Managing pasture for animals and soil carbon

A.J. PARSONS[1], J.S. ROWARTH[2] and P.C.D. NEWTON[1]
[1]*AgResearch Grasslands, Palmerston North, New Zealand*
[2]*Massey University, Palmerston North, New Zealand*
tony.parsons@agresearch.co.nz

Abstract

There has been growing interest in including soil carbon (C) sequestration, as an offset to greenhouse gas (GHG) emissions, within New Zealand's commitment to the Kyoto Protocol, even though national trends report soil C concentrations in many areas is declining. There are different schools of thought as to what drives changes in soil C (e.g. grazing management, fertiliser inputs, species) and so in our capacity to increase the rate of sequestration of C since 1990 to gain C credits. Difficulties in measuring *changes* in soil C with the confidence and resolution sufficient for reporting C sequestration rates is encouraging IPCC panels to look for industry 'rules of thumb' (e.g. devise C changes from regional stock numbers or fertiliser use trends). Prospects may differ substantially in areas of degraded soils with New Zealand's widespread already C–rich soils; interpretation *cf* trees could make these soils a major liability. Pasture managers, like policy advisers, face the complexities of the carbon cycle, uncertainty over the extent to which it can realistically be manipulated, and must recognise the difference between sequester*ing versus* maintaining sequester*ed* carbon, within likely Kyoto/ETS rules.

Introduction

New Zealand is in the unenviable position that a large proportion of its greenhouse gas (GHG) emissions are from agriculture. Whereas many developed nations can use 'smoke-stack-exhaust' technologies to reduce their commitments, under the Kyoto Protocol, New Zealand must somehow alter the biology of carbon (C) cycling in its major land-based industries. This is a major challenge.

The Kyoto Protocol is an agreement between some of the developed countries to address global warming by reducing (or offsetting) greenhouse gas (GHG) emissions. Countries which have signed up to the Protocol have accepted targets; New Zealand's is to reduce net GHG emissions to 1990 levels, and to accept responsibility financially (through purchase of C credits at global market price) if those targets are not met.

New Zealand's agricultural GHG emissions are predominantly methane (from ruminants) and nitrous oxide (from soils). These gases are referred to in carbon dioxide (CO_2) equivalents and, in short-hand, in units of C. New Zealand has already entered into schemes to offset emissions by sequestering C in 'Kyoto forests' (plantations established since 1990 on previously non-tree land). According to the current plan, agriculture could be brought into New Zealand's Emissions Trading Scheme (ETS) in 2013. This has resulted in great concern amongst farmers who see few ways of reducing or offsetting C emissions except through de-stocking of pastures – or planting more trees.

Emissions can be 'offset' under the Kyoto Protocol if a country can demonstrate it has increased its biological stocks of C. No credit is given to having stocks in a nominated baseline year (e.g. 1990); credit is given only to increasing stocks since the baseline year. Newly-planted trees are deemed to be sequestering C (building up stocks) for as long as the trees are increasing in size. In C accounting, all through this period they offer the prospect of receiving C credits. Once the trees are fully-grown, however, they reach a steady-state – a dynamic equilibrium where C in equals C out. In C accounting they are no longer a credit, for the C stock is no longer increasing. As long as the trees remain standing, a stock of C is simply being maintained, but this may be seen as a liability. If those trees fall over or are cut down, C debits must be re-paid. Replanting obviates the need for re-payment but is seen only as maintaining the C stock.

There are subtle differences between the Kyoto Protocol Article 3.3 and Article 3.4 (Appendix 1). Article 3.3 applies to a change in land use to trees, and effectively ignores the stock of C on that land before 1990; it focuses on the subsequent wood. Article 3.4 requires establishing what the C stock, or rate of sequestration of C, for an existing land use was in 1990, and showing that either this stock, or the rate of sequestration, has increased since then.

Recognising that pasture soil can also sequester C, some people are suggesting that soil-C should be included in the Kyoto Protocol and New Zealand's ETS, thus giving another way of offsetting liabilities. Could this be done and would it be beneficial to farmers?

If rules applied are similar to those for trees, farmers would not be rewarded for any stock of C as organic matter (OM) in soil that had been sequestered before 1990. They could be rewarded for periods of sequestering soil-C by changing their management system to build up soil-C. Under Article 3.4, however, to gain rewards they might have to prove that they are

sequestering C faster than they were before 1990.

