7 Evaluating Land Quality for Carbon Storage, Greenhouse Gas Emissions and Nutrient Leaching

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7.1 Introduction

Recently the importance of good soil structure in mitigating climate change and environmental contamination has been recognized because soil structure influences the storage of carbon (C) sources and sinks of greenhouse gases (GHGs) and cycling of nutrients, which are key soil system processes. This is because the maintenance of soil structure by aggregation, particle transport and formation of soil habitats operates across many spatial scales to regulate water drainage, water retention, air transfer to roots for favourable gas exchange and mineralization of nutrients for release to crop roots (Kibblewhite *et al.*, 2008; Ball *et al.*, 2013a). For the functions being considered, the most important aspect of soil structure is the soil pore network, which determines the movement of gases, liquids and associated solutes, as well as particulates and organisms, through the soil matrix (Haygarth and Ritz, 2009; Sakrabani *et al.*, 2012). Good soil structure also sustains a favourable rooting medium for plants; if roots can't get to the nutrients the plants can't use them.

Overall, the maintenance of good soil structure is linked to favourable soil quality for agricultural production. This chapter will first discuss how soil properties influence soil C storage, GHG emissions and nutrient leaching and how these subsequently influence land quality. Next, visual methods for evaluating soils relative to their potential for C storage, GHG emissions and nutrient leaching will be described using both measured and modelled data. Finally, future directions for research are summarized.

7.2 Soil Properties Regulating Carbon Storage, Greenhouse Gas Emissions and Nutrient Leaching and Their Relationship with Soil Structure

Key soil properties for the three functions of regulating C storage, the exchange of GHGs and nutrient leaching are summarized in Table 7.1. These properties can be categorized as dynamic, unstable and subject to change or static, stable and essentially unchanged over time. Dynamic properties related to water and air flow in the soil are more suitable for assessing the functionality of soil under different management practices (Cavalieri *et al.*, 2009). Some of these soil properties can be directly and semi-quantitatively estimated using visual evaluation (see Section 7.3). Soil properties can also exert a direct or indirect

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Table 7.1. Summary of soil properties which influence the three soil functions: carbon (C) storage, greenhouse gas (GHG) emissions and nutrient leaching.

Soil property	Dynamic property that changes over time	Static property that doesn't change over time	Direct effect on C storage, GHG emissions and nutrient leaching	Indirect effect on C storage, GHG emissions and nutrient leaching
Structure	√		√	
Organic matter content	√		√	
Inorganic and organic nitrogen content	√		√	
pH	√		√	
Texture		√		√
Mineralogy		√	√	
Porosity	√		√	
Drainage	√			√
Compaction status	√			√
Temperature	√		√	
Moisture/water content	√		√	
Land use/management[a]	√		√	

[a]Land use and management activities are important because they can damage soil structure and change dynamic soil properties. Soil properties are classified as dynamic or static and having a direct or indirect effect on these functions. Soil properties which themselves are influenced by soil structure are shown in italics.

influence on the three soil functions. The soil quality focused on here is soil structure. Texture, organic matter (OM), drainage and compaction status have both a direct effect and an indirect effect on the three functions – the latter via their influence on soil structure.

7.2.1 Soil carbon storage and soil structure

Soil organic carbon (SOC) plays a critical role in supporting the productive capacity of soils and their ability to store atmospherically derived carbon dioxide (CO_2) from photosynthesis. Plant material can be consumed by animals or become humified soil organic matter (SOM) through the action of micro-organisms (CAST, 2011). Storage of C as SOC is controlled by the soil environment and the quality of the OM in which the C resides. Formation of recalcitrant materials is one mechanism of protection (Ball, 2013). Roots comprise c.40% C and the greater the plant root and shoot mass, the greater the C input from root systems, additional surface litter and dung from any grazing animals (Shepherd, 2009). Estimates of total belowground C input into soils by cereals and grasses are c.1500 and c.1750 kg C ha^{-1} $year^{-1}$, respectively (Kuzyakov and Domanski, 2000).

The global soil C inventory is estimated to be 2500 Pg C to 1 m depth, comprising 1500 Pg of SOC and 950 Pg of soil inorganic C. The global SOC pool holds approximately four times more C than that stored in vegetation and double that in the atmosphere (IPCC, 2013). SOC levels are determined by the balance of net OM inputs (e.g. vegetation, crop residues, organic amendments) and net losses of C from the soil through decomposition, dissolved organic C (DOC) export and soil erosion (Cloy *et al.*, 2012; Cloy and Smith, 2015).

Decomposition rates and turnover of C in soils are controlled by water regimes, temperature, litter addition and rooting as well as the distribution of finer particles of C within the soil matrix and interactions with reactive minerals (e.g. clay surfaces) (Oades, 1988; CAST, 2011). Soil OM has a dominant effect on soil structure through its role in the formation of stable aggregates (e.g. via bonding of polysaccharides to clay minerals) (Oades, 1988). Aggregations of clay particles and OM are responsible for stabilizing soil structure and protecting C from microbial decomposition through occlusion within aggregates or small micropores (Oades, 1988).

A unique set of factors such as climate, waterlogging and low nutrient status and low pH are responsible for C storage and accumulation in organic soils such as peats (CAST, 2011). Soil C will only accumulate in mineral soils (such as agricultural soils) under a set of 'ideal' conditions and if their maximum soil C storage capacity has not been reached (Krull *et al.*, 2001). Soils will gain SOC if the rate of C addition is greater than the rate of C loss through decomposition and DOC export. If these rates are the same, the SOC levels are at steady state, that is, C inputs = outputs and total SOC is neither increasing nor decreasing. Soils will lose SOC if rates of C input are less than rates of C loss (Shepherd, 2009).

Land use and management are particularly important anthropic controls on soil C storage, with SOC levels generally following the order forest > grassland > arable land (Sakrabani *et al.*, 2012). Disruption of soil aggregates and deterioration of soil structure through management practices and mechanical stress such as tillage, trampling and erosion accelerates microbial oxidation of SOC and makes it more available as DOC (Oades, 1988; Hillel and Rosenzweig, 2009). Intensive use of nitrogen (N) fertilizers to increase crop yields is generally thought to increase SOC sequestration by increasing crop residue inputs, but Khan *et al.* (2007) observed a decline in SOC levels after 40–50 years of synthetic N fertilizer, despite increasing residue incorporation. Manure additions can result in greater and longer-lasting C sequestration than the addition of equivalent amounts of N as mineral fertilizer (Diacono and Montemurro, 2010), but initial soil C contents are important. A decline in SOC levels for soils from a long-term experiment receiving annual manure applications was attributed to a continuing decline in native SOM originating from vegetation that preceded cultivation (Christensen and Johnston, 1997).

High levels of inorganic water-soluble N and phosphorus (P) are reported to prevent the formation of stable forms of soil C (such as humus) due to the inhibition of the fungi and bacteria essential to C sequestration (Mulvaney *et al.*, 2009; Czarnecki *et al.*, 2013; Jones, 2014). Good farm management practices including rotational grazing management, maintaining good residual pastoral levels and cover crops, and avoiding high applications of mineral fertilizers can increase soil C levels significantly, particularly in the subsoil (Ampt and Doornbos, 2010; Jones, 2011). Differences in grassland soil C storage for soils receiving low amounts of NPK, slow-release or organic fertilizers vs. soils receiving high amounts of mineral fertilizer are illustrated in Fig. 7.1 (Shepherd, 2009).

Poor soil quality and fertility are associated with a decline in OM. Key benefits of increasing SOM are improved resistance to compaction (characterized by compression of soil aggregates and reduction in pore size and continuity), nutrient conservation and improved water infiltration and retention due to better soil structure (Kibblewhite *et al.*, 2008). Soil C sequestration and enhancing the photosynthetic capacity of plants to absorb atmospheric CO_2 play a major role in regulating the emission of GHGs.

7.2.2 Soil greenhouse gas exchange and soil structure

The 'greenhouse effect' is the enhanced warming of the Earth's surface and lower atmosphere due to additional emissions of GHGs as a result of human activity and the subsequent increase in adsorption of infrared radiation. The most important individual GHG is CO_2, but substantial contributions to global warming are made by trace gases, including methane (CH_4) and nitrous oxide (N_2O), both of which come in part from soil emissions. The Global Warming Potential (GWP) of a GHG is a measure of its contribution to atmospheric global warming over a fixed period of time relative to that of CO_2, which is assigned a GWP of one. One kilogram of CH_4 has a GWP 25 times greater than 1 kg of CO_2, over a 100-year period, while the GWP of 1 kg of N_2O is nearly 300 times greater (Cloy and Smith, 2015).

