, deer, kiwifruit and forests (indigenous and plantation). The number of sites sampled for each land use category was proportional to the area of land use. Sampling frequencies differ and depend on the degree of soil disturbance or cultivation. Thus, cropping sites are sampled every 3 years, dairy, deer, sheep and beef, and kiwifruit sites every 5 years, and forest sites every 10 years. The status of soil quality in the region has been reported periodically by Landcare Research (Sparling 2001; Sparling and Rijkse 2003; Sparling 2004; Sparling 2005; Sparling 2006a; Sparling 2006b). The objective of this report is to discuss the changes in topsoil qualities periodically sampled dairy pasture and maize cropping sites over a ten-year period.

Methods

Soil sampling and analyses

Twenty four soil quality sites consisting of 19 dairy pasture sites and 5 maize cropping sites from previously established sites were resampled in 2009. The standard protocol for New Zealand soil quality sampling was followed (Sparling and Schipper 2004). A 50-m transect was established in each site. For chemical analyses, topsoil samples (0-10 cm) were collected with a step-on soil sampler at 2-m intervals along the 50-m transect. The 25 individual samples collected were bulked and mixed thoroughly in a plastic bag. For physical analyses, three stainless steel soil cores (10 cm diameter, 7.5 cm high) were taken at 15-, 30- and 45-m along the transect. It should be noted that the standard 0-10 cm topsoil sampling depth represents a compromise for both land uses since dairy pasture soils are normally sampled at 0-7.5 cm while maize soils are sampled at 0-15 cm.

The samples were submitted to Landcare Research laboratories (Hamilton and Palmerston North) for the analysis of seven standard soil quality indicators, namely: pH, total carbon (C), total nitrogen (N), anaerobically mineralisable N, Olsen phosphorus (P), bulk density and macroporosity. The C/N ratio was obtained by dividing total C with total N. For maize sites, undisturbed topsoil samples were also taken and submitted to Plant and Food Research in Lincoln for the analysis of aggregate stability. Aggregate stability was expressed as a mean weight diameter in mm. All laboratory analyses were performed following the methods described in Sparling *et al*. (2008).

Data analysis

Mean values of topsoil qualities by land use class were compared with the "target" or "desirable" qualities set as provisional soil quality target values for New Zealand by Landcare Research (Sparling *et al*. 2008). These standards are grouped according to land use and/or soil classification with production and/or environmental criterion. Aggregate stability results from maize sites were compared with the standard given in Beare *et al*.

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(2009). Previous results from dairy and maize sites reported by Landcare Research were used in order to show changes over time.

Results

Temporal changes in topsoil qualities of dairy sites

Table 1 shows the trends of topsoil qualities of dairy sites over a ten-year period. There was no significant change in topsoil pH with time. The mean pH values in each year lie within the provisional target of 5.0 to 6.6.

Table 1. Temporal changes in topsoil qualities of dairy sites with respect to pH, total C, total N, C/N ratio, anaerobically mineralisable N, Olsen P, bulk density, and macroporosity (n=19)

Soil Quality	Year			
	1999/2000	2004	2009	P value
pH	5.65	5.86	5.79	0.181
Total C (%)	7.65	6.87	7.81	0.473
Total N (%)	0.63	0.64	0.73	0.294
C/N ratio	11.98	10.86	10.72	0.015
Anaerobically mineralisable N (mg/kg)	72	155	256	<0.001
Olsen P (mg/kg)	67	87	97	0.095
Bulk Density (t/m^3)	0.87	0.94	0.85	0.210
Macroporosity (%)	9.40	7.78	9.97	0.378

There was no significant change in total C. Mean values for each year are above the provisional target of >2%. For total N, a slight increase was observed . Mean values for each year are within the provisional range of 0.25 to 0.70%. There was a very slight decrease in the C/N ratio. The mean values lie within the provisional optimal target range of 8 to 12 for pasture soils (production criterion) and 7-30 (environmental criterion).

A steady increase in anaerobically mineralisable N over a ten-year period was observed. Mean anaerobically mineralisable N in 1999/2000 was 72 mg/kg and 155 mg/kg in 2004. These values are within the provisional target range 50 to 250 mg/kg. However, in the 2009 sampling, the mean value was 256 mg/kg exceeding the upper limit of the target range. This is 3.6 times the initial value in 1999/2000. This reflects a continual fertiliser N input in the pasture soils. If this trend continues in the future, concern for increased nitrate leaching will become a more significant issue for this land use. Excessive N and P fertility is already a concern in dairy pasture soils of the nearby Waikato region which has similar soils as the Bay of Plenty (Environment Waikato 2008).

Mean Olsen P value increased from 67 mg/kg in 1999 to 97 mg/kg in 2009. Although the increase was not statistically significant, this represents a 45% increase and reflects the continual application of phosphate fertilisers in these dairy farms. The mean Olsen P value for 2009 is near the upper limit of the 15-100 mg/kg provisional Olsen P target range. This buildup of P can become a concern in the near future if P-laden sediment is carried away by runoff and enters waterways.

There was little change in bulk density with time. All mean values lie within the provisional target range of 0.5 to 1.4 t/m^3. Macroporosity decreased slightly in 2004 but recovered to near starting values in 2009 which probably reflects the dynamic nature of this soil property in response grazing pressure. Mean values for each year lie within the provisional target range of 6 to 30%.

Temporal changes in topsoil qualities of maize sites

Table 2 shows the temporal changes of topsoil qualities of maize sites. There was little change in topsoil pH and the mean values lie within the provisional target range of 5 to 7.6 for cropping soils.

There was a decline in total C from 2000 to 2003 but stable afterwards. The cause of this decline is unclear. Nevertheless, all mean values were above the provisional target value of >2%. Similarly, there was a decline in total N from 2000 to 2003 but also stable afterwards. The cause of this decline is also unclear. Provisional target values are not established for cropping soils but low values are undesirable. Sparling and Rijkse (2003) noted that this apparent degree of change is far greater than can be accounted for by soil management over this short time period. They attempted to explain the declines in differences in sampling methods between these two years. In the initial sampling, the samples were collected in mid-season when the crop was still in place rather than the preferred method of sampling after harvest which was done in 2003. Even so, differences of the magnitudes observed was unusually large, and suggest substantial soil disturbance, with topsoil being mixed with subsoil giving a highly variable matrix. They indicated that the anomalous results can only be satisfactorily resolved through further sampling. Subsequent samplings from 2003 through to the present show that total C and total N values remain stable suggesting that the initial sampling results are most likely in error.

The C/N ratio appears to be a stable soil property. All mean values are within the 8 to 20 provisional target range for cropping soils.

Table 2. Temporal changes in topsoil qualities of maize sites with respect to pH, total C, total N, C/N ratio, anaerobically mineralisable N, Olsen P, bulk density, macroporosity, and aggregate stability (n=6 in 2000-2004; n=5 in 2006-2009)

Soil Quality	Year					
	2000	2003	2004	2006	2009	P value
pH	6.12	6.27	5.98	6.32	6.21	0.384
Total C (%)	5.47	3.18	2.99	3.17	3.24	0.027
Total N (%)	0.42	0.26	0.25	0.26	0.26	0.021
C/N ratio	13.04	12.28	11.76	12.33	12.43	0.590
Olsen P (mg/kg)	55	58	50	54	48	0.962
Anaerobically mineralisable N (mg/kg)	26	35	30	35	46	0.281
Bulk density (t/m^3)	0.98	no data	1.10	no data	0.98	0.234
Macroporosity (%)	17.9	no data	15.0	no data	20.5	0.430
Aggregate Stability (mm)	1.02	no data	0.95	no data	0.98	0.957

Except for a slight decrease in 2004, there is an increasing trend in anaerobically mineralisable N but the magnitude of increase is far less than the soils of the dairy sites. All mean values lie within the 20 to 200 mg/kg provisional target for cropping soils.