Whilst discussing the complexities of interpreting the IPCC, Kyoto Protocol and New Zealand's ETS (see IPCC 2000; Baisden *et al.* 2001; Ministry of Agriculture and Forestry website), this paper considers the management of pastures to optimise production for both animals and soil-C, and essential directions for research.

What do we know of our pasture C stocks and how have they been changing?
Measuring the rate of change in soil-C at the temporal and spatial scales necessary to generate data for different farm systems and for a net national balance for a 1990 baseline and beyond, is extremely difficult. The current soil-C budget was not developed to measure changes over time nor is it capable of recognising the influence of management on soil-C stocks in improved grassland (Tate *et al.* 2003, 2005). It is based on extrapolation (observation and generalisation) from samples collected in the field at different locations under different land uses (Tate *et al.* 2005). Based on 796 samples beneath grazed pasture, and the national inventory of 1095 samples (for all land uses) to a depth of 300 mm, Tate *et al.* determined that 85% of soil-C is managed by pastoral farmers but the methodology makes the explicit assumption that soil-C is at 'steady-state' for pastoral land (Tate *et al.* 2005) and that management has no measurable impact on pastoral soil-C (Tate *et al.* 2003). Changes in soil-C stocks over time were therefore considered *a priori* to be zero. However, there is strong evidence that soil-C can be changed substantially by management (e.g. Thornley 1998; Jones & Donnelly 2004).

More recent re-analyses of national soil C-stocks data reveals that soil C has been changing (Schipper *et al.* 2007) – it has decreased on dairy pastures but increased on dry stock hill country pastures. Advanced techniques have attempted to resolve what components of C in soil are being lost/gained (Baisden *et al.* 2008), but in a recent multi-authored report to the Ministry of Agriculture and Forestry (Kirschbaum *et al.* 2009), soil scientists stated that neither the drivers, nor the timeframes, for these changes were known. Further research in the processes controlling, and factors contributing to, gains and losses was recommended.

How can we be so uncertain?
A major problem appears to be that many changes in pasture management alter several factors simultaneously. Hence if some factors alter soil C in opposite directions, the ensuing observations, even after the 10+ years it may take to get a measurable change in soil C, may give confusing indications (Johnston *et al.*

2009). For instance, few farmers would add fertiliser without increasing stock density. Different fertilisers (N or P) alone can alter plant species diversity and pasture composition, notably the presence of legumes. Adding fertiliser alters not only the amount of carbon flowing to soil (if it increases total plant growth), but also the proportion of carbon partitioned to roots; it also alters the 'quality' (e.g. the C:N ratio) of all the material cycling in the system. Although it is tempting to look at historic data to try and gain insights about what is happening, on-farm records, and even many experimental treatments, inevitably have confounded some of these factors (Johnston *et al.* 2009).

Another possible concern is different schools of thought about the relative role of the above ground components of pasture plants, and, for example, leaf litter, compared with root material alone, in the flows of C to soils and its sequestration there.

One school of thought emphasises the role of root material. Soil scientists (e.g. Rasse *et al.* 2005; Denef & Six 2006; Fröberg *et al.* 2007) suggest that root material is more likely than shoot litter to be stabilised and enter the soil-C pool because of the spatial location of root litter input. Root material is argued to have a longer residence time than shoot litter (itself not an issue dynamically) and, because of the action of growing roots, root litter is said to be more likely to be stabilised on aggregates (Denef & Six 2006). The upshot is a gradual transformation of root litter to soil organic C particularly in those soils in which metal-organic complexes are formed such as Andisols. In this way of thinking, contribution of shoot litter to soil-C pools appears to be minor: the bulk of shoot litter is presumed to be removed by decomposition directly to CO_2 in above ground respiration (and so doesn't feature in soil-C pools). Indeed, increased supplies of shoot litter may lead to an increase, or no change in soil C (Catovsky *et al.* 2002; Rasse *et al.* 2005; Skinner *et al.* 2006). This perspective minimises prospects for manipulations of the shoot growth (by fertiliser, water, management and cultivars) to make positive contributions to soil-C.

