Chemical Components and Effects on Soil Quality in Temperate Grazed Pasture Systems

M. H. Beare
New Zealand Institute for Crop & Food Research

D. Curtin
New Zealand Institute for Crop & Food Research

S. Thomas
New Zealand Institute for Crop & Food Research

P. M. Fraser
New Zealand Institute for Crop & Food Research

G. S. Francis
New Zealand Institute for Crop & Food Research

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M.H. Beare, D. Curtin, S. Thomas, P.M. Fraser and G.S. Francis
New Zealand Institute for Crop & Food Research, Canterbury Agriculture and Science Centre, Private Bag 4704, Christchurch, New Zealand, Email: Bearem@crop.cri.nz

Key points

1. Legume/grass pastures retain feedbacks on N supply that may help to reduce losses of N to the wider environment.
2. Intensive use of N fertilisers tends to increase SOM turnover and increase losses of N to the environment.
3. Increased use of fertilisers has contributed little to soil organic matter storage in grazed pastoral systems.

Keywords: soil quality, grassland, pasture, chemical fertility, compaction, GHG emissions

Introduction

Soil quality is commonly defined as the *fitness of soil for a specific use* (Larson & Pierce, 1994). Much of the interest in soil quality pertains to the management of agricultural systems, i.e. the soil's ability to nurture and sustain plant and animal productivity (Beare, 2002), partition and regulate the flow of water, and function as an environmental buffer (Carter *et al*., 1997; Doran & Parkin, 1994). Chemical, physical and biological indicators of soil quality reflect the key properties and processes that support these functions and are aimed at assessing the soil's ability to satisfactorily provide them. In addition to extrinsic factors (e.g. climate), these functions are influenced both by the intrinsic characteristics of a soil (i.e. inherent soil quality) and the properties and processes that are influenced by its use and management (i.e. dynamic soil quality). The properties of greatest importance to soil quality in a particular agricultural system are often grouped into a minimum data set (Gregorich *et al*., 1994). These are best defined by the soil management issues (e.g. nutrient availability, compaction, water storage) that have the most influence on plant and animal performance and impact on the wider environment (Larson & Pierce, 1994; Beare *et al*., 1999). Sustainable production depends on choosing land uses that are well suited to the capability of the soil (and wider environment) and maintaining soil conditions that enhance productivity and environmental quality (Larson & Pierce, 1994; Beare, 2002).

The term 'grassland' covers a wide range of managed and unmanaged ecosystems that differ in their productivity, nutrient cycling, and pathways of herbage removal and return. Grasslands managed for livestock grazing are common in many regions of the world. They include legume/grass and grass-based pastures ranging from extensive low input rain-fed sheep systems to intensive high input irrigated dairy systems. This paper addresses the chemical components of dynamic soil quality in managed grasslands, with a particular focus on temperate grazed pasture systems. It explores how chemical inputs, both natural and anthropogenic, affect the productivity and environmental quality of these pastoral soils. It also describes how management influences chemical properties and processes through effects on the physical and biological condition of grazed pastures, which in turn impact on the sustainable management of these grasslands.

Chemical inputs and fertility

Soil chemical fertility testing represents one of the earliest and best established approaches to soil quality monitoring. Although chemical fertility describes only one aspect of soil quality, it remains an essential component of any comprehensive soil quality assessment programme. Fertility testing of grassland soil usually involves assessment of pH, plant available phosphorus (P), exchangeable cations (K^+, Ca^+, Mg^{2+}, Na^+), and SO_4^{2-} sulphur. Apart from mineral nitrogen (N) (NH_4^+, NO_3^-), widely accepted, commercially available tests for plant available N are decidedly lacking. Micronutrient (e.g. Mn, Cu, Mo, B, Se) testing is not normally included in regular monitoring programmes; rather it is usually directed at identifying specific nutrient deficiencies. Limitations of these tests and implications for soil quality monitoring are discussed below.