Olsen P values appear to be gradually decreasing implying that P applied as fertiliser is being taken up by the maize crop. However, all mean values are still within the 20 to 100 mg/kg target values for cropping soils.

Bulk density values appear stable. All mean values lie within the provisional target range of 0.5 to 1.4 t/m^3. Similar to the dairy sites, high macroporosity was maintained in maize sites. Mean values are within the 6-30% provisional target range. However, macroporosity values for maize sites are generally higher than the dairy sites which is probably caused by cattle grazing the latter sites.

Aggregate stability did not appear to decline over time. The aggregate stability values were close to 1 mm. These values, however, are less than the 1.5 mm desirable target set by Beare *et al*. (2009). The sandy nature of the maize soils of the Bay of Plenty and the effect of long-term cultivation could be contributory factors for not attaining the desired aggregate stability value.

Conclusion

For both dairy and maize sites, many of the topsoil quality parameters were being maintained and were within the provisional target values set by Landcare Research. However, the steady increase in the levels of anaerobically mineralisable N and Olsen P in dairy sites over a ten-year period is a concern. High values of anaerobically mineralisable N could lead to increased nitrate leaching while high values of Olsen P could lead to P-laden sediment polluting streams, rivers and lakes.

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The function of soil biodiversity as indicators of soil quality: Insights from the UK Defra SQID project

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Abstract
This presentation outlines the final results from the UK Defra SQID project which identified and piloted a suite of biological indicators of soil quality for deployment in national-scale soil monitoring programmes to meet a range of UK policy objectives. The indicators were selected to provide information on the soil biological processes which underpin soil function and therefore support ecosystem services. A semi-quantitative framework was used to systematically capture the wealth of information in the literature and from expert knowledge on potential indicators of soil quality with a total of 183 indicators assessed. Six soil biological methods have now been piloted in the UK using two complementary field approaches. These six reflect the genotypic, phenotypic and functional characteristics of soil biodiversity. Here we review the results from the two field approaches, the first to assess the temporal sensitivity of biological indicators to key environmental pressures across a 12 month window and the second to asses the ability of biological indicators to discriminate between different habitats. We discuss the relative performance of the indicators and how these were prioritised for national-scale soil monitoring and finally, how this process has revealed new insights into the distribution and characteristics of soil biological properties within UK soils and habitats.

Key Words
Soil quality, biological indicators, sensitivity, discrimination.

Introduction
Soil organisms are important to the maintenance of many ecosystem processes and properties which underpin vital soil functions (Bardgett *et al.* 2005) and it has been suggested that the diversity and behaviour of the soil biological community have great potential in detecting changes to soils brought about by various pressures such as climate change or pollution (Fließbach *et al.* 2007). Indeed a UK Royal Commission on Environmental Pollution concluded that biological indicators should be included in any future monitoring of soil quality (RCEP 1995). However, although many soil biological parameters are being proposed as indicators of soil quality, few have been tested for use in national monitoring programmes which operate over large spatial and temporal scales (Black *et al.* 2008). We have recently completed a project funded by the UK Department of Environment, Food and Rural Affairs to trial a limited range of soil biological parameters to assess their robustness for use in national scale soil monitoring programme. Here we present results from our two field comparisons of the following biological parameters; N, C, P & S enzyme responses using a multi-enzyme fluorometric assay; carbon substrate utilisation using a multiple substrate induced respiration assay (Microresp™); microbial community structure from phospholipids fatty acids and using MTRFLP on DNA extracts; and finally both microarthropod and nematode community structures from morphological identification. These indicators were prioritised through a robust assessment process and show high relevance and applicability to large-scale monitoring of soils (Ritz *et al.* 2009). The biological indicators under investigation also have specific relevance to the maintenance of soil health, via the delivery of ecological processes, and are highly relevant to the soil functions of food and fibre production, environmental interactions, and ecological habitats and biodiversity.

Methods
The project involved four aspects; logistical issues such as reproducibility of results from standard operating procedures; the sensitivity of the each biological indicator to three important environmental pressures (atmospheric pollution, land restoration and heavy metals); the ability of the indicators to differentiate between different habitats and, ultimately, the relative performance of the range of genotypic, phenotypic and functional biological indicators of soil quality. The primary purpose was to rigorously test these indicators under relevant field conditions and re-evaluate their suitability for national soil monitoring.
In the first trial, the sensitivity of each parameter was assessed against its temporal variability over 12 months within three field experiments, each representing a distinct pressure / driver (atmospheric N deposition,

applications of metals through sewage sludge and restoration of land). In the second trial, the parameters were tested to see if they could discriminate between habitats. The approaches to the field trials are outlined in the following.

A To test the biological indicators for their sensitivity to distinct environmental pressures
The aim of this objective was to evaluate the potential indicators in the specific context of distinct environment pressures, against the associated background of temporal and spatial heterogeneity. To this end, soils derived from well- established and replicated field experiments were utilised, where pressures have been defined and controlled, with appropriate replication. The three sites provide contrasting pressures that relate to the key soil functions (*viz*. food/fibre, environmental interactions, habitat/biodiversity). These are sewage sludge applications to agricultural land, simulated atmospheric nitrogen deposition on upland grassland habitats and restoration of open-cast mine sites. To assess sensitivity of the individual indicators against their temporal and spatial variability, soil samples were taken bi-monthly throughout a one-year period. Each member of the consortium was responsible for field sampling at their specified site and the distribution of soil samples to the relevant partners for laboratory analyses. Biological indictor responsiveness was assessed from all occasions and subsequent data analysis comprised a variety of statistical techniques, including multivariate analyses.

i) Site 1: Sewage sludge trial.
The re-cycling of wastewater sludge to land is a common practise on many grassland and arable soils and can result in considerable ecological and agricultural benefits. However, when sludge that is high in heavy metals is used the build up of potentially toxic elements can reduced the size and activity of the microbial biomass and reduce the numbers of effective N-fixing Rhizobium. It therefore makes a suitable test case to evaluate both the benefits and potential damage that such re-cycling practises might put on soil. The Hartwood field site (Scotland) was selected from UK Sewage Sludge Network. An advantage was the long term datasets for chemical and other microbiological data for comparison. The sampling design comprised four treatments by three field replicates on six occasions.

ii) Site 2: Atmospheric nitrogen deposition in an upland grassland trial.
There is now evidence of widespread changes in plant diversity in the UK and eutrophication is likely to be one of the main reasons for these changes, with atmospheric deposition of nitrogen playing a significant role. As well as affecting individual plant species, and potentially soil biodiversity, the deposition of nitrogen could pose a threat to conservation-status habitats and the already acidified freshwater ecosystems in upland areas of the UK. Critical load exceedance for nitrogen makes water quality at risk from increased nitrogen leaching from soils under many habitats. The ADAS Pwlllpeiran upland grassland site (Wales) was selected from several field experiments that address both the addition of nitrogen at varying concentrations are available via the Defra Terrestrial Umbrella network. This site had experience 7 years of long-term N additions. Again an advantage was the long term datasets for vegetation and soil chemical properties available. The sampling design comprised four treatments by three field replicates on six occasions.

iii) Site 3: Restoration gradient.
From available land restoration programmes, soils were taken from the Sutton Courtney mine reclamation site in S. England. This site has been subject to opencast coal mining operations and subsequently restoration over recent decades. There was a restoration gradient from undisturbed benchmark sites (e.g. woodland) and restoration counterparts at a variety of ages since re-instatement. The sampling design comprised twelve samples on a transect, aligned to the restoration gradient, taken on six occasions.