There is a second 'plant physiologists and ecologists' school of thought encapsulated in several major models (e.g. Hurley Pasture Model, Century, G'Day, EcoMod). This approach considers root and shoot litter as more equivalent with decomposition above and below ground determined by the chemical composition of the litter material. It assumes a substantial input of C to the soil from above ground growth and litter, possibly as much as 90% (Bardgett 2005). As a consequence any changes in above ground growth (e.g. management, global warming) are predicted to have a considerable impact on soil-C pools (e.g. Thornley & Cannell 1997; Cannell & Thornley 1998; Thornley 1998).

Is there any progress we can make from first principles?

Fig. 1 is a synthesis derived from decades of measurements of the photosynthesis, respiration, leaf turnover to litter, intake (and behaviour) of grazing animals, and the fate of C and N (and P) in animals. It describes how pasture management alters the C budget (values are presented as an annual total) above ground. The magnitude of C-flows this management would create are indicated by drawing an imaginary vertical line over one of the graphs: where the line cuts the curves on the graph, the Y-axis gives the size of the C flow. Unusually, but of importance for fundamental understanding, managements are described in terms of the average leaf area index ('LAI' , *cf* mean sward height, or mean standing biomass) that is being sustained (Parsons & Chapman 2000). Fig. 1 shows the size of the major flows of C through the pasture, into animals (per ha) over a grazing season, for both a low fertility and higher fertility case. The flows of C through grazed grassland are large, but of the C fixed in photosynthesis (being changed from CO_2 in the atmosphere into carbohydrate in the plant) nearly half is respired by shoots (returned as CO_2 to the atmosphere) and at least half of what remains dies un-harvested as grass leaves (and their below grazing height sheaths) turnover. It is the size of this turnover and flow of C potentially to soil that physiologists believe helps explain how even diminutive grass plants

in pastures can lead to the very substantial amount of C sequestered beneath grassland in soils, equivalent to that below tropical and temperate forests (Goudriaan 1992). Only 25% of the total flows (about half of the grass grown, net of respiration) is eaten by animals. (This must not be confused with 'pasture utilisation' figures of c. 80% that simply compare the amount eaten by animals to the amount that could have been cut from the same paddock on a given day.) The vertical bars on Fig. 1 give examples of the components that contribute to a supply of C to the soil: C partitioned to roots (for growth, root exudation, and ultimately root litter); surface litter, and approximately 25% of what the animal eats which is returned as organic matter in dung. The remainder of what is eaten is released as CO_2 in animal respiration (70%), and urine and methane (5%); little C is retained even in productive animals.

Increasing the intensity of utilisation (harvesting a greater proportion of what is grown, for example, by increasing stocking rate, and so maintaining a lower average pasture leaf area index and hence moving from left to right across each graph) predominantly reduces all the fluxes of C. It does increase intake per ha, but only up to a maximum, which occurs at what can be defined as an optimum mean vegetation state (a leaf area index of c. 2.0 in Fig.1a). Hence increasing stocking rate in general should decrease the flow of C to soil, and so reduce the potential for C sequestration.