Efforts to improve the productivity of grassland systems have relied heavily on the use of fertilisers. Legumes are an important component of grazed pastoral systems in Australia, New Zealand, Western Europe and both North and South America (Peoples *et al*., 2004). In Australia and New Zealand, P and sulphur (S) have traditionally been the main nutrients applied to these pastures, usually in the form of single superphosphate. The benefits of superphosphate for herbage dry matter (DM) production are well established (Haynes & Williams, 1993). Ensuring high P availability is especially important in stimulating clover growth and symbiotic N_2-fixation, which can markedly improve the N status of legume-based pastures (Lambert *et al*., 2000; Ledgard, 2001). Despite the obvious benefits of superphosphate for pasture and animal productivity, it may also have important adverse effects on soil and environmental quality. The increased use of P fertilisers and importation of P in feed, manure and effluent has resulted in accumulation of soil P in many locations worldwide (Haygarth & Jarvis, 1999; Sims *et al*., 2000). Haygarth *et al*. (1998) reported annual accumulation rates of 0.28 and 26 kg/ha for extensive sheep and intensive dairy pastures in the UK, respectively. There is mounting evidence that P transfer from agricultural land to water bodies also increases with adverse impacts on water quality (e.g. Daniel *et al*., 1998). High levels of P accumulation in soils are at least partly responsible for this increased transfer. The processes and pathways involved in the transfer of diffuse P from soil to surface and subsurface waters have been described in a number of recent reviews (e.g. Haygarth & Jarvis, 1999) and involve both soil solution P and mineral bound P (McDowell *et al*., 2001).

Relationships between P transport in overland or subsurface flow and soil P levels have been investigated extensively, prompted by a desire to establish management thresholds to assist in mitigation of P transport from agricultural landscapes (Sims *et al*., 2000). Several studies have shown markedly increased rates of P transfer when plant available P (e.g. Olsen P) exceeds a critical level (e.g. McDowell *et al*., 2001; Maquire & Sims, 2002). This critical level is often termed the 'change point' and is closely related to the degree of soil P saturation. Change point data have been used to assess risks of P transfer in subsurface and overland flow (McDowell & Sharpley, 2001). Soil P status is only one of the factors contributing to losses from pastoral soils. Transport of P from the landscape requires conditions favourable to overland or subsurface flow. Slope and drainage strongly influence the pathways of transfer at a field scale. The latter is affected by soil type and texture, structural properties (e.g. macroporosity) influencing infiltration, and plant cover. Preferential flow through root channels and earthworm burrows may help to bypass sorption surfaces and thereby promote rapid transfer of P down the soil profile (Haygarth & Jarvis, 1999). The magnitude of losses are also affected by the size of rainfall events and amount and timing of fertiliser applications.

Despite the advantages of legumes for improving soil N status and forage quality, there are also a number of disadvantages. In comparison to grasses, legumes tend to compete less effectively for nutrients, grow more slowly and produce less dry matter. They also tend to be more susceptible to cold, wet conditions; leading to greater seasonal variation in their DM production. Because of these problems, intensification of pastoral farming has led to the sowing of all-grass pastures and an increased reliance on fertiliser N in place of legume fixed N. In New Zealand, for example, urea use increased from 50,000 t/yr in 1992 to 310,000 t/yr in 2002. Between 1996 and 2002, fertiliser N rates increased from 39 to 102 kg/ha on dairy farms as compared with an increase from 0.7 to 6 kg/ha on sheep/beef farms. For each kg of fertiliser N applied, biologically fixed N may decrease by 0.3 to 0.7 kg/ha (Saggar, 2004). It is well known that the use of N fertilisers reduces the clover content and N-fixation rates of grass-clover swards (Ledgard *et al*., 2001).

Soil pH tends to decline with time under improved pasture management. The build up of surface soil organic matter (SOM) and N fertility in legume-based pastures results in a gradual increase in cation exchange capacity and H^+ saturation through high cation uptake by N_2-fixing legumes. The decline in pH is a consequence of high rates of N-fixation, greater mineralisation of organic matter (nitrification in urine patches) and leaching of nutrients down the profile (Haynes & Williams 1993). Relatively small decreases in soil pH can lead to large reductions in pasture production because of a build-up of phytotoxic levels of Al and Mn. Soil acidity is a problem that can be readily managed by regular monitoring of soil pH and appropriate applications of lime. Earthworm populations tend to increase following additions of lime to acidic soils (Springett, 1985) which may lead to increased organic breakdown and faster cycling of N and P as discussed below. Care should be taken to avoid excessive liming, which can change the availability of trace elements (e.g. iron, manganese, copper, cobalt) and increase the risk of some soil-borne diseases.