B To test the biological indicators for their ability to discriminate between a diverse range of habitats.
The aim was evaluate the discriminatory power of the potential indicators with respect to the typical range of habitats in the UK. The nature of the discrimination trial was not to specifically target extremes but rather to test the robustness of these indicators under a wide range of conditions likely to be encountered in a large scale monitoring exercise. Nine habitats were selected to include soils at the extremes of certain key properties e.g. acid to calcareous soils; moorland to arable soils (Figure 1).
Soil samples (0 – 15 cm depth; 10 samples per habitat) were obtained from the field survey of Countryside Survey 2007. The value of linking to CS2007 was the opportunity to access the wide-range of associated environmental data, much of it obtained from co-located sampling plots. These data include; plant species richness, broad habitat type, Ellenberg plant community scores (indicative of environmental gradients such as fertility, shade, etc), soil type/pH/carbon content etc, geological parent material, slope, altitude, Land Cover

(from LCM2000). Data were analysed using a variety of statistical techniques contingent on the type of data. This will include mixed-model ANOVA, principal component analysis, canonical and other correspondence analyses.

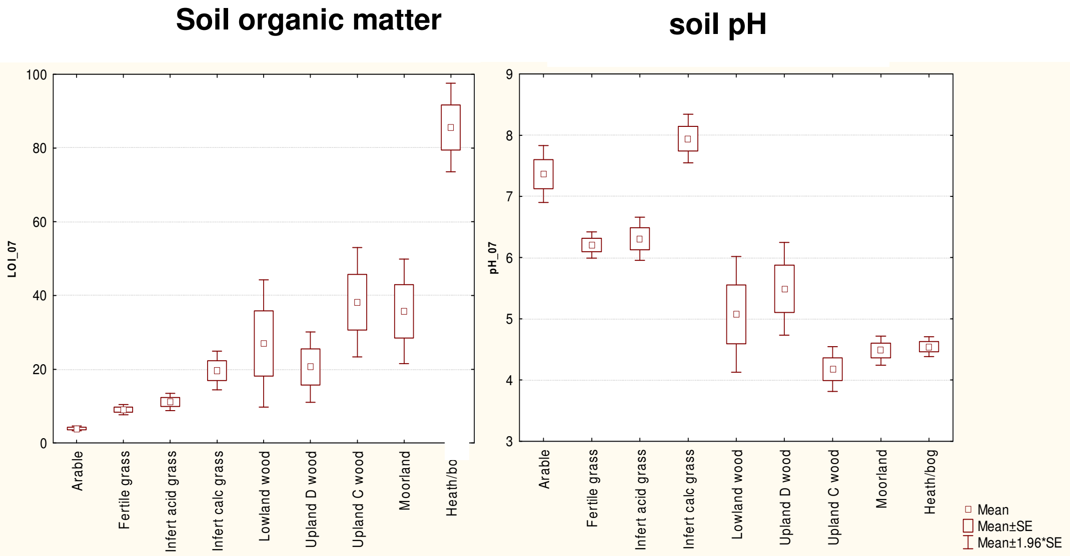

Figure 1. Soil organic matter and soil pH characteristics of the nine habitats sampled during Countryside Survey 2007 for the Defra SQID project (n = 100).

Conclusion

The results from the sensitivity and discrimination trial produced a wealth of statistically significant responses from direct, derived and multivariate measures of soil biological properties to the three environmental pressures and nine habitat types and clearly demonstrate the potential of different methods for application in monitoring. Using consistent approaches to the statistical analyses of all the biological indicators, and their associated measures, we have been able to directly compare the relative performance of genotypic, phenotypic and functional biological indicators and prioritise indicators for national-scale soil monitoring. The prioritisation also considers logistical issues such as reproducibility of results using standard operating procedures. In parallel, we have devised a novel approach that integrates the wealth of information that can be derived from biological indicators of soil quality using genotypic, phenotypic and functional characteristics. This trait-based approach is providing new insights into the distinct characteristics of soil biodiversity under different habitats and the potential consequences of environmental change on soil biodiversity.

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Systems integration: enhancing soil information delivery to Victoria

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Abstract
Data, process management and information sharing are among the biggest challenges facing the scientific community today. The integration of heterogeneous primary and derived data sets arising from soil and land survey activities is imperative to provide easily accessible information required for informed decision making regarding future farming systems and their impact on the environment. Currently the Department of Primary Industries, Victoria (DPI) utilises a suite of software applications in tandem to maintain and manage its digital and hardcopy land resource data sets. An overall soil survey information strategy is evolving to enable the multiple systems to work together as an integrated system. The integration of these systems will provide a sound foundation for building a comprehensive knowledge management strategy based on both current and historical data sets. This will result in improved access, increased usability, efficient information exchange and improved standardisation and consistency of data and its derivatives. Rapid access to samples and data is being used for calibration and prediction of soil chemical and physical properties using MIR. Full integration of these systems will provide an efficient foundation for building a comprehensive knowledge management strategy. This paper presents the concept for the integration of the diverse data sets that comprise DPI's soil site data collection.

Key Words
Data and information management, legacy data, soil archive.

Introduction
DPI and its predecessors have collected soil site data over many years and these sites remain relevant today but not readily accessible for widespread utilisation. DPI's collection comprises physical samples, as well as hardcopy and digital records stored in a number of current and defunct systems. A multidisciplinary team, involving soil scientists, soil surveyors, spatial information scientists, quality systems specialists and laboratory information management systems (LIMS) specialists is developing a systematic framework to enable the integration of soil site data. The Victorian soil site archive comprises almost 30,000 physical soil samples (derived from soil survey and research studies carried out over the past 80 years). Associated with these samples are records and analytical data sets held either as hardcopy or in digital form in a variety of systems. DPI is creating a durable soil sample archive and associated digital inventory to optimise access and utilisation of current and historical samples into the future. A systematic process for the digital capture and storage of ancillary data (consisting of many hundreds of air photos (annotated with soil mapping boundaries and sites identified), field books, working maps and analytical data) is being developed. Preserving this material through digital capture will enable future entry of data associated with many thousands of soil sites across Victoria into a soil information system (SIS) while preserving part of DPI soil and land survey history.

Methods
Victorian Soil Archive
The soil sample archive comprises almost 30,000 soil samples that are currently stored within more than 50 wooden crates (1.2 m x 1.2 m x 0.6 m) containing cardboard boxes and containers of soil labelled only with sample number. A variety of plastic and cardboard containers, which were never intended for long term storage, have been used over the years and many of these are deteriorating with age. At present there is only a manual list of the sample numbers stored in each crate. The samples are being progressively transferred to archive quality containers (high-density polyethylene white jars with white polypropylene screw caps). An inventory is being developed and a process of linking the sample number to the ancillary survey record is being undertaken

to enable georeferencing and linkage to the Victorian Soils Information System (VSIS). Once linkages to the original survey and geographic references have been established the samples will be labelled with a barcode and key sample information as per protocols established for the Australian Soil Information System (ASRIS) using an archive quality label.

Historical records

The data collected from previous soil/landform and agricultural and landscape research sampling programmes consists of landform, morphological, physical, chemical and biological descriptor records stored as hardcopy reports as well as maps, aerial photos, field books, site and locality summary sheets, and associated data tables (examples are provided in Figure 1). Although summaries of this information have been collated into hardcopy reports or linked to landscape classification systems such as Land Systems of Victoria Zones, much of the raw data needed for ongoing research, monitoring and systems modelling remains in poorly maintained paper-based archives. Records from 1930 to the present day are currently stored in a range of boxes and filing cabinets. The survey methods have been evolving over time and as such earlier survey data may be incompatible or require interpretation and translation to align to current soil survey data standards. Consequently not all historical survey data is of equal utility or value.

Figure 1. Example of page details in hardcopy field books; Aerial photo with site details and locations identified; Historic soil survey maps; Agriculture lab submission sheets. These records provide opportunities to track sample numbers associated with soil survey sites and link them to their laboratory data and geographic location.