Figure 1   The major flows of C (tonnes C/ha/year) through plants and animals in grazed pastures in relation to the intensity of grazing, as defined by the leaf area index sustained (*cf* annual mean vegetation cover) under (a) low fertility and (b) high fertility conditions. Vertical bars show examples of the potential total flow of C to soil (after Parsons & Chapman 2000).
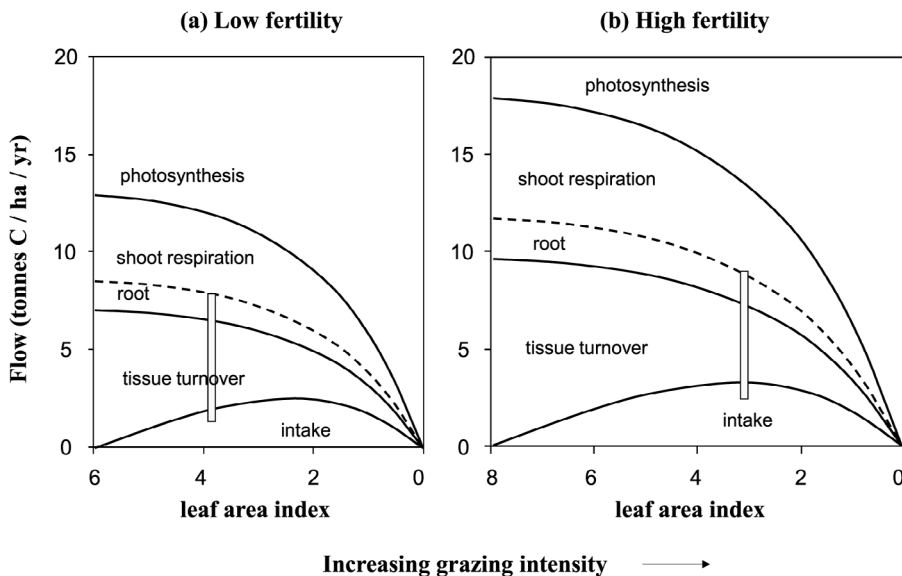
Fig. 1b illustrates how the same general principles may apply in a situation where there is faster plant growth, for the same vegetation state. This could be due to soil 'fertility', conditions which are warmer, brighter and moister, or a faster growing plant species. Fertility increases the flows of C at any given vegetation state (each gram of plant fixes more C). The curves in Fig.1b are higher on the Y-axis than they are in Fig 1a.

Factors that increase plant growth (such as fertility, higher residuals, and greater biomass) *increase* flows of C to soil and the potential for C sequestration; an increase in stocking rate *per se decreases* the potential for C sequestration. 'Intensification' of land use is a term that unfortunately confounds these two opposing trends. Farmers apply fertiliser and increase grazing intensity (usually by increasing animal numbers) at the same time. The size and position of the vertical bars in Fig. 1 indicates that the overall result can be very little change in fluxes of C to soil. Indeed, very different combinations of fertility and grazing intensity may give very similar flows of C.

What these graphs do reveal is that there may be scope for increasing the potential for soil C sequestration, without greatly reducing animal production per ha, and so the profitability of pasture farming. Although in general to increase the flow of C to soil a greater standing LAI or biomass must be maintained (e.g. greater residuals and/or longer re-growth periods), intake and performance per ha is relatively insensitive to pasture management around the optimum sward state (the intake curve in Fig.1a, for example, is relatively 'flat' around the optimum LAI of approximately 2). However, a small increase in pasture LAI can be seen to have a major effect in increasing the flow of C to soil (notably via the increased turnover of ungrazed litter). This can be seen from the way the 'gap' between the dotted line and the top of the intake curve widens rapidly.

Fig. 1 does not feature the dynamics of soil C, but details of the impact of many aspects of environment and management on the multitude of flows and the fate of C in soil associated with the principles described above have been produced. Hence the C sequestration consequences for a considerable number of climate and land use scenarios in different localities have been predicted (Thornley 1998).

Can we then extract some simple 'rules' of thumb?
The Kyoto Protocol Article 3.4 makes clear that, unlike the case for trees, changes in soil-C stocks must be related to the baseline stock (or rate of sequestration) in an identified year. While there are difficulties with obtaining a national estimate of soil-C stocks in 1990, as discussed earlier, it is even more difficult to assess the rate of sequestration on individual farms. Without these pieces of information, the evidence that a monitoring scheme would need to show (that soil-C had been increased by a landowner), is absent. It is easier to measure the girth of a growing pine tree trunk than it is to measure changes in soil-C.

Obtaining estimates of soil-C are difficult because it is spatially variable, and changes due to management are likely to be small in comparison with the background, making them difficult to detect (Parsons & Rowarth 2009). Furthermore, past measurements have usually been to 75 or 150 mm soil depth, whereas the likely requirement for the Kyoto Protocol C stock estimations is 0-300 mm. This raises two further problems. Soil-C can be increased by putting more C into a given soil horizon (raising the concentration), and/or by increasing the depth of the soil that contains C. Measuring only the top 300 mm will mean that no credit can be given for increasing soil-C at depth. Conversely, other ways of improving soil, such as turning the top 100 mm of compacted soil into a well-aerated layer (now fluffed up to say 200 mm) builds soil 'upwards'. Harvesting the top 100 mm of this layer could result in a false report of a decrease in soil-C stocks. (This should be apparent if the result was corrected for the decreased soil bulk density.)