Applications of sewage sludge to grasslands is increasing in many areas of the world (Towers & Horne, 1997). While sewage sludge can have significant agronomic benefits as a source of nutrients (N, P) and organic matter, it can also increase soil greenhouse gas (e.g. nitrous oxide - N_2O) emissions (Scott *et al*., 2000) and the eutrophication of water bodies (Towers & Horne, 1997). Although, the application of sewage sludge can contribute to OM accumulation and C sequestration, it is less effective in this regard than other land management practices (Smith *et al*., 2000), particularly in grassland systems. The adverse impacts of sewage sludge are most often associated with the inputs of heavy metals such as Cd, Zn, Cu, Pb and Ni (Towers & Horne, 1997). Heavy metals (principally Cd, Zn and Ni) enter the food chain through plant uptake, which tends to be greatest in acidic and coarse textured soils (Hooda *et al*., 1997). Many grassland soils have a relatively strong metal binding capacity at neutral to basic pH levels. Soil acidification could markedly reduce this binding capacity and, therefore, increase the risk that heavy metals will enter the food chain and pollute water. This information will be important for interpreting soil pH data to improve the management of soil quality on sewage sludge amended soils.

Soil organic matter and soil quality

SOM contributes to soil quality through its effects on a wide range of soil properties and processes that influence soil physical condition and fertility. Grassland soils can have substantially more SOM than comparable soils used for arable cropping because of continuous inputs of plant roots, senescent herbage and animal dung. SOM content represents the balance between inputs and decomposition of organic matter. Although nutrient (N, P) application can substantially increase the amount of organic matter returned to the soil in plant litter and dung, studies suggest that there is relatively little effect on SOM in the long term (Metherell, 2003).

The Winchmore trial in New Zealand has provided long-term data on the effects of superphosphate on SOM in an irrigated ryegrass-clover pasture used for sheep grazing. The trial, which consists of three rates of superphosphate (0, 188, and 376 kg/ha), was initiated in 1952 on an intensively cropped, degraded soil. Superphosphate increased herbage dry matter and also the proportion of clover in the pasture. Annual average DM production (1952-2002) was 4.8 t/ha in the control, increasing to 11.9 t/ha at the 376 kg/ha superphosphate rate (Table 1). SOM increased rapidly during the first 15 years following pasture establishment. Contrary to the common perception that SOM increases with increasing DM production (and thus increased organic matter inputs), recent measurements of soil C show no significant increase associated with fertiliser addition. A similar steady-state soil C level was achieved, regardless of whether or not superphosphate was used. Tate *et al*. (1997) also concluded that there was no evidence for a net accumulation of soil C (i.e. C sequestration) under long-term pasture management based on monitoring of soil C levels from a large number of New Zealand grassland sites over the last 50 years, despite large increases in fertiliser-derived production. Lack of a SOM response to this very large increase in DM production is consistent with suggestions that above-ground C inputs have less influence on soil C storage than root C. Recently, Rasse *et al*. (2004) estimated that the mean residence time of root C in soil is 2.4 times that of shoot C. Evidence from the Winchmore trial shows that plants growing in low P soil have larger root systems that turnover more slowly than those in P fertilised soil (Table 1). Where native (e.g. tussock) grasslands have been oversown by improved grass/clover mixes, some studies suggest that there may be an inverse relationship between pasture production (and associated differences in stocking rate) and soil C content (Metherell, 2003).

Table 1 Effects of long-term superphosphate rates on above and below ground production, and turnover and soil organic C levels (adapted from Metherell, 2003)

	Superphosphate rate (kg/ha per year)			LSD
	0	188	376	p=0.05
Herbage production (t/ha per year)	4.8	10.9	11.9	0.9
Root mass (t C/ha per year)	2.8	-	2.1	0.2
Root turnover time (years)	1.9	-	1.3	0.7
Root production (t C/ha per year)	1.7	-	1.7	Ns
Total soil C (t C/ha)	72.3	75.1	73.0	Ns