Victorian Resources Online (VRO)

DPI's Victorian Resources Online (VRO) website is the key portal for accessing information about Victoria's natural resources (e.g. soil, landform, water, biodiversity) and their management. It contains maps and associated information, downloadable reports as well as many related links to other information sources. A wide range of information products generated from Victorian soil and land survey work over the past 80 years is made available online. The website is a repository for many contemporary and historical reports. Many of these have been digitally captured (scanned and/or re-typed) and made available as downloadable pdf format reports that have been customised to optimise loading and readability. A 'Soil and Land Survey Directory' facilitates searching of over 100 soil and land survey reports via key words or by a specific Local Government Area or Catchment Management region (refer to Figure 2).

Figure 2. Victorian Catchment Management Authority regions.

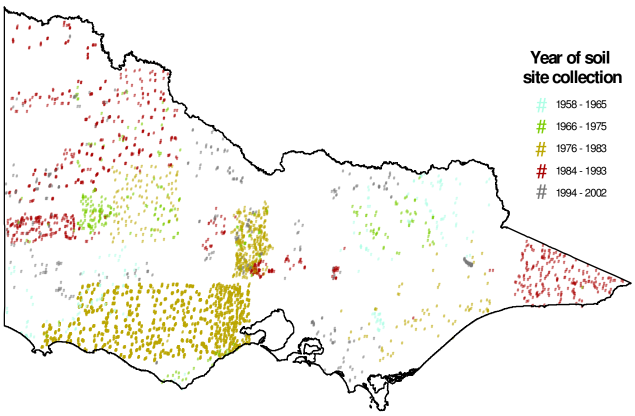

Figure 3. Soil site locations and periods (years of collection, for site data currently in VSIS).

Victorian Soil Information System (VSIS)
The VSIS is being used to capture soil site data (both field and laboratory generated) in a systematic way that will allow the data to be stored, utilised and mapped by a diverse range of users at a local, regional, state and national scales. It is a primary centralised database that provides access to soil site data that adheres to national soil and land survey standards. VSIS consists of an entry module (Victorian Soil Entry System - VSES) used to ensure data adheres to current national standards for soil and land survey guidelines (McKenzie *et al*. 2008) and for field survey (National Committee on Soil and Terrain 2009), and an output (query) module to interrogate the data. Key historical soil sites are progressively georeferenced and then manually entered into the Victorian Soil Entry System (VSES). VSIS contains "high quality" data for 2,800 sites (refer to Figure 3). A further 1000 soil sites have been entered in the VSES and are under going quality checking protocols prior to upload into VSIS.

Laboratory Information System (LIMS)
The DPI-Werribee laboratory utilises a LIMS (relational database hosted on a SQL server) to manage all soil samples that have been submitted for analytical testing. This system is used to register, label, track, report and invoice all samples submitted for testing. Data is stored and tracked in the system using the sample number as the key identifier. Although the system can store geographic reference information it was seldom provided by those submitting samples for analysis. Additional information provided by the submitter for quality checking purposes, including references to the original soil site location and observations was often stored in a free form text field. Unfortunately, due to the myriad of information and formats used in the free text field it is of little use for linking data to the soil sites. Mostly it provides additional information for checking the links established by other mechanisms (e.g. field books, original soil observation and description cards, or survey indexes). Analytical extraction procedures and methods have evolved with advances in technology. During data extraction analyst knowledge and method references are utilised to describe the analytical process in terms of established National Standards (Rayment and Higginson 1992) for soil testing. The LIMS has operated through implementation of two software platforms including LabWare (current system containing records for soil testing from 2006 to the present day) and SMS (defunct system containing records for soil testing from 1988 to 2006).

Data Track
Data Track is a DPI business initiative to further develop policies, procedures and protocols for data storage and management to improve the security and custodianship of existing soil data assets and improve their accessibility, utility and quality. An important part of this process is the development business capability, capacity and intelligence to ensure best value is derived from the DPI soil data assets. Data standards and management protocols are being used to improve the soil information data quality, consistency and availability and thereby preserve them for future users. The initial design and development of the VSIS was fostered within this initiative to provide a flagship model for management of a major data area for the DPI Future Farming Systems Research (FFSR).

Results
The Soil and Land Directory will be a basis for inventory and categorisation of the hardcopy records not yet digitised. The information is progressively inventoried and categorised with respect to Victoria's Catchment Management Regions and then scanned and stored. Three key surveys from the Corangamite, Mallee and East Gippsland regions have been used to refine and develop the system for integrating this information. Samples and records are linked to the original records/maps and then georeferenced to facilitate entry into VSIS. Electronic links are being established between VSIS, VRO and LIMS-SMS / LIMS-LabWare outlined in Figure 4. These links will provide considerable gains in efficiency and accuracy for the further inclusion of historical information from 1988 in VSIS by significantly reducing manual data entry. A mechanism for the provision of a VSIS link during sample registration for all future soil site samples will ensure efficient information exchange between the systems.

Mid infrared (MIR) spectroscopy is being used by DPI as a powerful tool for rapid multi-parameter screening of soil properties. Soil chemical and physical analyses are time consuming and becoming increasingly expensive. Recent research has demonstrated the utility of using MIR spectroscopy to complement and perhaps one day replace aspects of traditional soil analysis. MIR calibrations are being developed from the chemical and physical data that has already been collected for the samples in the DPI Victorian Soil Archive for the major Victorian agricultural soils. In conjunction with this, MIR will be used to generate results for other analytes that were not part of the original analytical data set. The georeferencing of these samples will enable mapping of soil properties such as pH, EC, exchangeable cations, organic carbon, total nitrogen, and physical properties such as plant available water capacity, particle size analysis and even clay mineralogy once satisfactory calibrations have been established.