Research groups in different countries have summarised the effects of management factors such as stocking rate, fertiliser input, and pasture species on C sequestration. Experimental evidence and first principles have been used more than observations of on-farm C. In North America, West *et al.* (2004), Ogle *et al.* (2004) and Conant & Paustian (2004) have proposed simple 'multipliers' to compare contrasting managements. These multipliers recognise that an increase in fertiliser input can increase C sequestration; that an 'improvement in plant species, notably the inclusion of legumes, will have an effect in increasing C sequestration; and that an increase in stocking rate (if inconsistent with the increased plant growth, and so an increase in grazing 'pressure') will reduce soil C sequestration. Confidence in these factors is relatively high because these researchers are operating in areas where pastures and rangeland are currently quite degraded in pasture species, and soils are low in C (Lal 2003). In contrast, New Zealand pastures frequently contain legumes, and grazing management has been carefully controlled. Soils are relatively high in C and simple 'correction' factors to score farms for likely soil C sequestration may be harder to devise. It may be even harder to convince the IPCC of the credibility of such factors. Work on devising such factors is, however, underway.

While there is global consistency in the concept that grazing pressure may limit soil C sequestration, whereas

the plants above ground have a role in increasing it, there is less certainty on the role of fertiliser.

Application of fertiliser (N or P) will increase the flux of C through a grassland ecosystem and potentially lead to the input of greater *amounts* of litter and a consequent increase in soil C (as implied in Fig. 1). However, the litter returned may be more decomposable and the production of labile organic compounds can increase resulting in mineralisation of old soil C. This 'priming' effect (Fontaine *et al.* 2007) is generally positively related to plant productivity and negatively related to root N concentration (de Deyn *et al.* 2008). Losses of soil C can result from priming although some dissolved organic-C simply moves to deeper layers in the profile and escapes further mineralisation (Steinbeiss *et al.* 2008a). Fertiliser can also increase the allocation of C to above ground material at the expense of investment in roots; where this results in a reduction in the amount of C, not just in the proportion, it has been argued that this reduces the potential for C sequestration – the assumption being that root C is more likely to be stabilised than leaf and shoot litter C. For the European situation (Soussana *et al.* 2004), soil C will increase if there is a reduction in the fertiliser used in intensive systems and, at the other extreme, an increase in the fertiliser applied to poor grasslands. Soussana *et al.* were referring to intensive systems where application of 400 kg N ha^{-1} are common; New Zealand systems rarely approach this level but re-cycling of N through animals produces patches of very high N concentration. Urine application can introduce a priming effect and losses of C can be expected (Clough *et al.* 2003).

Long-term New Zealand experiments provide little assistance in unravelling the fertiliser effect: in some there was an increase in soil C with fertiliser application but no change with different rates of fertiliser (Bolan *et al.* 1996; Sarathchandra *et al.* 1988; Nguyen & Goh 1990); in others there was no difference between fertilised and unfertilised (Saggar *et al.* 2001). A number of factors may explain these apparent differences, including the trial duration, the previous history of the site and the trial management. Few experiments isolate fertiliser as a factor because increased stocking rates invariably accompany increased fertiliser use and plant growth stimulation. A good description of the confusion caused by the misinterpretation of experimental data is given by Johnston *et al.* (2009); here they show how the conclusion of Khan *et al.* (2007), that N application resulted in a loss of soil C from permanent grassland, was more likely to be due to the change in farm system associated with the N application rather than a direct effect of the N itself.