Clover requires relatively high soil P levels and fertilisation is important to stimulate clover growth and symbiotic fixation of N. Superphosphate can substantially increase the quantities of N cycling in legume-based pasture (Metherell, 2003) and some of this may be retained in SOM. Increases in total soil N following phosphate application have been reported by Haynes & Williams (1992) and Parfitt *et al*. (2005). However, annual increases in soil N under high *v.* low P fertility pasture may not be large (e.g. 19 kg/ha per annum; Lambert *et al*., 2000): once steady state is achieved, further retention of N is likely to be low. Changes in SOM quality in response to fertilisation include decreased ratios of C-to-N, C-to-organic P, and C-to-S (Haynes & Williams, 1992), and increased microbial biomass and mineralisable N (Parfitt *et al*., 2005). These changes should generally result in more rapid cycling of nutrients. Results from Parfitt *et al*. (2005) indicate that N mineralisation in clover-ryegrass pasture increases as soil P status (Olsen P test) increases and Haynes and Williams (1992) showed higher rates of S mineralisation in soil fertilised with superphosphate compared with unfertilised soil.

Biological indicators and impacts

Microorganisms and soil fauna contribute in many different ways to the chemical fertility of grassland soils. Their populations and activity are also affected by chemical inputs to soils. Soil fertility has been shown to influence both the size and turnover rate of biologically active pools of soil organic matter. There have been numerous reports of improvements in soil N status under high fertility legume-based pastures (Lamberts *et al*., 2000; Ledgard, 2001) with annual increments in total N commonly ranging from 30 to 80 kg N ha^{-1} (Peoples & Baldock, 2001). Most studies report that active pools of organic matter show a positive to neutral response to fertiliser P (Tate *et al*., 1991; Sarathchandra *et al*., 1994; Ghani *et al*., 2003). Ghani *et al*. (2003) provided results from several fertility studies that showed increasing levels of microbial biomass C and N and hot water extractable C with increases in P fertility. This effect appears to be related to differences in above ground dry matter production along P fertility gradients.

In contrast to P, applications of N to established pastures often have adverse effects on active pools of organic matter (Okano *et al*., 1991; Ghani *et al*., 2003) that may be attributed to an increase in shoot-to-root ratios, a reduction in root biomass and increased turnover of biologically active organic matter pools. Ghani *et al*. (2003) showed that microbial biomass C and N, and hot water extractable C decreased with increases in N fertiliser applications from 0 to 400 kg/ha. Results from their monitoring of dairy (n=12) and sheep/beef (n=12) farms (Allophanic soils) showed that clover/grass pastures fertilised with P and grazed by sheep had significantly higher levels of microbial C, N and S, mineralisable N and hot water extractable C than intensively managed, N fertilised dairy pastures. Ghani *et al*. (2003) recommended hot water extractable C (HWC) as a useful indicator of soil quality based on its strong correlation to other commonly used measures of organic matter content/quality (i.e. microbial biomass C and N, mineralisable N and total carbohydrates) and its sensitivity to grassland fertiliser management practices. These authors also showed that there can be fairly significant inter-annual (seasonal) and intra-annual variation in these indicators. Understanding the effects of this variation on OM turnover and nutrient cycling is important in providing practical interpretations of this measurement.

European earthworms are now commonly found in temperate pastoral soils around the world. In New Zealand, their introduction to grazed pastures has been shown to lift pasture production by an average of 25-30% or the equivalent of about 2.5 stock units/ha (Stockdill, 1982). Earthworms perform a number of important functions in soils that influence plant nutrition and pasture production. They have been shown to increase the solubility of plant nutrients (Sharpley & Syers, 1976), accelerate mineralisation of organic N (Willems *et al*., 1996) and assist in the incorporation of surface-applied lime in pastures (Springett, 1983). On hill country pastures, reductions in runoff and soil erosion have been attributed to improvements in infiltration resulting from earthworm activity. However, other studies suggest that earthworms may lower the quality of runoff water (e.g. Sharpley & Syers, 1976). Their effects on soil aggregation and macropore formation can also contribute to increased bypass flow and greater leaching of surface-applied pesticides and nitrogen (Edwards *et al*., 1992). Because much of the existing data pertains to cropping soils, there is a need for more research on the benefits and impacts of earthworms on the chemical quality of grassland systems, in particular their impact on the accumulation and loss of SOM.