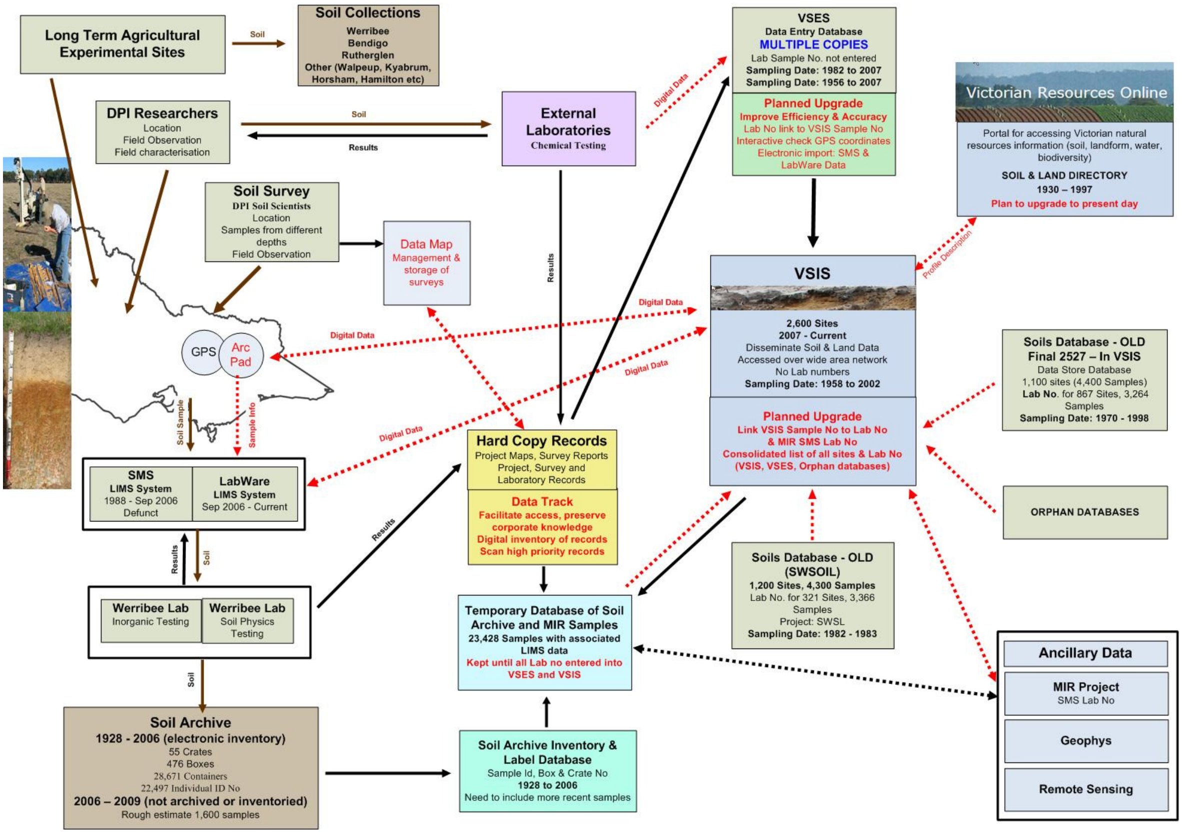

Figure 4. Relationships between soil survey data and archive management systems for Department of Primary Industries, Victoria.

Conclusion

Consideration of future farming systems and the impact on the environment is increasingly using modelling, underpinned by data to understand land use impacts and increase productivity in response to environmental factors including climate change. The cost of developing new data sets has demanded re-analysis and comparative analysis of existing soil samples and information. Currently there exists only a manual record of these samples. The biggest challenge to systems integration currently is the volume of historical samples to process and archive and the diversity of record keeping and data management systems that were used. Linking soil samples to original survey and sample number, and then establishing geographic references is proving most challenging. Most survey reports did not include the sample number in the final tables and so ancillary records need to be traced. An electronic inventory will greatly facilitate this process. The integration of historical and laboratory systems (LIMS) with VSIS and VRO will support the growing need for raw data to support research, monitoring, modelling and soil health initiatives.

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Time to establish a ^{137}Cs-derived net soil redistribution baseline for Australia?

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Abstract
Reliable quantitative data on the extent and rates of soil erosion are needed to underpin the selection of effective soil conservation measures, to inform carbon balances for greenhouse gas abatement and carbon storage and in understanding soil function across landscapes for sustainable agricultural systems. The caesium-137 (^{137}Cs) technique has been used successfully in many parts of the world to estimate net (ca. 30-50 years) soil redistribution by wind and water erosion and tillage activities. It is a point-based technique that lends itself to mapping over large areas but which has hitherto been confined largely to individual fields and hillslopes. The application of the ^{137}Cs technique to map soil redistribution every ≈5 km across Australia is achieved using geostatistics and nationally coordinated measurements from ≈200 locations. Between the mid-1950s to early 1990s the median net soil redistribution for the Australian continent was -0.19 t ha^{-1} yr^{-1}. Soil erosion exceeding 0.5 t ha^{-1} yr^{-1} was estimated to occur over 16% of Australia, mainly in the cultivated regions where the median net soil redistribution was -1.26 t ha^{-1} yr^{-1}, more than eight times larger than the rate on uncultivated land (-0.16 t ha^{-1} yr^{-1}). The approach demonstrates a viable methodology for establishing a national baseline which could be readily applied in other countries. The baseline map of net soil redistribution and its uncertainty provide the opportunity to optimise a future ^{137}Cs survey before ^{137}Cs detection is not viable. We propose that this baseline map of net soil redistribution is updated using measurements from samples collected as part of other national campaigns.

Key Words
Caesium-137 (^{137}Cs), soil erosion, geostatistics, sequential indicator co-simulation, rainfall, land-use

Introduction
Soil erosion may be a highly selective process causing fine, nutrient-rich material to be removed, progressively coarsening the soil and reducing the moisture capacity. Accumulation of dust, however, can significantly improve soil fertility. Degradation of soil particularly by erosion decreases agricultural productivity and has considerable on-site and off-site impacts and costs, evident in the September, 2009 Australian dust storms. Changes in land use are widely recognised as capable of greatly accelerating soil erosion and erosion in excess of soil production may result in decreased agricultural potential. The loss of vegetation cover increases the susceptibility of the soil to erosion (erodibility) by wind, water and tillage. Whilst erodibility maps raise the awareness of the risk of soil erosion they may be misleading as management tools because soil redistribution processes may not be adequately taken into account.

Most soil erosion measurement and monitoring approaches have limited representativeness and insufficient duration, to provide reliable estimates of soil erosion. It is particularly problematic in semi-arid environments where soil erosion is highly variable in space and time and often insidious requiring the removal of considerable quantities before loss is noticeable. Extrapolation of results from small experimental plots across large areas is notoriously unreliable. These difficulties in measuring and monitoring soil erosion in space and time are likely responsible for its common neglect in carbon balances for greenhouse gas abatement and carbon storage and in understanding soil function across landscapes and agricultural systems. The aim here is to provide the first map of ^{137}Cs-derived net (mid 1950s to early 1990s) soil redistribution across Australia. The intention is for this map to serve as a baseline against which subsequent national assessments may be made and to raise awareness of the methodology for sampling and mapping over the continent using few samples.

The artificial radioactive tracer caesium-137 (^{137}Cs) has been used successfully to measure net (ca. 40 year) soil redistribution rates in many environments including Australia (Ritchie and McHenry, 1990). Its widespread use is probably because the ^{137}Cs technique overcomes many of the problems of monitoring soil erosion and deposition over the medium-term (5 to 50 years) and at the hillslope scale (Walling and Quine, 1991). In this respect, the ^{137}Cs technique offers the greatest potential for measuring net soil flux in semi-arid environments where soil flux monitoring difficulties are compounded by considerable spatial and temporal variability of the controlling factors. The ^{137}Cs technique has commonly been applied to small areas (fields or hillslopes) often because the use of traditional approaches to sampling using regular grids and large area mapping is prohibitively expensive and therefore representative baselines using the ^{137}Cs technique over very large areas have not previously been attempted. However, the combination of nested sampling and geostatistics has considerably extended the size of the area that may be investigated and mapped (de Roo, 1991; Chappell, 1998; Chappell and Warren, 2003). A national reconnaissance survey of ^{137}Cs-derived soil erosion in Australia was instigated during the early 1990s for selected agro-economic sites (206 sites) and the national perspective was presented by Loughran *et al*. (2004). These samples are used here with geostatistics to produce a preliminary map of net soil redistribution across the continent. A related map of the spatial uncertainty is used to discuss improvements in the maps and for establishing a baseline before the ^{137}Cs technique becomes no longer viable.

Methods

Field and laboratory methods

This study used the measurements from the national reconnaissance survey. The survey collected soil along single transects, down complete slopes, at paired sites in the same locality as typical land management practices within selected agricultural-economic regions. Field sampling commenced in 1991 and over 5000 samples were collected and analysed for 206 sites throughout Australia (Figure 1a). The way in which the samples were obtained from the soil, their preparation for, and the method of, gamma-ray spectrometry to measure ^{137}Cs activity is described in Loughran *et al*. (2004).