Future Research
Pasture composition and plant traits
The role of plants in C sequestration needs further research to align the approaches of the soil scientist and plant physiologist. Also warranting further investigation is the increasing evidence that different functional types of plants (or simply different plant species traits) can have a profound affect on soil function (de Deyn *et al.* 2008). Grassland plants can differ widely in the quantity of litter and root exudations they produce, as well as the chemical composition of these products. Two strategies are commonly distinguished: slow-growing plants which produce nutrient-poor, recalcitrant litter, and fast-growing species of high nutrient content that produce easily-decomposable litter and which are likely to exude decomposable organic products (Personeni & Loiseau 2004; Personeni *et al.* 2005; de Deyn *et al.* 2008). The slow-growing plants may result in increased soil-C because recalcitrant C forms, particularly lignins and secondary compounds such as tannins, can retard decomposition. In addition, when lignin is broken down it produces humic substances that can form stable complexes with other organic substances. In contrast, sequestration under fast-growing species depends upon the quantity of litter produced outweighing the rapid decomposition of low C:N material. As well as the addition of new organic matter (OM) there is also the possibility that old OM can be decomposed in priming. As root exudation appears to be centrally involved in priming, species composition is implicated in soil-C sequestration. Of further interest is that ecosystems which are fungal-dominated tend to have higher OM (Six *et al.* 2006), and the fungi-bacteria balance can be changed by changing plant species because of specific plant traits (Bardgett *et al.* 1999). Legumes have been reported to increase the accumulation of soil-C in comparison with non-legumes, particularly when soil-N is low (Fornara & Tilman 2008). However, it may be that it is diversity *per se* that is important in increasing soil-C rather than the presence of specific functional groups (Steinbeiss *et al.* 2008b).

Understanding the relative contribution of legumes and grasses, and their above-and below-ground inputs, to soil-C under New Zealand conditions is clearly a research priority, at least in part because legumes have the capacity to self-regulate the balance of C and N cycles in grazed pastures (Schwinning & Parsons 1996 a, b).

Pragmatism and paradox
Making the wrong decisions could be costly. At a stocking rate of 3 dairy cows per ha, GHG emissions are 3.5 t C/ha per annum (based on total CO_2 equivalents of methane, nitrous oxide from urine, and nitrous oxide

from fertiliser use). This means that sequestering 3.5 t C/ha in soil would offset the emissions, but likewise a loss of 3.5 t C/ha would double the liability. If emissions trading is administered at sector level, across the 13 million ha of pastoral enterprise, the sequestration (or loss) of each 1 t soil C/ha per annum is worth a gain (or loss) of $260m (at $20 for 1 t C on the carbon market) to $1b (at $100 for 1 t C) in C credits on a national basis. Failure to offset emissions threatens pastoral industry profitability in less than 10 years (MAF 2008).

Our analysis so far suggests some scope to increase C sequestration rates by maintaining higher mean vegetation cover and that this might not greatly reduce animal production per ha. There are possibilities that more diverse species pastures, and more legume presence, could further aid C sequestration. However, the global demand for food, and the national desire for economic growth, is unlikely to be satisfied with de-intensification. The danger is that de-intensification would lead to a worsening of the ratio of GHG emissions per unit animal product - well fed animals have greater margin of production over maintenance.

One paradoxical solution (Trewavas 2001) is that it may be beneficial to *intensify* agriculture, to obtain greatest production efficiencies, but on a consequently smaller area of land. Areas of less productive land could consequently be de-intensified, and raise the possibility that C sequestration could become one source of revenue there. It is, however, likely that any intensification would be associated with increased nitrous oxide and methane emission; research is required to quantify the effect on GHG of different patterns of intensification of land use.

The detail required for effective political negotiations on such high financial stakes international stage is difficult to over-state. Research is required to underpin policy.

Conclusions

The intent of the International Panel on Climate Change (IPCC) is to encourage a change in behaviour, so that GHG emissions in future will be less than they would have been if 'business as usual' had continued. The IPCC accounting and interpretation is, however, as complicated as the biology affecting C cycling. Before soil-C is included in New Zealand's Kyoto agreement and ETS, research is urgently required. This must investigate the potential for C gain on soils which are already relatively high in C, as well as the mechanisms, processes and factors that create gains or losses. In keeping with other GHG mitigation options, New Zealand needs to create a process by which the effect of increasing fertiliser input or stocking rate on soil-C can be gauged. C-credits or debits could then be assessed based not on direct measurement (given the difficulties in achieving a reliable measurement, as discussed), but on theory/rules of thumb, based on sound principles acceptable to IPCC.

Until all this is done, including soil-C in the Kyoto calculations could mean that New Zealand farmers will be faced with a hefty liability. It could also leave the very farmers who have been leading the charge in 'C farming', and hence have soils with higher sequestered C than the norm, most at risk for payment.