Soil porosity, temperature, SOM content, soil mineralogy, pH and soil N content influence the emission and exchange of GHGs at a range of scales (Table 7.1). Climatic conditions, particularly rainfall and temperature, also regulate GHG emissions from soil directly through their influence on microbial activity. Soil structure and properties such as texture and drainage

Fig. 7.1. Soils under dairy grazing. (a) Soil profile with increasing carbon (C) levels, dark soil colour, good potential rooting depth and pasture colour and growth compared with urine patches, moderately good earthworm numbers and root length and root density. (b) Soil profile with steady-state C levels, light soil colour, moderate earthworm numbers, potential rooting depth, pasture colour and growth compared with urine patches and moderately poor root length and root density. The difference between (a) and (b) soil profiles was related to long-term differences in management and land use. NPK, slow-release bio-fertilizers (fertilizers containing living micro-organisms that promote rhizosphere activity and plant growth) were applied in (a), whereas highly soluble, synthetic fertilizers and mineral N were applied at higher rates in (b). (From Shepherd, 2009.)

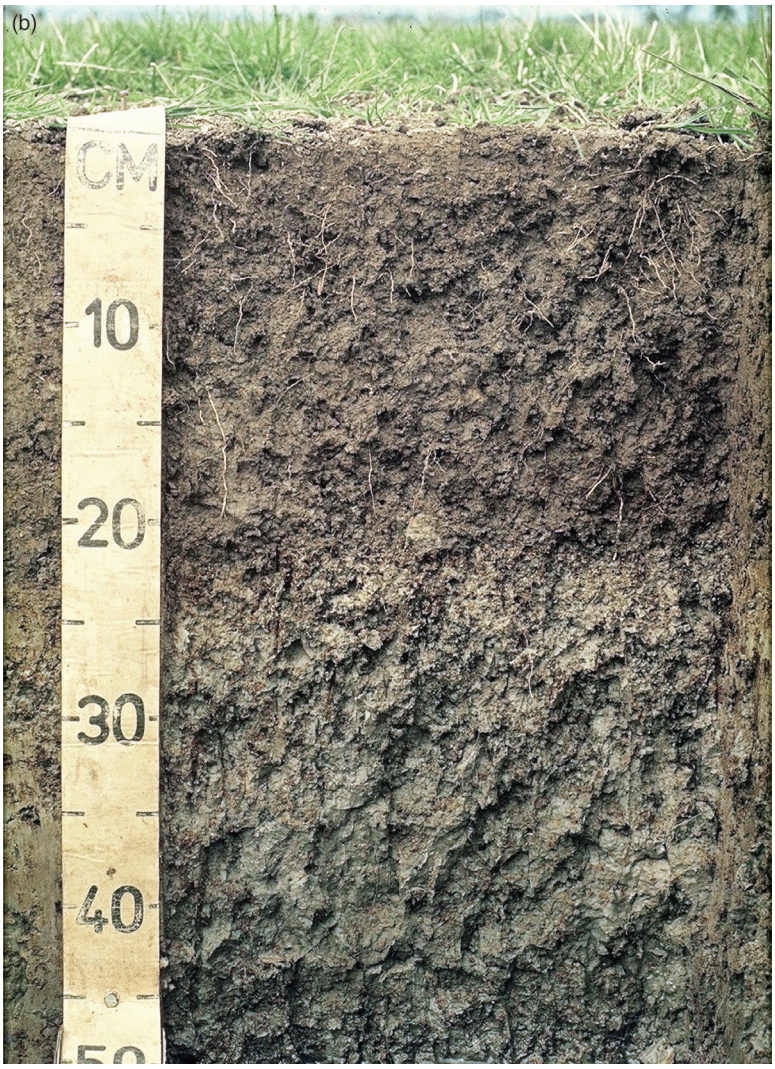

Fig. 7.1. Continued.

affect emissions indirectly by their influence on other properties mainly soil moisture content and distribution, compaction status, pore size and continuity, and the distribution of organic residues (Ball, 2013). Soil porosity or more specifically water-filled pore space (WFPS) – the proportion of porosity filled with water – has a major bearing on the generation and release of GHGs. As WFPS increases to saturation, CO_2 and N_2O, and finally CH_4 are emitted (Fig. 7.2). As macropores, mesopores and pore continuity decrease due to compaction, saturation is reached more quickly and lasts longer so that more GHGs are emitted than from well-structured and well-aerated soils with good porosity and inter-pore drainage (Shepherd, 2009). Clayey soils are often poorly drained and are therefore more likely to emit GHGs (Shepherd, 2009). Structure can override the influence of texture in regulating gas exchange mainly because of its substantial influence on soil water content and pore continuity in soils of the same type (Ball *et al.*, 2013a). Poorly drained soils generally emit greater amounts of GHGs than well-drained soils (Cloy and Smith, 2015). The role of soil structure will now be discussed for CO_2, N_2O and CH_4 in relation to their production in soils.

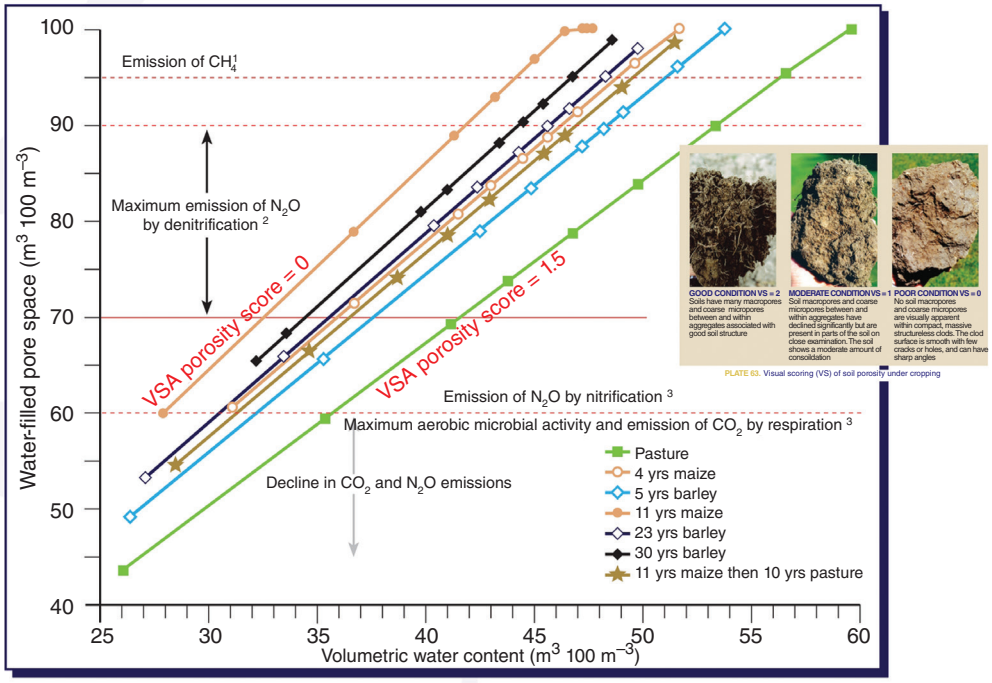

Fig. 7.2. Water-filled pore space (WFPS) and water content at which greenhouse gases are emitted in a Kairanga silty clay soil under pasture and at varying degrees of structural degradation under increasing periods of continuous cropping and conventional cultivation. [1] MacDonald *et al.* (1996); [2] Dobbie *et al.* (1999); [3] Linn and Doran (1984). (From Shepherd, 2009.)

7.2.2.1 CO_2

The main mechanism for CO_2 release is microbial decomposition of plant material and SOM *via* aerobic decay processes in aerated soils. Both CO_2 and CH_4 are released during slow anaerobic decay processes in waterlogged soils (Cloy and Smith, 2015):

$$\text{Aerobic decay: } CH_2O + O_2 \rightarrow CO_2 + H_2O$$

$$\text{Anaerobic decay: } 2CH_2O \rightarrow CH_4 + CO_2$$

Spatial location of OM within the soil matrix determines physical accessibility to decomposers. Soil structure influences this accessibility and is crucial for decomposition. Physical protection of OM is achieved through aggregation and adsorption of OM on mineral surfaces.

Decomposition rates are also modified by temperature and moisture. Moist well aerated soils with loose, well-aggregated structures, such as those found in sandy loam soils, favour organic carbon (OC) mineralization and CO_2 exchange (Ball *et al.*, 2013a).

Soil CO_2 emissions decrease substantially after heavy rainfall because high WFPS and poor gas diffusivity and air-filled porosity restrict respiration and increase anaerobic conditions (Ball *et al.*, 2013a). Soil CO_2 emissions increase linearly with increasing water content to a maximum of approximately 60% WFPS before decreasing (see Fig. 7.2) (Linn and Doran, 1984; Shepherd, 2009). For soils with different clay contents, Franzluebbers (1999) found that cumulative C mineralization during a 24-day incubation at 25°C increased with increasing WFPS to a maximum of 0.53–0.66 $m^3\ m^{-3}$. Clay content had no effect on the level of WFPS required to achieve maximum C mineralization under compressed and uncompressed conditions (Fig. 7.3). Unlike cumulative C mineralization, net N mineralization (i.e. conversion of organic forms of N to inorganic N) decreased strongly when the 'optimum' WFPS level of 0.53–0.66 $m^3\ m^{-3}$ was exceeded and net N mineralization approached zero at values near 0.8–0.9 $m^3\ m^{-3}$ (Fig. 7.3) (Franzluebbers, 1999).