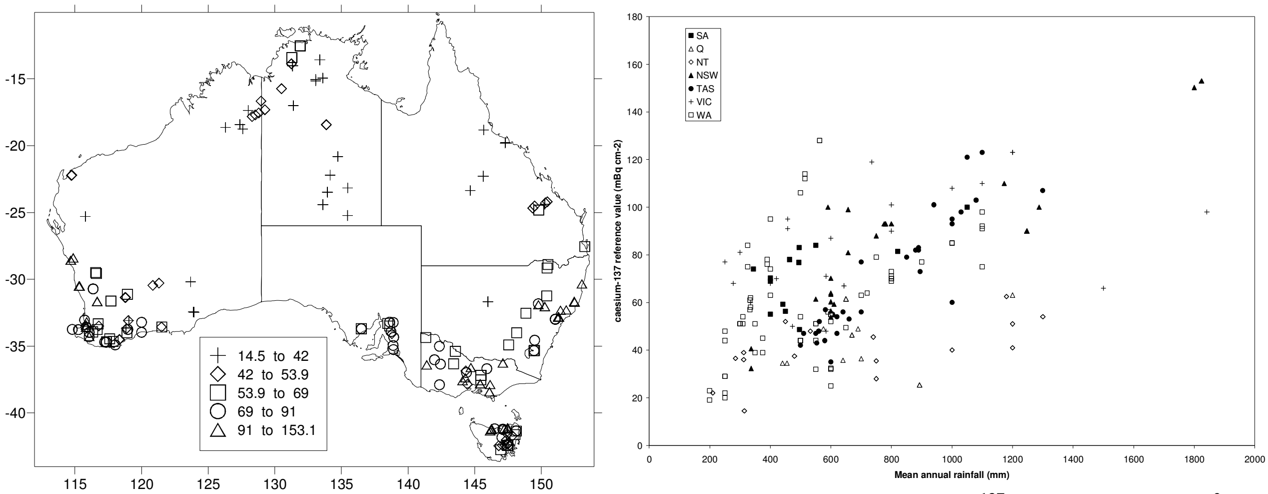

Figure 1. Locations of sites in Australia where soil was sampled for use as a reference ^{137}Cs value (mBq cm^2) (a) and the relationship between long-term (1954-1990) mean annual rainfall and ^{137}Cs reference values separated for each state and territory (b).

^{137}Cs technique for estimating net soil redistribution

Thermonuclear weapons tests performed above ground in the 1950s released the artificial radionuclide caesium-137 (^{137}Cs) into the atmosphere where it circulated within each of the Earth's hemispheres. The ^{137}Cs arrived at the Earth's surface with rainfall and is strongly associated with its spatial distribution (Figure 1b). It is assumed that once the ^{137}Cs reached the soil surface it was fixed rapidly and firmly to soils and becomes an effective tracer of soil redistribution (Ritchie and McHenry, 1990). At locations in the landscape where there is little or no soil erosion or deposition (undisturbed) the amount of ^{137}Cs in the soil is reduced by radioactive decay (^{137}Cs half-life 30.2 years). At other locations the amount of soil ^{137}Cs is a function of the erosion and / or deposition intensity and duration. A calibration relationship is required to convert the percentage of ^{137}Cs lost or gained (X) relative to the ^{137}Cs inventory at the undisturbed reference location. This study follows that of Loughran *et al.* (2004) and uses two models; one for calculating net soil loss (Y; kg ha^{-1} yr^{-1}) for sites which had never been cultivated (N=31; Equation 1) and the other for sites which had been used for cultivation (N=61; Equation 2):

$$Y=17.49\ (1.0821)^X \quad (1)$$

$$Y=296.1\ (1.0539)^X \quad (2)$$

Net soil accumulation was calculated using these equations in reverse mode for sites which had gained ^{137}Cs. In this case, Y is net soil gain and X is the percentage ^{137}Cs gain relative to the reference value. Both relationships were derived from long-term soil-loss measurements using runoff-erosion plots, or similar experiments in New South Wales, Queensland and Western Australia (Loughran and Elliott, 1996).

Mapping net soil redistribution

The few studies that have accurately mapped net soil flux over large areas have used innovative sampling designs and/or geostatistical procedures. However, the map of local estimates from (co-)kriging often smoothes out local details. Such conditional bias is a serious shortcoming when trying to detect patterns of extremes such as zones of large and small ^{137}Cs and soil erosion. Sequential indicator co-simulation for uncertainty modelling is used here to generate an ensemble of equally probable realisations of the property spatial distribution and enable differences amongst the realisations to be used as a measure of uncertainty. A map of reference ^{137}Cs and its uncertainty is produced using sequential indicator co-simulation with long-term (1954-1990) mean annual rainfall data for Australia (Jeffrey *et al*., 2001). A map of ^{137}Cs activity and its uncertainty is produced using the same technique but combined with the Australian Soil Classification. The percentage difference between these maps relative to the reference value is used to calculate net soil redistribution using the equations (1 and 2) above. Land use data (1992/93) mapped at the national scale provided by the Bureau of Rural Sciences determined whether a point was ever cultivated or not and therefore which equation to use in the estimate.

Results

The median indicator variograms of ^{137}Cs reference inventory and ^{137}Cs inventory (not shown) were fitted best with linear-with-sill models and identified a range of spatial dependence of around 2150 km (21°). In both cases a large proportion (>50%) of the total variance was nugget. A considerable amount of variation in these properties had not been captured by the median values of the reconnaissance survey samples. The values of the optimised model parameters were used in the median approximation sequential indicator (co-) simulations. The linear-with-sill model could not be used in this process and was replaced by the spherical model as an adequate approximation of the former model.

The map of per-point median ^{137}Cs reference inventory is shown in Figure 2a. There is good correspondence between the measured reference values (Figure 1a) and the map. Notably, the ^{137}Cs reference pattern resembles that of rainfall but does not over-estimate the reference values in northern Australia where rainfall is large. This pattern is consistent with the expected fallout pattern. The per-point interquartile range is small considering the relatively few samples across the continent because of the relationship with the rainfall data. Where the variance is large there are very few, if any, samples and it is in these areas that future samples should be obtained to improve the map and establish the reference inventory.

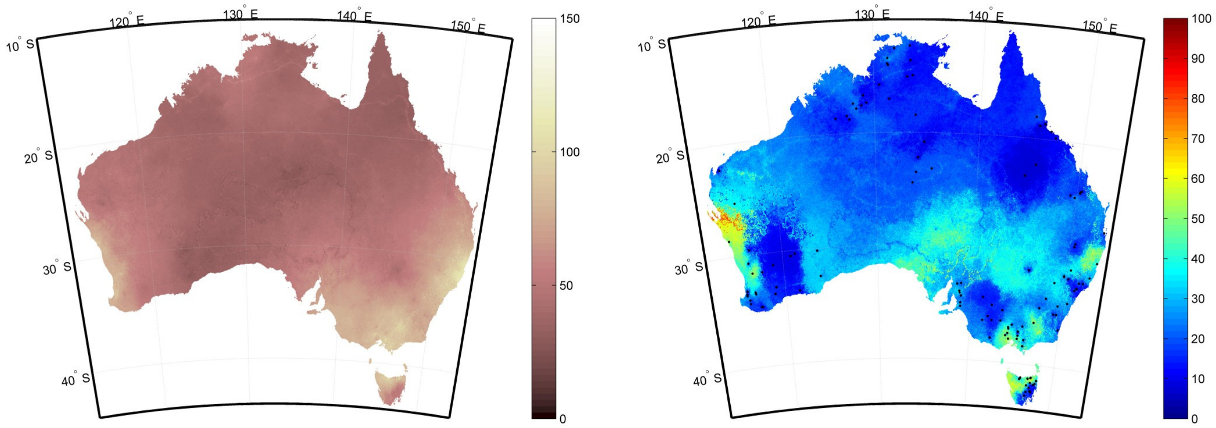

Figure 2. The per-point median ^{137}Cs reference inventory (mBq cm^{-2}) (a) and interquartile range (mBq cm^{-2}) from the median approximation of sequential indicator co-simulation with long-term (1954-1990) mean annual rainfall.