Nutrient cycling

The effects of livestock farming practices on nutrient cycling depend on whether the nutrients removed in forage are returned as dung and urine during grazing, as effluent, as fertilisers, or as a combination of these. The amount of nutrients returned to soil from livestock excreta differ widely between pastoral farming systems. Dairy cows, for example, excrete *ca* 60-99% of the nutrients that they ingest from forage and feed. Some nutrients (e.g. P, Ca, Mg, Cu, Zn etc.) are excreted predominantly in faeces while others (e.g. K) are excreted mainly in urine (Haynes & Williams, 1993). Significant amounts of N, Na, Cl, and S are usually found in both urine and faeces. A relatively high proportion of the ingested N (21%), P (35%) and Ca (22%), but a low proportion of the K (7%) is removed in meat and milk (Haynes & Williams, 1993). The urine patches of dairy cows can have very high loadings of N (1000 kg/ha), K (500 kg/ha) and other nutrients (e.g. Ca, Mg, S), often exceeding the amount that pasture can take up during a growing season (Williams *et al*., 1990; Haynes & Williams, 1993; Di & Cameron, 2002). Once mineralised, the surplus nutrients are susceptible to leaching (i.e. NO_3^-, K^+, Ca^{2+}, Mg^{2+}) where there is adequate drainage. Following the hydrolysis of urea to NH_4^+, nitrification and denitrification can contribute to large emissions losses of N as N_2 and N_2O.

Nitrogen fixation in legume-based pastures is regulated by soil inorganic N levels and pasture species composition. Increases in soil available N tend to lower the legume component of pastures and, as a consequence, decrease the N derived from fixation. This regulation of N_2 fixation acts to limit the N input from legumes in legume/grass pastures and thereby reduce the potential for N losses to the wider environment (Ledgard, 2001; Figure 1). Ledgard (2001) compiled data from several studies which showed that NO_3^- leaching losses are similar for grazed legume/grass and N-fertilised grass pastures at similar rates of N input. Furthermore, available data suggest that the proportion of total N lost (to volatilisation, denitrification and NO_3 leaching) to the wider environment tends to increase as the rate of N input increases.

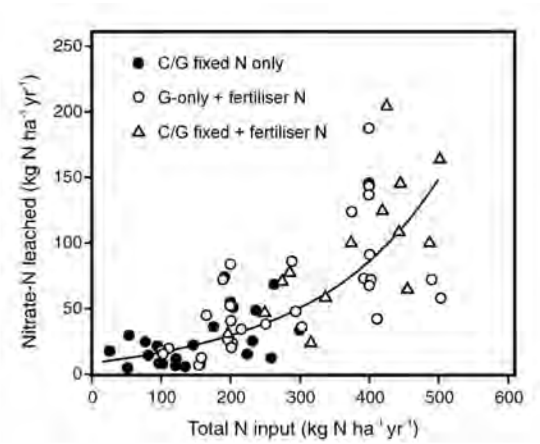

Figure 1 Nitrate leaching as a function of total N inputs from clover/grass (C/G) or grass only pastures, with or without N fertiliser additions (redrawn from Ledgard, 2001)

Nitrate leaching from grasslands is a major environmental concern in many countries (Jarvis *et al*., 1995). This is a particular concern under intensive land uses where there are high inputs of N in the form of fertilisers or effluents, or where N is returned in the urine of grazing animals. Goh & Williams (1999) compared N budgets for a wide range of temperate grazed pastoral systems. Their results showed that losses of N by leaching, volatilisation and denitrification from clover-based sheep/beef and dairy pastures were relatively low (7 - 74 kg/ha per year) and much less than the N removed in animal products. Nitrogen budgets for extensive, clover/grass sheep production systems in New Zealand and the UK indicate that leaching losses are very low (5-14 kg/ha per year) and much less than the total N inputs. By comparison, intensive N fertilised dairy and beef production systems in the Netherlands and the UK have reported N losses up to 390 kg/ha per year. In intensive systems, leaching can account for >50% of the total N losses and represent a high proportion of the total N inputs. In New Zealand, N_2O accounts for *ca* 20% of the country's greenhouse gas inventory, 50% of which has been attributed to animal excreta (de Klein *et al*., 2001). Recently, Di & Cameron (2002) showed that treating urine patches with a nitrification inhibitor (dicyandiamide [DCD]) can reduce nitrate leaching from these areas by 42-76%. Leaching of cations (i.e. K^+, Ca^{2+}, Mg^{2+}) that typically accompany NO_3^- was also reduced by 50-65% where DCD was applied (Di & Cameron, 2004).