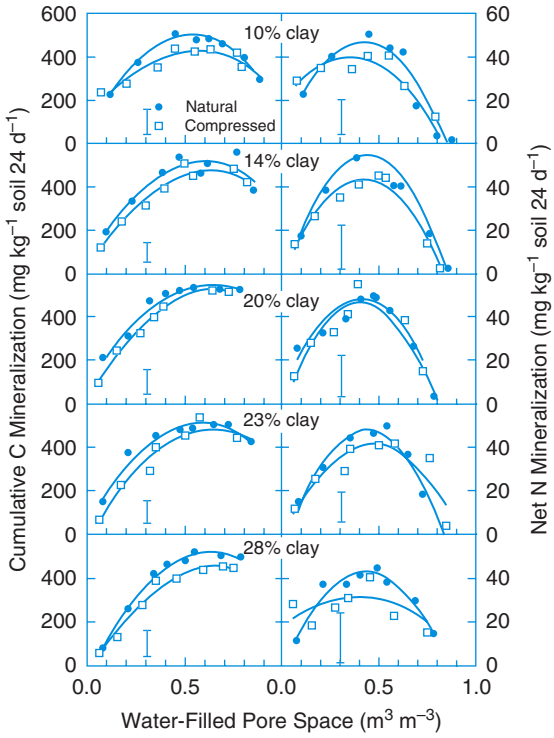

Fig. 7.3. Cumulative carbon (C) mineralization and net nitrogen (N) mineralization as affected by bulk density, clay content and water-filled pore space. Observations are means of three soils. Error bars are least significant difference at $P \leq 0.05$. (From Franzluebbers, 1999.)

Most agricultural soils contain low SOC concentrations relative to unmanaged natural soils because of higher rates of mineralization accelerated by soil temperature and moisture regimes, lower input of biomass C and higher losses caused by accelerated erosion and leaching. The use of high yielding plant varieties, fertilizers, irrigation and residue management can reduce CO_2 losses and enhance uptake within managed areas (Lal, 2010). Several researchers have discovered the quick release of CO_2 in the 1 or 2 days immediately after ploughing (Reicosky, 1997) or even the hours after ploughing (Vinten *et al.*, 2002) due to the flush of microbial CO_2 released from the large voids created by the ploughing. It has been proposed that no-till and reduced till practices cause an accumulation of OC (Lal, 2010), but Powlson *et al.* (2014) suggested that no-till may be beneficial for soil quality and adaptation of agriculture to climate change but that its role in mitigation has been widely overstated. The estimation of vegetative cover at the soil surface and the distribution of roots and residues within the topsoil permitted by VSE methods are relevant to CO_2.

7.2.2.2 N_2O

Approximately 65% of all atmospheric emissions of N_2O are from soils (Cloy and Smith, 2015) derived mainly from microbial nitrification and denitrification, which are controlled by soil mineral N content, soil temperature and pH, water and WFPS (Lilly, 1997; Shepherd, 2009). A small soil sink (uptake of N_2O in soil) has been suggested, and attributed to reduction of atmospheric N_2O to molecular N, N_2 during denitrification. Nitrification is the microbial oxidation of ammonium (NH_4^+) to nitrite (NO_2^-) and thence to nitrate (NO_3^-). The nitrification process is fundamentally an aerobic one, for which the presence of molecular oxygen (O_2) is essential.

As for CO_2, emission of N_2O by nitrification increases linearly with increasing soil water content to a maximum of 60% WFPS and then decreases (see Fig. 7.2) (Shepherd, 2009).

The other main microbial process producing N_2O is denitrification, involving the reduction of NO_3^-, in the absence of O_2. Anaerobic zones within the soil profile then occurs, and within these zones NO_3^- is the chemical species that most readily acts as an electron acceptor, and so it becomes reduced by a succession of enzymes, to NO_2^-, NO, N_2O and finally N_2 (Cloy and Smith, 2015):

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$$

While the WFPS needs to be 60–65% for substantial emissions of N_2O to occur (i.e. critical WFPS), the highest emissions occur by denitrification when the WFPS is between 70 and 90% with lowest at WFPS <50% (see Fig. 7.2). The critical WFPS is a major driver of GHG emissions and in finer textured soils the critical WFPS and the subsequent degree of saturation required to generate GHGs decreases so that these soils tend to emit more GHGs than coarser textured soils.

The level of N_2O emissions varies according to soil properties listed in Table 7.1, but the application of fertilizer N has the biggest impact. In soils, soil matric potential, volumetric water content, relative gas diffusivity and WFPS are indicators of soil aeration status (Ball, 2013). Aeration influences WFPS and the air-filled pore network, thereby influencing N_2O production and emission. Blockage of the air-filled pores by water near the surface of compacted soils can dramatically increase N_2O emission and decrease CO_2 emission (Ball, 2013). The fraction of the total gaseous products of denitrification that is actually emitted as N_2O depends heavily on soil structure and soil wetness. On the one hand, small anaerobic microsites may form in an otherwise well aerated soil profile, caused by a localized region of high respiration, and any N_2O formed in the microsite will have a high probability of escaping before being reduced to N_2. On the other hand, the soil may contain large, virtually saturated anaerobic clods, into which NO_3^- ions may diffuse in solution. Any N_2O produced well within the clod can only escape after diffusing to the surface, and is much more likely to be reduced to N_2 before this occurs (Cloy and Smith, 2015).

On grazed land, the deposition of excreted N is spatially very variable. A small patch of soil surface may receive N as urine at a rate equivalent to some hundreds of kilograms per hectare, whereas land between patches receives none. Also, the treading of the soil surface in wet conditions by animal hooves can create localized compacted zones in which water collects and the soil becomes anaerobic. Such factors lead to the occurrence of 'hotspots' of microbial activity and N_2O emissions that vary significantly in size; thus the average overall emission rate from grazed grassland may be two to three times greater than that from N-fertilized grass grown as a crop to be cut for winter feed (Cloy and Smith, 2015). The estimation of surface damage to soil and vegetation in areas of restricted macroporosity and of general greying and mottling, indicative of anaerobic zones, permitted by VSE methods are relevant to N_2O.

7.2.2.3 CH_4

Production and emission of CH_4 only occurs in very wet mineral soils after organic fertilizer is applied or in organic soils. The microbial breakdown of organic compounds in strictly anaerobic conditions, a process called methanogenesis is responsible for CH_4 formation in soils. A very low redox potential is required for this process, and CH_4 production does not begin until reduction of O_2, NO_3^-, iron(iii), manganese(iv) and sulfate (all of which maintain the potential at higher levels) is complete. Such low-redox conditions are predominantly found in soils where prolonged waterlogging is a normal feature, for example, natural wetlands and flooded rice fields.

The CH_4 formed in soils can migrate to the surface and be emitted into the atmosphere. Diffusion can take place in solution from the point of formation in an anaerobic layer, upward to water layers containing O_2, where much of the CH_4 is oxidized and only a fraction outgasses to the atmosphere. If sufficient CH_4 gas is produced, bubbles form in the water layer and force their way to the surface before significant oxidation can occur (Cloy and Smith, 2015).

In well-aerated agricultural soils, uptake of atmospheric CH_4 is more common. Moist well aerated soil conditions favour CH_4 oxidation by methanotrophic bacteria, being optimal in soils of intermediate textures and moderate water contents, which permit a reasonable rate of diffusion through the soil matrix and thus allow oxidation to take place. Soil bulk density and water content, and their consequent effects on gas movement and penetration in the soil profile, have a major impact on the rate of oxidation of atmospheric CH_4 in soils (Cloy and Smith, 2015). Good aeration conditions

provided by the presence of continuous macropores (detectable from measurements of air permeability) are important for access of the atmospheric CH_4 to oxidation sites (Ball *et al.*, 1997). However, excessive soil disturbance and excessive use of nitrogenous products can reduce the capacity of soils to take up and oxidize atmospheric CH_4 as they can reduce the activity of methanotrophic bacteria (Shepherd, 2009). The estimation of the distribution and degree of decomposition of residues, compaction status and of macroporosity and its restriction permitted by VSE methods are relevant to CH_4.

7.2.3 Soil nutrient leaching and soil structure

Poor soil quality and fertility are associated with low nutrient retention and subsequent leaching into groundwater and waterways. The loss of nutrients such N, P, potassium (K), sulfur (S), calcium (Ca), magnesium (Mg) and sodium (Na) has major implications on land quality because it further affects soil health and agricultural productivity adversely and can lead to environmental problems such as accelerated GHG emissions and eutrophication (Siemens *et al.*, 2004; Shepherd, 2009).