The map of per-point median ^{137}Cs activity shows that across Australia there is less ^{137}Cs activity than the reference values (not shown). The percentage difference between the ^{137}Cs reference map and the activity map shows this pattern and also identifies those locations where there is net ^{137}Cs gain (not shown). The land-use classification was used to determine which of the empirical models to use in the conversion of the percentage ^{137}Cs difference to net soil redistribution. The per-point realisations median net soil redistribution shows areas that are stable, depositional or have very small erosion rates across the majority of Australia. Areas that have large net erosion rates include the main cultivated areas along the coastal regions of Western Australia, South Australia, Victoria, New South Wales and Queensland. The most eroded area is in the western most region of Western Australia (Pilbara region). Assuming a symmetrical distribution in the realisations this region has an erosion rate with uncertainty of >6±3 t ha^{-1} yr^{-1}. The areas of greatest uncertainty include mid South Australia, mid New South Wales and south-west Queensland. The areas of greatest uncertainty are those associated with net soil gain. The main cultivation regions of Australia are similarly uncertain (±2 t ha^{-1} yr^{-1}) and there is more than a 60% probability that these regions are eroding.

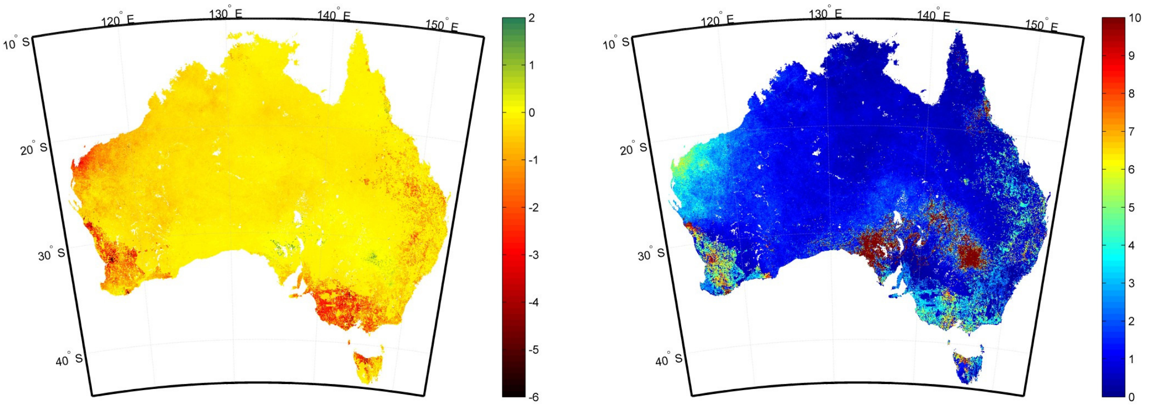

Figure 3. The per-point median ^{137}Cs-derived net (1950s-1990) soil redistribution rate (t ha^{-1} yr^{-1}) of Australia (a) and its interquartile range (t ha^{-1} yr^{-1}) (b).

Conclusion

We presented a workflow for the estimation of a baseline ^{137}Cs-derived net soil redistribution across Australia which represents its status in the early 1990s. The workflow is based on the soil ^{137}Cs samples collected at that time and the sequential indicator simulations with ancillary (rainfall and soil type) data. The results demonstrated that the geostatistical techniques applied here provide a powerful method to estimate ^{137}Cs and net soil redistribution across Australia using few samples. Many soil profiles have been collected across Australia and stored in the national soil archive and others are being collected as part of new national programmes. These samples may be suitable for ^{137}Cs measurement to improve the current map and to produce a new map which enables consideration of previous management practices and soil conservation policies. Any future national ^{137}Cs samples for comparison with the baseline should be undertaken soon before detection is not viable.

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Some statistical aspects of monitoring of soil change in Slovakia

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Abstract
Stratified soil monitoring network in Slovakia has been constructed on ecological principles where all soils, substrates, climatic regions, polluted and non-polluted areas are included. There are 318 monitoring sites on agricultural soils in Slovakia. Soil parameters are monitored and evaluated in harmony with EC for soil monitoring according to threats to soil (soil contamination, salinisation and sodification, decline in soil organic matter, compaction and erosion). The obtained results are evaluated separately according to soil type, subtype, geology, land use with regard to their area. If small spatial variability of measured parameters is significantly decreases. Basic statistical procedures are used with regard to some statistical aspects (to evaluate comparable couple files, frequency of measured data; the object of statistical evaluation must be correctly defined – area of site, soil type, respective subtype, evaluated parameter, depth of soil profile, etc.). Finally, so-called environmental statistical evaluation could be one of important statistical aspects of monitoring soil change.

Key Words
Soil monitoring, statistical aspects, environmental statistical evaluation, soil variability

Introduction
Soil monitoring in Slovakia is a vital component, alongside soil database and maps. Its importance consists of providing information about changing with space and time and answering questions about whether the quality of a soil is improving, deteriorating or staying about the same under a particular use and management practice.

Behaviour of soils and their variability

Individual soil units are open dynamic systems which are the results of the mostly long to very long sometimes medium to relatively short evolution-genesis of soil. Old soils have achieved the state of dynamic equilibrum with the other component of environment (climate, vegetation, groundwater, etc.). These soils are often characteristic with narrow range of parameters in space opposite the young soils, where the values of soil properties are rather dynamic and heterogenous. Variability of soils is often changed and decreased in the following sequence: initial soils – forming soils – strongly weathered soils. In the soil monitoring network of Slovakia are included all mentioned soils and therefore the area of monitoring site must accept this reality.

Design

The monitoring site is of circular shape with a radius of 10 m and an area of 314 m^2 where the variability of soil properties in space is low. The soil monitoring network was constructed on the basis of ecological principles. It means all soil types and subtypes, geology, various climatic and emission regions, lowland and highland are included. The result of this principle is 318 monitoring site in stratified network on agricultural land in Slovakia. Every monitoring site is located by GPS in WGS 84 coordinates system.

Basic principles of evaluation of monitored data

Basic soil parameters are monitored and evaluated in harmony with the European Commission for soil monitoring according to threats to soil (soil contamination, salinisation and sodification, decline in soil organic matter, soil compaction and erosion). During the evaluation process some statistical aspects are accepted, as follows (Kobza, 2008):

- basic characteristics of variability (range of variability, mean and standard deviation, dispersion, coefficient of variability)
- testing of statistical hypothesis (comparison of two comparable files, where parametric and non-parametric tests could be used)
- evaluation of dependence between parameters (e.g. regression)
- reciprocal correlation of several parameters using correlation matrix with values of correlated coefficients
- factor analysis
- prediction (estimation of development of observed parameters in time).

During the evaluation process it is necessary to take into account not only method of statistical evaluation, but also the concrete area of evaluated object (in this case-soil) which consists of units (e.g. Chernozems, Cambisols, etc.) with other additional specifications (e.g. Cambisols under grassland, on arable land, cultivated, non-cultivated, protected, non-protected, etc.). In addition, it may be said that, so-called **environmental statistical evaluation** (accepted previous principles) could be one of important statistical aspects of national-scale soil monitoring in effort to increase the objectivity of obtained results. In the following table 1 the basic statistical data are given on the example of three various soil bodies (all soils included in soil monitoring network, one soil unit-Stagnosol and finally the basic monitoring site with area 314 m^2).

Table 1. Basic statistical characteristics of three various soil levels.