Spatial variability in the chemical characteristics of grazed pastures can be very high. The location of shelter, water sources, gateways and farm roads can cause the establishment of fertility gradients resulting from uneven deposition of livestock dung and urine (Matthew *et al*., 1994). This effect can be particularly pronounced on hill country sheep farms where, for example, Ledgard (2001) estimated a net loss of 40 kg N/ha year in excreta from steep slope areas and a net gain of 180 kg N/ha year in stock camps on lower slopes. These nutrient gradients in turn can contribute to variation in the species composition and growth rate of pastures which can affect nutrient removal and N_2-fixation (Ledgard, 2001).

Physical effects of livestock treading

The detrimental effects of livestock treading on soil physical properties have been described in several studies (e.g. Drewry *et al*., 2004, Naeth *et al*., 1990). Treading can lead to localised, if not wide-spread, compaction of topsoil, especially under wet conditions. Compaction tends to reduce total soil porosity, especially the abundance of macropores. The decrease in macroporosity and associated physical properties has been linked to yield reduction in spring pastures (Drewry *et al*., 2004, Naeth *et al*., 1990). Compaction also retards water infiltration and gas diffusivity, leading to the development of anaerobic sites where N_2O production can be high. On average, N_2O emission from grazed pastures is more than twice that of mown pastures. Oenema *et al*. (1997) estimated that emissions from grazed pastures accounted for 0.2- 9.9% of the N excreted by grazing animals.

Intensification of livestock farming on high P fertility soils may increase the risk that damage from stock treading will increase surface runoff. Evidence from New Zealand hill country farms (steep land, slope >20 degrees) indicates that sediment losses are much greater from cattle-grazed (2740 kg/ha per year) than from sheep-grazed (1220 kg/ha per year) pastures (Lambert *et al*., 1985). In this study, the P fertility level (and associated differences in stocking rate) had no effect on sediment loss, possibly because there was less runoff from the high fertility pastures because of better surface cover. In contrast, McDowell *et al*. (2003) showed that increased losses of sediment and P were associated with reductions in macroporosity resulting from cattle treading. Additional research is needed to quantify the relationships between soil characteristics (e.g. texture, OM content etc), P fertility, grazing type and intensity, and sediment losses. Furthermore, little is known about effects of pastoral management practices on the transport of other sediment-bound compounds (e.g. Cd, DDT).

Intensification of livestock farming has also resulted in an increased reliance on supplementary feed and forage crops, particularly during periods of lower pasture production. In New Zealand and elsewhere, forage crops are often grown as a break-crop prior to resowing pastures. Stock treading on cultivated soils can cause severe compaction which may adversely affect subsequent crop production. Thomas *et al*. (2004) showed that using no-tillage practices to establish winter forage crops (ex pasture) can reduce soil compaction during grazing under wet conditions (≥ field capacity) and markedly improve the re-growth of multi-graze crops as compared with those established with minimum or intensive tillage practices. The more stable soil surface created by direct drilling crops into long-term pasture appears to lower the risk of surface compaction from heavy stock treading, even under very wet conditions. This reduction in compaction not only improved the re-growth of the forage crops but also greatly reduced N_2O emissions (Table 2: Thomas *et al*., 2004).

Table 2 Dry matter production and cumulative N_20 flux following simulated treading of crops established with three different tillage practices

Moisture content at treading[1]	Crop re-growth (kg DM/ha)[2]			Cumulative N_2O flux (kg N/ha)		
	IT [3]	MT	NT	IT	MT	NT
< FC	14.3	13.3	15.1	1.45	1.87	2.04
FC	10.9	15.1	14.6	5.73	3.18	3.00
> FC	7.1	9.5	17.8	14.86	12.68	4.97

[1]FC = Field capacity
[2]Triticale crop regrowth (4 months) after herbage removal in June 2003. DM = dry matter
[3]IT = intensive tillage; MT = minimum tillage; NT = no-tillage

Natural amelioration of soil physical properties in compacted grasslands occurs over periods ranging from several months to several years (Drewry *et al*., 2004). Knowledge of these recovery rates will be important to interpreting the effects of compaction on soil chemical properties and processes and devising monitoring schedules that reflect these trends.