Only 40–50% of applied fertilizer N may actually be utilized by plants (Mengel, 1992; Shepherd, 2009). The remaining N is either included in the organic and inorganic soil N pools where it may be utilized by plants in subsequent years, or lost to the environment. Apart from the losses to the atmosphere as N_2O and N_2 (see Section 7.2.2.2), and ammonia (NH_3) via volatilization, N is leached into the groundwater and lost as runoff into waterways (Shepherd, 2009). The potential for nutrient loss via leaching into groundwater and waterways is influenced by soil cation exchange capacity (CEC) and anion exchange capacity (AEC). Soil CEC and AEC provide measures of the ability of soil to adsorb and exchange nutrient cations (e.g. NH_4^+, Ca^{2+}) and anions (e.g. NO_3^-, phosphate (PO_4^{3-})), respectively. Clayey soils have much higher CECs than sandy soils and usually retain more nutrients and OM (Shepherd, 2009). The concentration and composition of SOM, particularly DOC, are also important drivers of nutrient loss, particularly metal cations. Nutrients held within or on OM surfaces can remain and be retained in the solid phase or become associated with and mobilized alongside DOC in the solution phase. Nutrients associated with soil mineral colloid solutions, particularly P, can also be mobilized in this way (Siemens *et al.*, 2004).

Nutrients dissolved or adsorbed on particles in soil water are transported down the soil profile through soil pores or across the soil surface as runoff. Soil matrix pore size, largely determined by soil structure, has a major influence on nutrient transport as DOC and nutrients in macro- and mesopores are subjected to convective transport by seepage and preferential flow (Kalbitz *et al.*, 2000; Shepherd, 2009). The drainage status, hydraulic conductivity and infiltration capacity of soils are properties that change under different soil types and poor management practices such as compaction. These properties have an important influence on nutrient leaching as they determine the ability of water and solutes to move through soil and be soaked up by the soil. The intensive use of well-drained, sandy and coarse loamy soils in the UK was found to produce soil structural damage and enhanced surface water runoff from fields that should naturally absorb winter rain (Palmer and Smith, 2013). Nutrient leaching has generally been found to be greater in sandy soils with low water holding capacity and high hydraulic conductivity than clayey soils with high water holding capacity and low hydraulic conductivity (Palmer and Smith, 2013; Pulido *et al.*, 2014) but reactive mineral colloids, prevalent in clayey soils, can lead to increased leaching of nutrients, particularly P (Siemens *et al.*, 2004). Silty textured soils are prone to capping, where the surface forms a hard crust, preventing water from infiltrating and resulting in water running off the surface (Palmer and Smith, 2013). Land topography and slope also influence nutrient loss. Well-structured soils on flat land with a higher infiltration and permeability are more susceptible to nutrient leaching than poorly structured soils with a slower infiltration and permeability. On the other hand, poorly structured soils with a slower infiltration and permeability on undulating and rolling land are more susceptible to nutrient loss by runoff than well-structured soils (Shepherd, 2009). Also, in poorly structured soils with poor water storage and infiltration capacity there is less opportunity for N uptake by plants, utilization by soil microbes (e.g. denitrification of NO_3^-)

and microbial or chemical immobilization to remove N from the soil solution (Lilly, 1997; Shepherd, 2009). Overall, good soil structure and functional field drainage systems are key properties for achieving soil nutrient retention, but also for good water quality and minimizing flood risk. In general the susceptibility of soils to nutrient and DOC losses increases when soil structure is degraded but soil texture and OM content, which also influence drainage and related properties, may override the influence of soil structure.

Enhanced nutrient use efficiency by plants reduces leaching of nutrients to water courses. Soils with high root density and deep rooting plants have increased capability for nutrient use, reducing the likelihood of leaching compared with soils with shallow, sparse root systems (Shepherd, 2009). Linkages between plants and soil microbes may play a major role in controlling N transformations. For example, fungal-dominated microbial communities were found to enhance N retention and reduce N loss in extensively managed grassland (de Vries and Bargett, 2012). Use of agricultural practices that encourage N to remain in the root zone long enough for plant uptake will help to prevent N pollution (Lilly, 1997). Under certain environmental conditions, nitrification (conversion of NH_4^+ to NO_3^-, see Section 7.2.2.2) occurs rapidly. In soils subject to leaching, nitrification inhibitors can be applied to slow nitrification and delay N losses (Lilly, 1997; Cameron *et al.*, 2013). Aeration equipment can also be used to improve structure and thence soil infiltration and nutrient movement (Lilly, 1997).

7.3 Estimation of Soil C Storage, GHG Emissions and Nutrient Leaching Using Visual Techniques

Quantitative indicators of flow and macroporosity have been shown to relate to visual evaluation scores and clearly show the relevance of such scores to properties governing GHG emissions and nutrient leaching. Shepherd (2003) found that the VSA structure score related well to saturated hydraulic conductivity and that the VSA porosity score related well to macroporosity in a range of soils from New Zealand (Fig. 7.4).

Guimarães *et al.* (2013) showed that air permeability correlated negatively and significantly ($P < 0.01$) with visual evaluation of soil structure (VESS) on two contrasting soil types and cite that air permeability (Ka) values <1 μm^2 can be used as a reference for impermeable soils that can restrict aeration. Such values occurred mostly between VESS Sq3 and Sq5. McQueen and Shepherd (2002) also reported air permeability to be a good indicator of soil structural degradation, particularly at a water potential of −10 kPa.

Examples of soil properties that can be observed visually to assess potential for functions of soil C storage, GHG emissions and nutrient leaching are provided below. Indicators for the above functions have been produced using modified scorecards of the visual soil assessment (VSA) technique for scoring soil quality and plant performance using soil and plant indicators.

7.3.1 Soil C storage

For soil C storage, indicators for the scorecards are: (i) textural group; (ii) clay mineralogy; (iii) soil colour; (iv) earthworm numbers; (v) potential rooting depth; and (vi) root length and density (Shepherd, 2009). Other indirect, non-soil visual indicators required include crop/pasture growth, colour and growth relative to urine patches (for pasture), the amount and form of fertilizer and N applied, and method of cultivation (for cropping) (Shepherd, 2009). Measured changes in C storage and the VSA Soil C Index of a soil under dairying in the Manawatu Region of New Zealand demonstrated the close relationship between measured and observed values (Table 7.2). Total SOC declined initially over time reaching a steady state (neither gaining nor losing C) with a VSA Soil C Index of 21 (Shepherd, 2009).

Grassland compaction can impair the ability of soil to store C and to allow water infiltration. Newell-Price *et al.* (2013) conducted a survey of grassland soil compaction in England and Wales using both the VSA technique and regular physical measurements of soil compaction. They found that the most important factors influencing VSA ranking scores, alongside compaction status, were SOM content (positive

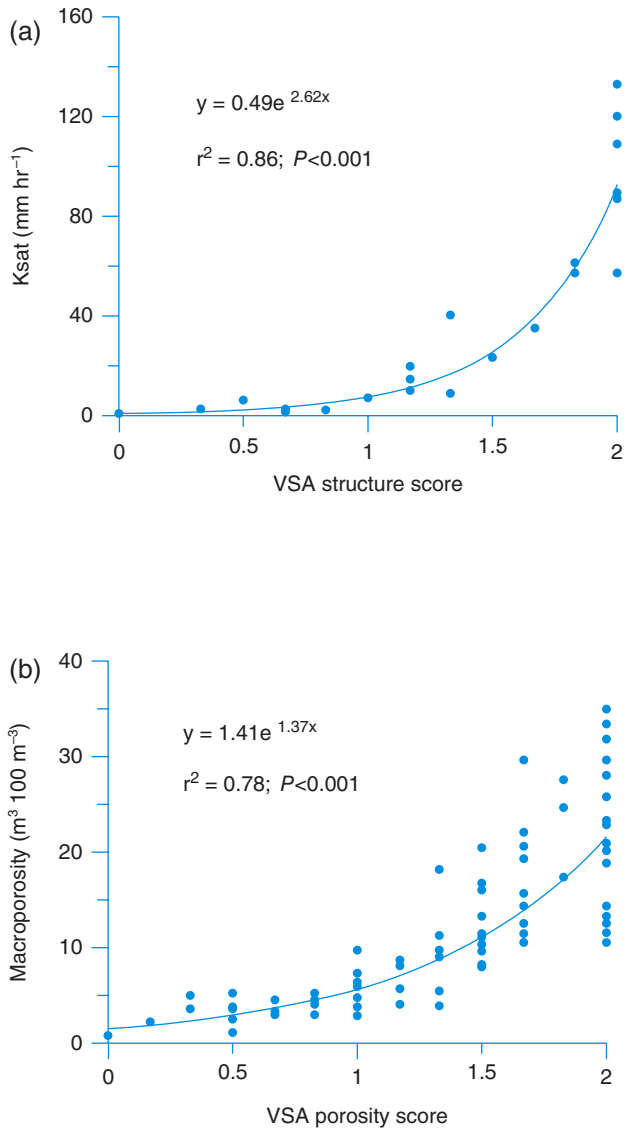

Fig. 7.4. Relationship between (a) VSA field structure and saturated hydraulic conductivity (Ksat), and (b) VSA porosity scores and macroporosity (>30 μm) measured in core samples in a range of soils in New Zealand. (From Shepherd, 2003.)

relationship) and soil sand content (positive relationship), indicating the potential for these visual techniques to estimate SOC content.