REFERENCES

Baisden, W.T.; Beets, P.; Carran, R.A.; Clark, H.; Ford-Robertson, J.B.; Francis, G.S.; Maclaren, P.; Marshall, H.; Manning, M.R.; Newton, P.C.D.; Saggar, S.; Tate, K.R. 2001. An assessment of the significance to New Zealand of Article 3.4 activities under the Kyoto Protocol. Report prepared for Ministry of Agriculture and Forestry, 8 June 2001. 55 pp.

Baisden, W.T.; Parfitt, R.L.; Schipper, L.A.; Ross, C. 2008. Using Cs-137, C-14 and biomarker compounds to identify reasons for C and N losses in re-sampled profiles. Soils 2008: The Living Skin of Planet Earth Palmerston North 1-5th December.

Bardgett, R.D. 2005. The Biology of Soil: a community and ecosystem approach. Oxford University Press, Oxford, UK.

Bardgett, R.D.; Mawdsley, J.L.; Edwards, S.; Hobbs, P.J.; Rodwell, J.S.; Davies, W.J. 1999. Plant species and nitrogen effects on soil biological properties of temperate upland grasslands. *Functional Ecology 13*: 650-660.

Bolan, N.S.; Currie L.D.; Baskaran, S. 1996. Assessment of the influence of phosphate fertilizers on the microbial activity of pasture soils. *Biology and Fertility of Soils 21*: 284-292.

Cannell, M.G.R.; Thornley, J.H.M. 1998. N-poor ecosystems may respond more to elevated CO_2 than N-rich ones in the longer term. *Global Change Biology 4*: 431-442.

Catovsky, S.; Bradford, M.A.; Hector, A. 2002. Biodiversity and ecosystem productivity: implications for carbon storage. *Oikos 97*: 443-448.

Clough, T.J.; Sherlock, R.R.; Kelliher, F.M. 2003. Can liming mitigate N_2O fluxes from a urine-amended soil? *Australian Journal of Soil Research 41*: 439-457.

Conant, R.T.; Paustian, K. 2004. Grassland management activity data: current sources and future needs. *Environmental Management 33*: 467-473.

De Deyn, G.B.; Cornelissen, H.C.; Bardgett, R.D. 2008. Plant functional traits and soil carbon sequestration in contrasting biomes. *Ecology Letters 11*: 516-531.

Denef, K.; Six, J. 2006. Contributions of incorporated residue and living roots to aggregate-associated and

microbial carbon in two soils with different clay mineralogy. *European Journal of Soil Science 57*: 774-786.

Fontaine, S.; Barot, S.; Barre, P.; Bdioui, N.; Mary, B.; Rumpel, C. 2007. Stability of organic carbon in deep soil layers controlled by fresh carbon supply. *Nature 450*: 277-280.

Fornara, D.A.; Tilman, D. 2008. Plant functional composition influences rates of soil carbon and nitrogen accumulation. *Journal of Ecology 96*: 314-322.

Fröberg, M.; Jardine, P.M.; Hanson, P.J.; Swanston, C.W.; Todd, D.E.; Tarver, J.R.; Garten, Jr. C.T. 2007. Low dissolved organic carbon input from fresh litter to deep mineral soil. *Soil Science Society of America Journal 771*: 347-354.

Goudriaan, J. 1992. Biosphere structure, carbon sequestering potential and the ^{14}C carbon atmospheric record. *Journal of Experimental Botany 43*: 1111-1120.

IPCC 2000. IPCC Special Report. Summary for Policymakers. Land Use, Land Use Change and Forestry (LULUCF) 30 pp. http://www.ipcc.ch/pdf/special-reports/spm/srl-en.pdf

Johnston, A.E.; Poulton, P.R.; Coleman, K. 2009. Soil organic matter: Its importance in sustainable agriculture and carbon dioxide fluxes. *Advances in Agronomy 101*: 1-57.

Jones, M.B.; Donnelly, A. 2004. Carbon sequestration in temperate grassland ecosystems and the influence of management, climate and elevated CO_2. *New Phytologist 164*: 423-439.

Khan, S.A.; Mulvaney, R.L.; Ellsworthy, T.R.; Boast, C.W. 2007. The myth of nitrogen fertilisation for soil carbon sequestration. *Journal of Environmental Quality 36*: 1821-1832.