Soil quality monitoring

The use of soil quality information to evaluate the sustainability of soil management systems has involved two approaches (Larson *et al*., 1994). With the *comparative assessment* approach, the condition of the soil from one system is compared with another using one-off measures of soil properties. This approach is often used to compare the effects of different land uses or management practices. With the *dynamic assessment* approach, the condition of soil under changing land use or management practices is assessed by monitoring soil properties over time. In agricultural systems, applications of the dynamic assessment approach usually involve both monitoring (i.e. repeated measurement) and control (i.e. regulation or management), and the focus shifts from describing soil conditions to maintaining soil conditions that sustain high productivity and minimise environmental impacts through improved management. To this end, soil quality monitoring may be coupled with best management practice recommendations to develop soil management decision support systems that improve the sustainability of production systems. The soil management decision support systems that incorporate soil quality monitoring have been described for mixed-cropping (arable/pastoral) (Beare *et al*., 1999), extensive sheep/beef (Shepherd, 2000) and dairy farming systems (de Klein *et al*., 2004). Most of these systems focus heavily on the physical aspects of soil quality and better integration with chemical fertility data is needed.

State-of-the-Environment (SOE) monitoring programmes in New Zealand and elsewhere have shown clear differences in the condition of soils under different grass-based land uses. Sparling & Schipper (2004) recently reported results from a three-year SOE monitoring programme involving a total of 511 sites, representing 11 different land uses, and covering 15 different soil orders. Not surprisingly perhaps, they showed that chemical fertility indicators including available P (i.e. Olsen P), mineralisable N, total C and total N differed among land uses in the order: dairy pastures > drystock (extensive sheep/beef) pastures > tussock grasslands. The higher levels of total C, total N and mineralisable N reflect an increase in SOM content under dairying. The higher Olsen P and mineralisable N values under dairying reflect the regular use of superphosphate fertilisers and resulting improvements in N-fixation.

Regular monitoring of soil fertility status is a key aspect of fertiliser management on many pastoral farms. Soil fertility analyses are usually carried out by private or government testing laboratories, which consequently hold large and potentially useful databases for assessing local, regional and even national trends in soil fertility. Where the economic optimum range and environmental thresholds of different fertility tests have been defined, information of this type can be useful to industry groups and regulatory authorities for sustainable land use planning and policy development. In a recent example from New Zealand, Wheeler *et al.* (2003) evaluated changes in the fertility status (pH, Olsen P, and exchangeable Ca, Mg, and K) of sheep/beef and dairy farms based on analysis of 246,000 commercial samples taken between 1988 and 2001. In terms of soil P status, their analyses showed a general rise in Olsen P values under both pastoral land uses that corresponded with increases in the recommended ranges for optimum economic production (Figure 2). Their data also showed that *ca* 30% of the dairy farms had Olsen P values below the target for maximum dairy production while *ca* 20% could lower fertiliser P inputs without loss of production. Approximately 30% of the sheep/beef farms had Olsen P values below the lower economic target. This coupled with the recent decline in median Olsen P values suggests that there is an opportunity to lift the production and profitability of this sector by increasing soil P levels.

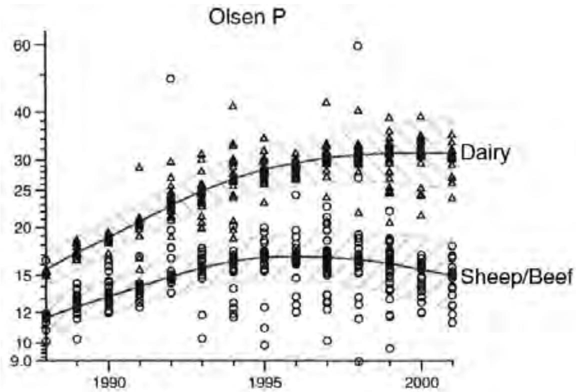

Figure 2 Trends in Olsen-P (log-transformed) in New Zealand sheep/beef and dairy pastures on sedimentary soils (redrawn from Wheeler *et al*., 2003)

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