The visual property most indicative of C storage that the VSA and VESS techniques make use of is soil colour. Munsell soil colour charts developed in the early 1900s are frequently used to make colour assessments. Soil OM (and therefore SOC) contents can be roughly estimated using soil colour. Generally the darker brown the soil, the higher the OM content (see Fig. 7.2) but the role of soil texture, moisture, carbonate and mineral contents on soil colour should be included (Escadafal *et al.*, 1989). For example, Fe oxides and hydroxides have characteristic colours: hematite alpha-Fe_2O_3

Table 7.2. Changes in soil carbon (C) storage versus the VSA Soil C Index scores in the top 10 cm of a fine clayey soil[a] under dairying over time.

Year	Total organic C (g kg^{-1})	Bulk density (Mg m^{-3})	Total organic C (t ha^{-1})	Soil C Index[b]
1982	56.0[c]	1.02	57.12	31.5
1988	55.0[d]	1.03	56.65	31.5
1989	52.4[d, e]	1.03	53.97	24.5
1992	51.0[f]	1.00	51.00	21
1997	49.9[g]	1.03	51.40	21

[a]Kairanga silty clay loam soil (Eutric Gleysol, FAO classification; fine, mixed, mesic, Typic Endoaquept, Soil Survey Staff, 2014) formed from quartzo-feldspathic alluvium; [b]Shepherd (2009); [c]Shepherd (1992); [d]Sparling *et al*. (1992); [e]Shepherd *et al*. (2001); [f]McQueen and Shepherd (2002); [g]Saggar *et al*. (2001).

(brownish red), magnetite Fe_3O_4 (blackish grey), wustite FeO (greyish blue), goethite alpha-FeOOH (bright yellowish brown) and lepidocrocite gamma-FeOOH (orange) and ferrihydrite Fe_2O_3 (reddish brown) (Schwertmann, 1993). Colour chips in Munsell colour charts can be used to visually estimate a soil's SOM content. For example, Wills *et al*. (2007) used Munsell colours to show that SOC could be predicted from field measurements and that separating samples by land use improved predictions. Limitations to the use of the Munsell charts were individual perceptions of colour, soil type and water content. For reassurance, it is good practice for farmers to consider getting SOM contents measured at the same time as soil fertility testing, bearing in mind that SOM contents can change very slowly. Sonneveld *et al*. (2014) used Munsell colour chart and SOM content data from a national soil survey database for soil series in a dairy farming area of the Netherlands to develop a scheme for visually scoring soil colour at farm level. Colour scores using their visual soil examination and an evaluation method that was based on the VSA approach were defined as 0 for Munsell value >4, 1 for Munsell value = 4 or 3.5 and 2 for Munsell value ≤3, where 0 reflected poor condition and lower SOM content than 1 (moderate condition) and 2 (good condition).

Greater pasture growth might be expected to be associated with greater removal of CO_2 from the atmosphere. Soil C loss, associated changes in soil colour and CO_2 emissions from soils that were initially under pasture before a switch to continuous maize and barley cropping using conventional cultivation are shown in Fig. 7.5. There is a quick and significant decline in total C from 90.8 t C ha^{-1} in the upper 20 cm to 69.8 t C ha^{-1} in just 4 years under cropping. Soil colour becomes lighter with increasing C loss and, if we assume that 1 t of OC oxidizes to 3.67 t of CO_2, the loss of 31.6 t C ha^{-1} after 11 years of conventionally cultivated maize results in the emission of approximately 116 t CO_2 ha^{-1} (Fig. 7.5). The loss of 49.6 t C ha^{-1} after 35 years of continuous barley produces 182 t CO_2 ha^{-1}.

As well as cropping systems and type of cultivation, the degree of soil cover by vegetation also influences soil decomposition and CO_2 emissions (CAST, 2011). This can be estimated visually by the VSA or as surface condition (e.g. Soil Quality Scoring Procedure of Ball and Douglas, 2003).

7.3.2 GHG emissions

For GHG emissions (principally N_2O) the indicators for the scorecards are: (i) textural group; (ii) porosity; and (iii) soil mottles and colour (Shepherd, 2009). Other indirect, non-soil visual indicators include pasture/crop quality; pasture/crop yield; colour and growth relative to urine patches (for pasture); the amount and form of N applied; and method of cultivation (for cropping). For pastures, stocking rate is also a useful indicator for the N deposited in the form of animal urine and dung, which are major sources of N_2O (Shepherd, 2009). Pasture quality can be used as a visual indicator of N_2O and CH_4 emissions because the efficiency and function of rumen microorganisms are reduced when pasture quality is poor (e.g. high

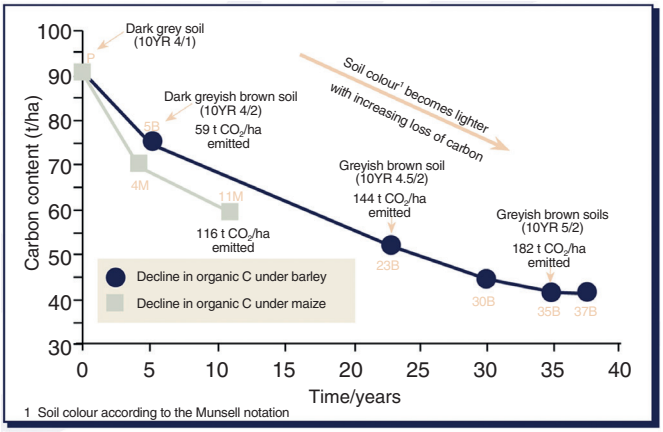

Fig. 7.5. Soil carbon loss, associated soil colour and CO_2 emissions from soils that were under pasture (P) before a switch to continuous maize (M) and barley (B) cropping using conventional cultivation. Data point labels indicate crop and number of years since the experiment started. Soil colour descriptions according to Munsell notation are also labelled. (From Shepherd, 2009.)

levels of NO_3^--N, low sugar and therefore energy levels) (McAllister *et al.*, 1996; Shepherd, 2009). Poor quality pasture results in high emissions of CH_4 (and NH_3) from livestock and excretion of N-rich urine. Poor growth and chlorosis-induced yellow pasture between urine patches with strong growth indicates poor quality pasture with reduced plant N uptake (Bolan and Kemp, 2003) and the subsequent release of N as N_2O and leached NO_3^--N (Shepherd, 2009).

Although gaseous exchange is not related directly to the topsoil appearance, assessment of soil structure changes with depth using VESS is important in identifying layers active in the production and transmission of gases or layers that restrict gas exchange or are likely to be anaerobic (Ball, 2013; Ball *et al.*, 2013a). These zones are where further measurements of soil properties related to aeration status and mineral N might be assessed (Ball *et al.*, 2013b).

Organic carrot production involves considerable tractor traffic for the many management operations, including mechanical and hand weeding. Ball (2013) studied soil damage under a former tractor 'tramline' route at an organic carrot production site in east Scotland. Residual compaction damage resulted in very poor soil structure with a VESS score of Sq5 because the soil consisted mostly of very large, compact clods with minimal macroporosity. High soil moisture contents, combined with the presence of straw used to protect the former carrot crop, resulted in anaerobic conditions shown by the grey-blue appearance of the soil below the straw layer. This was confirmed by the large N_2O emissions from cores taken at 15–20-cm depth.

No-tillage is effective in many countries for controlling erosion by preserving soil structure. The aspect of erosion control is particularly important in Brazil where the influence of structural changes due to surface compaction on GHG emissions was investigated. Intact cores of Oxisol clays taken from a southern Brazilian field site under long-term no-tillage were used to assess VESS Sq score and CO_2 and N_2O fluxes along a transect aligned so that the looser areas within the crop rows and the compacted areas between the rows (interrows) were alternately sampled (see Fig. 7.6). VESS scores and physical properties were more favourable in the crop rows than in the compacted interrows and these changes were found to affect soil CO_2 and N_2O emissions (da Silva *et al.*, 2014).