Parameters	**Basic monitoring network in Slovakia - all soil types**							
	Arable soil (n = 223)				Grassland (n = 95)			
	Xmin	Xmax	X	V	Xmin	Xmax	X	V
Cox (%)	0.45	4.21	1.29	0.27	0.75	14.48	2.73	6.12
pH/KCl	3.89	7.92	6.46	0.75	3.56	7.16	5.24	1.29
Parameters	**Basic monitoring network in Slovakia- Stagnosols**							
	Arable soils (n = 37)				Grassland (n = 12)			
	Xmin	Xmax	X	V	Xmin	Xmax	X	V
Cox (%)	0.50	2.34	1.03	0.10	0.85	3.66	1.94	0.75
pH/KCl	4.47	7.42	5.96	0.62	4.59	7.04	5.83	0.68
Parameters	**Basic monitoring network in Slovakia – one monitoring site**							
	Arable soils (n = 5)				Grassland (n = 5)			
	Xmin.	Xmax.	X	V	Xmin.	Xmax.	X	V
Cox (%)	1.48	2.36	1.94	0.12	1.91	2.18	2.04	0.04
pH/KCl	4.92	5.13	5.05	0.02	5.42	5.68	5.46	0.02

V – coefficient of variability, x – arithmetic mean

In Slovakia, we have completely realised three monitoring cycles for the time being. In the following Figure 1 is presented the histogram of available potassium development on Cambisols which are the most extended soils in Slovakia.

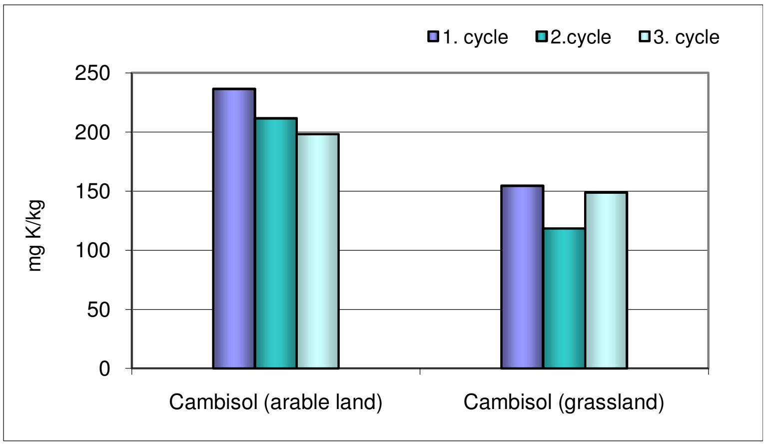

Figure 1. Development of available potassium in the arable layer of agricultural soils in Slovakia.

There are differences between arable soils and soils covered by grass. The higher content of potassium on arable soils is caused by fertilisation level, which is slightly decreased in time. Change in time is not statistically significant and it is very similar to variability in space of monitoring site area, which is a very important factor of evaluation (under conditions of Slovakia was max. 300-350 m^2 determined in effort not to change variability in space for variability in time). The statistical evaluation is often used for **prognosis of monitored parameters for the future**. This evaluation is provided so that outside conditions affecting the development of measured parameters in time are not changed (so-called "ceteris paribus" principle). Minimum evaluated data set of 8 in this case is recommended. An example on the path of fluorine in the air and soil in the surroundings of an aluminium factory in the Žiarska kotlina (depression) in Central part of Slovakia is given in Figure 2.

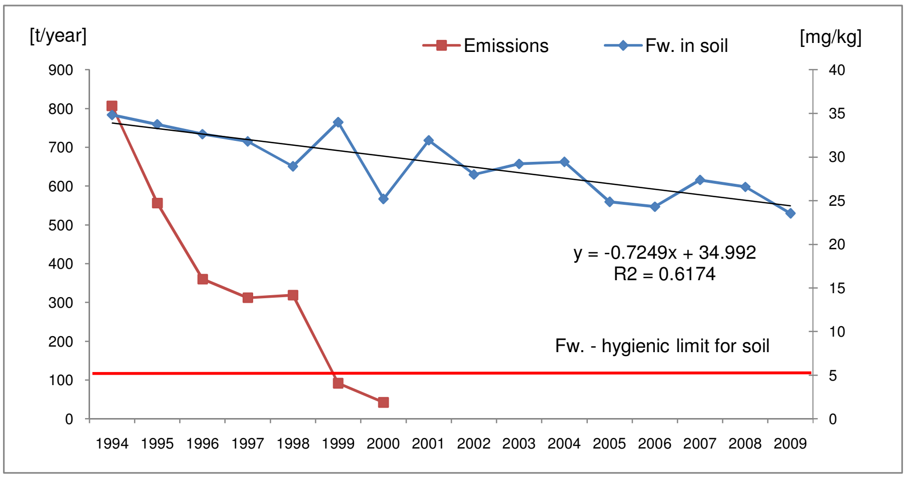

Figure 2. Amounts of fluorine in air and soil in the surroundings of an aluminium factory.

This figure refers to very important role of soil in environment where despite a strong decrease of fluorine in air, the content of this element in soil is still high (5-times the valid hygienic limit for Slovakia).

In addition, also some **basic interpolation methods using GIS** are included in national-scale soil monitoring in Slovakia, as follows (Blišťan, 2005):

- triangle methods (linear interpolation)
- inverse distance square (IDS)
- kriging

In the following Figure 3 is shown the kriging semivariogram of pH/KCl values as a function of their distance in the Medzibodrožie region, which has been studied as pilot area of an ENVASSO project (Environmental Assessment of Soil for Monitoring) in the framework of 6. FP (Kobza and Dobos, 2008).

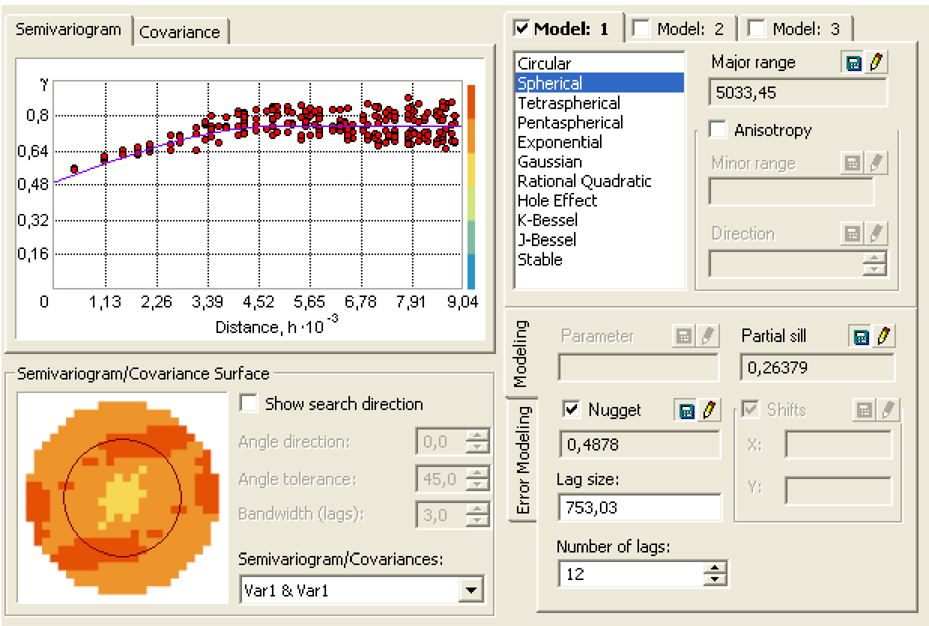

Figure 3. Semivariogram of pH/KCl in Medzibodrožie region.

It may be said that traditionally used technologies of GIS are mostly directed to management and reporting of 2D data for the time being. In such systems (Arc Info, Arc View) it is possible to create a wide range of maps which are used especially for evaluation of sensitive regions of Slovakia.

Conclusions

On the basis of obtained results it was determined that spatial variability decreases as the statistically evaluated object is getting smaller (e.g. in direction: national soil area – district – cadaster – field – soil type – soil subtype, etc.). Spatial variability is increasing for soil parameters which are more influenced by human activity (e.g. pH, soil organic carbon, available nutrients – P and K). Finally, during the evaluation process it is necessary to take into account not only statistical methods, but also area of the evaluated object (in this case – soil) which consists of units (e.g. Cambisols, Stagnosols, etc.) with other additional specifications (arable land, grassland, cultivated, non-cultivated, protected, non-protected, etc.).

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