Soil structural damage from animal treading is expected to increase soil N_2O emissions and to limit C storage, thereby impairing the C balance of pasture dairy farming and long-term sustainability of diary production from pasture. Interactions with N fertilizer application rate

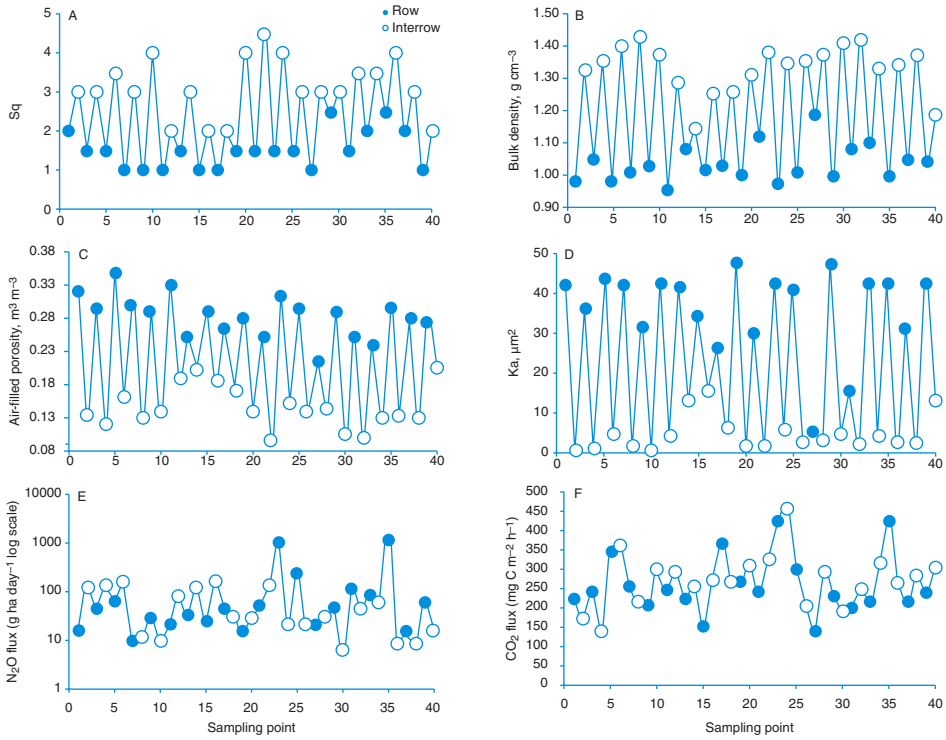

Fig. 7.6. Variation in Oxisol structural quality (VESS Sq score) and CO_2 and N_2O fluxes with sampling point along a transect according to crop row or interrow position in a long-term no tillage experiment in Southern Brazil. (From da Silva *et al*., 2014.)

and type are likely. N uptake can appear poor at high N application rates. To investigate this, Ball *et al*. (unpublished data) measured soil structural and pasture quality using visual techniques (VESS and VSA), alongside other key soil data, to identify N_2O emission potential at farms from an area of intensive dairy production near Palmerston North, New Zealand. Soil sampling site details and results are listed in Table 7.3. Sites 1 to 6 were located on Kairanga silty clay loam soils (Typic Endoaquepts, Soil Survey Staff, 2014), with two each on pasture receiving low, medium and high N applications. Sites 7 and 8 were located on the Manawatu fine sandy loam (Dystric Fluventic Eutrochrept, Soil Survey Staff, 2014), a flood plain soil vulnerable to damage. Farms were chosen according to three levels of N input. At each level, fields containing soils of poor and good quality were identified. The VSA scorecards were used to estimate the likelihood and relative magnitude of N_2O flux at each site from visual estimates of soil and pasture quality. The likely magnitude of N_2O fluxes were confirmed using a simple model of N_2O emissions based on measurements of soil mineral N, WFPS and soil temperature (Conen *et al*., 2000). Poor quality soils were more common at high N inputs (Table 7.3). Nevertheless some poor structures as a result of treading damage were identified at low N inputs. The high N input, poorly structured soils were deemed most likely to emit high levels of N_2O due to their likely high WFPS even at low soil water contents, high soil temperature, low porosity and air permeability, poor aeration status near the soil surface and high exposed soil surface area due to poaching (Ball *et al*., unpublished data).

Soils with stable structure resist compaction damage and have satisfactory macroporosity permitting good water drainage and aeration while retaining sufficient moisture for good crop growth (Ball, 2013; Ball *et al*., 2013a). Soils with

Table 7.3. Details of field sites, N application, structural quality, water-filled pore space (WFPS), air permeability, mineral nitrogen (N) contents, soil temperature and estimated greenhouse gas (GHG) index on two soil types under pasture.

Site[a]	N status[b] (kg ha^{-1} year^{-1})	Soil structure (VSA and VESS)	WFPS (%)	Air permeability (μm^2)	Soil NH_4^+-N content (mg kg^{-1})	Soil NO_3^--N content (mg kg^{-1})	Soil temperature at 5 cm depth (°C)	GHG emission index
1	Low–45	Poor	67	43	4	24	20.3	Mod.–high
2	Low–35	Mod. good	64	137	0.3	25	22.4	Moderate
3	Mod high – 115	Poor	59	52	9.1	11.4	22.4	High
4	Mod high – 250	Mod. good	54	106	2.6	13.8	22.4	Moderate
5	High – 435	Poor	56	68	6.4	8.7	23.4	High
6	High – 435	Mod. poor	54	138	5.9	6.8	23.4	High
7	High – 435	Mod. poor	47	17	20.1	16.5	23.4	Mod.–high
8	High – 435	Mod. good	38	20	12.5	9.9	22.5	Moderate

[a]Soils 1–6 are Kairanga silty clay loams and soils 7–8 are Manawatu fine sandy loams.
[b]N was applied as a foliar spray at sites 1 and 2, and as solid urea at remaining sites.

less stable structure are prone to compaction and low macroporosity (<10% m^3 m^{-3} soil) and high WFPS (>65%) so that poor water drainage and aeration may result. The relationship between soil WFPS and the VSA visual assessment of soil porosity has been proposed as an immediate and effective guide to the susceptibility of a soil to emit GHGs. Figure 7.2 in Section 7.2.2 illustrates the WFPS and water content at which GHGs are emitted from the Kairanga series soils, New Zealand. It demonstrates that moderately well-structured soil with a VSA soil porosity score of 1.5 requires a water content of approximately 42% (v/v) to ensure >70% of the soil pores are water filled and therefore able to generate significant emissions of N_2O. In contrast, a severely compacted soil after 11 years of poorly managed maize cropping with a VSA porosity score of 0 requires a water content of only 33% (v/v) to reach 70% WFPS to increase N_2O emissions significantly. The severely compacted soil will therefore produce more GHGs than the well-structured soil because of the greater number of days during the year when the soil water content is at or above 70% WFPS.

Tractor compaction and animal trampling at a grassland site in south-west Scotland with imperfectly drained clay loam soils were also found to decrease structural quality measured using VESS during two consecutive autumns. This impairment of quality was found to increase soil N_2O emissions (Ball *et al.*, 2013a).

7.3.3 Nutrient leaching

For nutrient leaching the indicators for the scorecards are: (i) textural group; (ii) soil structure; and (iii) potential rooting depth (Shepherd, 2009). Other indirect, non-soil visual indicators include root length and density; root development and soil erosion (for cropping); pasture/crop quality; pasture colour and growth relative to urine patches; the amount and form of fertilizer and N applied; and rainfall. For pastures, stocking rate is also a useful indicator for nutrients deposited in the form of animal urine and dung (Shepherd, 2009). Moderate to good relationships were found between soil visual evaluation scores (VSA and VESS) and hydraulic conductivity measurements in soils with contrasting textures and land uses (Pulido *et al.*, 2014). The potential for nutrient loss on a dry-stock farm adjacent to Lake Taupo in New Zealand on highly permeable, coarse textured pumice soils in a moderately high rainfall area (1480 mm year^{-1}) with medium CEC, was assessed to be low according to the VSA scoring system. The assessment was in close agreement with the low levels of N measured in two streams running through the farm into the lake. For each stream, measured total NH_4^+-N levels were 0.01 g m^{-3}, total Kjeldahl N levels were 0.2 g m^{-3} and measured total NO_3^--N + NO_2^--N levels ranged from 1.5–2.2 g m^{-3} for water samples taken in July 2006.

Soil erosion and leaching can be prevented by protecting the soil surface and improving soil structure. This can be done using vegetation and crop residues, high-residue crops, winter cover crops, no-till or conservation tillage methods, windbreaks, maintaining good soil fertility (especially levels of the soil flocculent Ca), avoiding soil compaction and adding OM (Shepherd, 2003). Maintaining crop residues not only reduces erosion but encourages maximum water infiltration and storage. Estimations of soil leaching based on amounts of fertilizer applied may be limited to N because the relationship between surplus nutrients and leaching to surface waters is more direct for N than for P (van Beek *et al.*, 2003). Assessments of and the use of visual soil techniques to estimate nutrient leaching are not well documented but the use and potential ability of visual field examinations for assessing soil structural quality has been evaluated through comparison with soil physical and hydraulic properties related to soil function (Pulido *et al.*, 2014).

7.4 Future Directions

Soil C storage, GHG emissions and nutrient leaching are key soil functions that are affected by agricultural activities such as changes in the structure of soils, reduction in soil fertility and processes such as soil erosion and destruction of humus (Sakrabani *et al.*, 2012). Soil structure is variable and increasingly vulnerable to compaction and erosion damage as agriculture intensifies and the climate changes (Ball *et al.*, 2013b). As the climate warms and rainfall patterns change, there is a growing risk that soil GHG emissions to the atmosphere will increase, in turn causing further climate change as well as reducing the soil's productive capacity. Suboptimal agricultural soils with C deficits or unstable structure offer an opportunity to absorb CO_2 from the atmosphere and store it as OM in the coming decades and off-set GHG emissions and reduce nutrient leaching (Hillel and Rosenzweig, 2009; Shepherd, 2009). Changes in rainfall intensity and amount, vegetative cover and patterns of land use will affect soil erosion and nutrient leaching (Hillel and Rosenzweig, 2009). Structural stability is improved by addition of OM with a significant labile fraction, which also contributes to the overall capture of OC in the soil (Ball, 2013).

Farmer involvement is crucial in conserving soil quality. Repeated and consistent use of quick visual tools such as the VSA and VESS techniques to survey agricultural land and identify problems in advance should be promoted. Farmers need encouragement to adopt improved management practices that protect soil structure, particularly cost-effective C strategies such as adopting agroecological management (Lal, 2010). Good farm management strategies that ensure good soil structure, aeration, available moisture and nutrient status will increase soil health, earthworm numbers, microbial biomass and activity, and enhance plant root systems and plant photosynthetic rate and capacity (Shepherd, 2009). It is important to check that soil visual techniques and soil quality scoring are reflected in crop yields and any variability in N-fertilizer management that is often visible in the crop (Mueller *et al.*, 2013). Visual soil evaluation can help farmers to identify areas for amendment with OM or tillage in order to improve the three functions: soil C storage, GHG emissions and nutrient leaching. The role of soil physics in conserving N in soil is becoming increasingly important as fertilizer prices increase and as climate change results in soil conditions more conducive to N loss (Ball, 2013). Ball (2013) suggested that mitigation of soil N_2O emissions could involve increasing porosity and reducing moisture content through tillage and drainage. Soil chemistry and biology also play important roles and detailed soil analyses to exploit biologically active and slow release fertilizers is important since mitigation of the ongoing consequences of soil deterioration and NO_3^- pollution of ground and surface waters requires management of N fertilization by site-specific assessment of soil N availability (Khan *et al.*, 2007). Scientists need increasingly to use visual evaluation to target areas for measurements to quantify the three functions and their drivers.

Detailed quantification of soil structure is conducted using measurements or observations of a soil's component aggregates and pores using micromorphology or CT scanning. Future research directions could include the comparison of soil assessments conducted using quick visual tools with these more sophisticated methods for describing soil structure (Garbout

et al., 2013; Munkholm *et al.*, 2013), which can then be linked to measured soil C storage, GHG emission and nutrient leaching processes.

7.5 Conclusions

Agricultural and environmental protection of soils through the maintenance of good soil quality and the implementation of good farm management practices are key improvements as they can reduce degradation of C stores, losses of GHGs and nutrient leaching. Poor soil quality is associated with a decline in OM, high emissions of N_2O, low uptake of CH_4 and poor nutrient retention. Improving and maintaining the physical condition of the soil is an effective means of mitigating GHG emissions. Soils with good structure resist erosion and nutrient loss. Virtually all soils need a moderately good to good structure through the soil profile in order to function effectively as a growing medium. In particular, soil structure influences the main soil and plant root functions: aeration; drainage; root development. Without structure, soils will suffer from anaerobism, waterlogging and nutrient lock-up and, ultimately, crops will fail. Assessment of agricultural land in terms of soil quality and soil structure using quick visual tools such as the VSA and VESS techniques provide an indication of the potential for soils to store C, release GHGs and lose nutrients, and are therefore critical for helping farmers identify problems as well as combat environmental change.

References

Ampt, P. and Doornbos, S. (2010) *Communities in Landscapes Project Benchmark Study of Innovators*. Gulgong, Central West Catchment, New South Wales, University of Sydney, Sydney, Australia, pp. 1–19.

Ball, B.C. (2013) Soil structure and greenhouse gas emissions: a synthesis of 20 years of experimentation. *European Journal of Soil Science* 64, 357–373.

Ball, B.C. and Douglas, J.T. (2003) A simple procedure for assessing soil structural, rooting and surface conditions. *Soil Use and Management* 19, 50–56.

Ball, B.C., Dobbie, K.E., Parker, J.P. and Smith, K.A. (1997) The influence of gas transport and porosity on methane oxidation in soils. *Journal of Geophysical Research* 102, 23, 301–323, 308.

Ball, B.C., Hargreaves, P.R. and Cloy, J.M. (2013a) Soil structure and greenhouse gas emissions. *Proceedings of the International Fertiliser Society Conference.* International Fertiliser Society, York, UK.

Ball, B.C., Munkholm, L.J. and Batey, T. (2013b) Applications of visual soil evaluation. *Soil and Tillage Research* 127, 1–2.

Bolan, N.S. and Kemp, P.D. (2003) A review of factors affecting and prevention of pasture-induced nitrate toxicity in grazing animals. *Proceedings of the New Zealand Grassland Association* 65, 171–178, New Zealand Grassland Association Inc., Dunedin, New Zealand.

Council for Agricultural Science and Technology (CAST) (2011) *Carbon Sequestration and Greenhouse gas Fluxes in Agriculture: Challenges and Opportunities*. Task Force Report No. 142, Ames, Iowa.

Cameron, K.C., Di, H. and Moir, J.L. (2013) Nitrogen losses from the soil/plant system: a review. *Annals of Applied Biology* 162, 145–173.

Cavalieri, K.M.V., Silva, A.P., Tormena, C.A., Leáo, T.P., Dexter, A.R. and Hakansson, I. (2009) Long-term effects of no-tillage on dynamic soil physical properties in a Rhodic Ferralsol in Paraná, Brazil. *Soil and Tillage Research* 103, 158–164.

Christensen, B.T. and Johnston, A.E. (1997) Soil organic matter and soil quality – lessons learned from long-term experiments at Askov and Rothamsted. In: Gregorich, E.G. and Carter, M.R. (eds) *Soil Quality for Crop Production and Ecosystem Health*. Elsevier, Amsterdam, the Netherlands.

Cloy, J.M. and Smith, K.A. (2015) Greenhouse gas emissions. In: *Reference Module in Earth Systems and Environmental Sciences*. Elsevier, Oxford, UK.

Cloy, J.M., Rees, R.M., Smith, K.A., Goulding, K., Smith, P., Waterhouse, A. and Chadwick, D.R. (2012) Impacts of agriculture upon greenhouse gas budgets. In: Hester, R.E. and Harrison, R.M. (eds) *Environmental Impacts of Modern Agriculture.* Issues in Environmental Science and Technology, Issue 34, Royal Society of Chemistry, Cambridge, UK.

Conen, F., Dobbie, K.E. and Smith, K.A. (2000) Predicting N_2O emissions from agricultural land through related soil parameters. *Global Change Biology* 6, 417–426.

Czarnecki, O., Yang, J., Weston D.J., Tuskan, G.A. and Chen, J.G. (2013) A dual role of strigolactones in phosphate acquisition and utilization in plants. *International Journal of Molecular Sciences* 14, 7681–7701.

da Silva, A.P., Ball, B.C., Tormena, C.A., Giarola, N.F.B. and Guimarães, R.M.L. (2014) Soil structure and greenhouse gas production differences between row and interrow positions under no-tillage. *Scientia Agricola* 71, 157–162.

de Vries, F.T. and Bardgett, R.D. (2012) Plant–microbial linkages and ecosystem nitrogen retention: lessons for sustainable agriculture. *Frontiers in Ecology and the Environment* 10, 425–432.

Diacono, M. and Montemurro, F. (2010) Long-term effects of organic amendments on soil fertility. A review. *Agronomy for Sustainable Development* 30, 401–422.

Dobbie, K.E., McTaggart, I.P. and Smith, K.A. (1999) Nitrous oxide emissions from intensive agricultural systems: variations between crops and seasons, key driving variables, and mean emission factors. *Journal of Geophysical Research* 104, 26,981–26899.

Escadafal, R., Girard, M-C. and Courault, D. (1989) Munsell soil color and soil reflectance in the visible spectral bands of Landsat MSS and TM data. *Remote Sensing of Environment* 27, 37–46.

Franzluebbers, A.J. (1999) Microbial activity in response to water-filled pore space of variably eroded southern piedmont soils. *Applied Soil Ecology* 11, 91–101.

Garbout, A., Munkholm, L.J. and Hansen, S.B. (2013) Tillage effects on topsoil structural quality assessed using x-ray CT, soil cores and visual soil evaluation. *Soil and Tillage Research* 128, 104–109.

Guimarães, R.M.L., Ball, B.C., Tormena, C.A., Giarola, N.F.B. and da Silva, A.P. (2013) Relating visual evaluation of soil structure to other physical properties in soils of contrasting texture and management. *Soil and Tillage Research* 127, 92–99.

Haygarth, P.M. and Ritz, K. (2009) The future of soils and land use in the UK: soil systems for the provision of land-based ecosystem services. *Land Use Policy* 26S, S187–S197.

Hillel, D. and Rosenzweig, C. (2009) Soil carbon and climate change. *Crops, Soils, Agronomy News.* V54, N06, American Society of Agronomy, Madison, Wisconsin.

Intergovernmental Panel on Climate Change (IPCC) (2013) *Climate Change 2013: The Physical Science Basis.* Contribution of Working Group 1 to the Fifth Assessment Report. University Press, Cambridge, UK.

Jones, C.E. (2011) *Carbon that Counts. New England and North West Landcare Adventure.* Guyra, New South Wales, University of Sydney, Australia 16–17 March, pp. 1–5.

Jones, C.E. (2014) Nitrogen: the double-edged sword. *WANTFA New Frontiers in Agriculture* 22, 58–61.

Kalbitz, K., Solinger, S., Park, J.H., Michalzik, B. and Matzner, E. (2000) Controls on the dynamics of dissolved organic matter in soils: a review. *Soil Science* 165, 277–304.

Khan, S.A., Mulvaney, R.L., Ellsworth, T.R. and Boast, C.W. (2007) The myth of nitrogen fertilization for soil carbon sequestration. *Journal of Environmental Quality* 36, 1821–1832.

Kibblewhite, M.G., Ritz, K. and Swift, M.J. (2008) Soil health in agricultural systems. *Philosophical Transactions of the Royal Society B* 363, 685–701.

Krull, E., Baldock, J. and Skjemstad, J. (2001) Soil texture effects on decomposition and soil carbon storage. *Net Ecosystem Exchange Workshop Proceedings*. Cooperative Research Centre for Greenhouse Accounting, Canberra, Australia, pp. 103–110.

Kuzyakov, Y. and Domanski, D. (2000) Carbon input by plants into the soil. Review. *Journal of Plant Nutrition and Soil Science* 163, 421–431.

Lal, R. (2010) Enhancing eco-efficiency in agro-ecosystems through soil carbon sequestration. *Crop Science* 50, 120–131.

Lilly, J.P. (1997) *Soil Facts Best Management Practices for Agricultural Nutrients.* North Carolina Cooperative Extension Service, Publication AG-439-20, Raleigh, North Carolina.

Linn, D.M. and Doran, J.W. (1984) Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and non tilled soils. *Soil Science Society of America Journal* 48, 1267–1272.

MacDonald, J.A., Skiba, U., Sheppard, L.J., Hargreaves, K.J. and Fowler, D. (1996) Soil environmental variables affecting the flux of methane from a range of forest, moorland and agricultural soils. *Biogeochemistry* 34, 113–132.

McAllister, T.A., Cheng, K.-J., Okine, E.K. and Mathison, G.W. (1996) Dietary, environmental and microbiological aspects of methane production in ruminants. *Canadian Journal of Animal Science* 76, 231–243.

McQueen, D.J. and Shepherd, T.G. (2002) Physical changes and compaction sensitivity of a fine-textured, poorly drained soil (typic endoaquept) under varying durations of cropping. *Soil and Tillage Research* 63, 93–107.

Mengel, K. (1992) Nitrogen: agricultural productivity and environmental problems. In: Mengel, K. and Pilbeam, D.J. (eds) *Nitrogen Metabolism of Plants*. Clarendon Press, Oxford, UK.

Mueller, L., Shepherd, G., Schindler, U., Ball, B.C., Munkholm, L.J. *et al*. (2013) Evaluation of soil structure in the framework of an overall soil quality rating. *Soil and Tillage Research* 127, 74–84.

Mulvaney, R.L., Khan, S.A. and Ellsworth, T.R. (2009) Synthetic nitrogen fertilizers deplete soil nitrogen: a global dilemma for sustainable cereal production. *Journal of Environmental Quality* 38, 2295–2314.

Munkholm, L.J., Heck, R.J. and Deen, B. (2013) Long-term rotation and tillage effects on soil structure and crop yield. *Soil and Tillage Research* 127, 85–91.

Newell-Price, J.P., Whittingham, M.J., Chambers, B.J. and Peel, S. (2013) Visual soil evaluation in relation to measured soil physical properties in a survey of grassland soil compaction in England and Wales. *Soil and Tillage Research* 127, 65–73.

Oades, J.M. (1988) The retention of organic matter in soils. *Biogeochemistry* 5, 35–70.

Palmer, R.C. and Smith, R.P. (2013) Soil structural degradation in SW England and its impact on surface-water runoff generation. *Soil Use and Management* 29, 567–575.

Powlson, D.S., Stirling, C.M., Jat, M.L., Gerard, B.G., Palm, C.A. *et al*. (2014) Limited potential of no-till agriculture for climate change mitigation. *Nature Climate Change* 4, 678–683.

Pulido Moncada, M., Helwig Penning, L., Carlos Timm, L., Gabriels, D. and Cornelius, W.M. (2014) Visual examinations and soil physical and hydraulic properties for assessing soil structural quality of soils with contrasting textures and land uses. *Soil and Tillage Research* 140, 20–28.

Reicosky, D.C. (1997) Tillage-induced CO_2 emission from soil. *Nutrient Cycling in Agroecosystems* 49, 273–285.

Saggar, S. Yeates, G.W. and Shepherd, T.G. (2001) Cultivation effects on soil biological properties, microfauna and organic matter dynamics in Eutric Gleysol and Gleyic Luvisol soils in New Zealand. *Soil and Tillage Research* 58, 55–68.

Sakrabani, R., Deeks, L.K., Kibblewhite, M.G. and Ritz, K. (2012) Impacts of agriculture upon soil quality. In: Hester, R.E. and Harrison, R.M. (eds) *Environmental Impacts of Modern Agriculture.* Issues in Environmental Science and Technology, Issue 34, Royal Society of Chemistry, Cambridge, UK.

Schwertmann, U. (1993) Relations between iron oxides, soil color, and soil formation. In: Bigham, J.M. and Ciolkosz, E.J. (eds) *Soil Colour*. Special Publication 31, Soil Science Society of America, Madison, Wisconsin.

Shepherd, T.G. (1992) Sustainable soil-crop management and its economic implications for grain growers. *Proceedings of the International Conference on Sustainable Land Management*. Hawkes Bay Regional Council, Napier, New Zealand, pp. 141–152.

Shepherd, T.G. (2003) Assessing soil quality using visual soil assessment. In: Currie, L.D. and Hanly, J.A. (eds) *Tools for Nutrient and Pollutant Management: Applications to Agriculture and Environmental Quality*. Occasional Report No. 17. Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand, pp. 153–166.

Shepherd, T.G. (2009) Visual soil assessment. Vol. 1. *Field Guide for Pastoral Grazing and Cropping on Flat Rolling Country*, 2nd edn. Horizons Regional Council, Palmerston North, New Zealand, p. 119.

Shepherd, T.G., Saggar, S. Newman, R.H., Ross, C.W. and Dando, J.L. (2001) Tillage induced changes to soil organic matter fractions and soil structure in New Zealand soils. *Australian Journal of Soil Research* 39, 465–489.

Siemens, J., Ilg, K., Lang, F. and Kaupenjohann, M. (2004) Adsorption controls mobilisation of colloids and leaching of dissolved phosphorus. *European Journal of Soil Science* 55, 253–263.

Soil Survey Staff (2014) Keys to soil taxonomy. *US Department of Agriculture,* 12th edn. Natural Resources Conservation Service*,* Washington DC, p. 360.

Sonneveld, M.P.W., Heuvelink, G.B.M. and Moolenaar, S.W. (2014) Application of a visual soil examination and evaluation technique at site and farm level. *Soil Use and Management* 30, 263–271.

Sparling, G.P. Shepherd, T.G. and Kettles, H.A. (1992) Changes in soil organic C, microbial C, and aggregate stability under maize and cereal cropping, and after restoration to pasture in soils from the Manawatu region, New Zealand. *Soil and Tillage Research* 24, 225–241.

van Beek, C.L., Brouwer, L. and Oenema, O. (2003) The use of farmgate balances and soil surface balances as estimator for nitrogen leaching to surface water. *Nutrient Cycling in Agroecosystems* 67, 233–244.

Vinten, A.J.A., Ball, B.C., O'Sullivan, M.F. and Henshall, J.K. (2002) The effects of cultivation method, fertilizer input and previous sward type on organic C and N storage and gaseous losses under spring and winter barley following long-term leys. *Journal of Agricultural Science* 139, 231–243.

Wills, S.A., Burras, C.L. and Sandor, J.A. (2007) Prediction of soil organic carbon content using field and laboratory measurements of soil color. *Soil Science Society of America Journal* 71, 380–388.

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