Kirschbaum, M.; Trotter, C.; Wakelin, S.; Baisden, W.T.; Curtin, D.; Dymond, J.; Ghani, A.; Jones, H.; Duerer, M.; Arnold, G.; Beets, P.; Davis, M.; Hedley, C.; Peltzer, D.; Ross, C.; Schipper, L.; Sutherland, A.; Wang, H.; Beare, M.; Clothier, B.; Mason, N.; Ward, M. 2009. Carbon stocks and changes in New Zealand's soils and forests, and implications of post-2012 accounting options for land-based emissions offsets and mitigation opportunities – including appendices. Report prepared for MAF, April 2009, 465 pp.

Lal, R. 2003. Carbon sequestration in dryland eco-systems. *Environmental Management 33(4)*: 528-544.

Ministry of Agriculture and Forestry 2009. Forestry sinks and the Kyoto Protocol (also describes article 3.4 interpretation for 'Additional Sink Activities'. http://www.maf.govt.nz/mafnet/rural-nz/sustainable-resource-use/climate/sinks/climate-04.htm Accessed May 2009.

MAF 2008. Projected Impacts of the New Zealand Emissions Trading scheme at the Farm Level. 16 pp.

Nguyen, M.L.; Goh, K.M. 1990. Accumulation of soil sulphur fractions in grazed pastures receiving long-term superphosphate applications. *New Zealand Journal of Agricultural Research 33*: 111-128.

Ogle, S.M.; Conant, R.T.; Paustian, K. 2004. Deriving grassland management factors for a carbon accounting method developed by the Intergovernmental Panel on Climate Change. *Environmental Management 33*: 474-484.

Parsons, A.J.; Chapman, D.F. 2000. The principles of pasture growth and utilisation. pp. 31-89. *In*: Grass. Ed. Hopkins A. Third Edition. British Grassland Society. Blackwells Scientific Publications.

Parsons, A.J.; Rowarth J.S. 2009. Soil carbon- rates, stocks, monitoring and cost. NZGA Newsletter: 2,5-6.

Personeni, E.; Loiseau, P. 2004. How does the nature of living and dead roots affect the residence time of carbon in the root litter continuum? *Plant and Soil 267*: 129-141.

Personeni, E.; Luscher, A; Loiseau, P. 2005. Rhizo-sphere activity, grass species and N availability effects on the soil C and N cycles. *Soil Biology and Biochemistry 37*: 819-827.

Rasse, D.P.; Rumpel, C.; Dignac, M.-F. 2005. Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. *Plant and Soil 269*: 341-356.

Saggar, S.; Tate, K.R.; Hedley, C.; Perrott, K.; Loganathan, P. 2001. Are soil carbon levels in our established pastures at or near steady state? *New Zealand Soil News 49*: 73-78.

Sarathchandra, S.U.; Perrott, K.W.; Boase, M.R.; Waller, J.E. 1988. Seasonal changes and effects of fertiliser on some chemical, biochemical and microbiological characteristics of high-producing pastoral soil. *Biology and Fertility of Soils 6*: 328-335.

Schipper, L.A.; Baisden, W.T.; Parfitt, R.L.; Ross, C.; Claydon, J.J.; Arnold, G. 2007. Large losses of soil C and N from soil profiles under pasture in New Zealand during the past 20 years. *Global Change Biology 13*: 1138-1144.

Schwinning, S.; Parsons, A.J. 1996a. Analysis of the coexistence mechanisms for grasses and legumes in grazing systems. *Journal of Ecology 84*: 799-813.

Schwinning, S.; Parsons, A.J. 1996b. A spatially explicit population model of stoloniferous N-fixing legumes in mixed pasture with grass. *Journal of Ecology 84*: 815-826.

Six, J.; Frey, S.D.; Thiet, R.K.; Batten, K.M. 2006. Bacterial and fungal contributions to carbon se-questration in agroecosystems. *Soil Science Society of America Journal 770*: 555-569.

Skinner, R.H.; Sanderson, M.A.; Tracy, B.F.; Dell, C.J.

2006. Above- and below-ground productivity and soil carbon dynamics of pasture mixtures. *Agronomy Journal 98*: 320-326.

Soussana, J-F.; Loiseau, P.; Vuichard, N.; Ceschia, E.; Balesdent, J.; Chevallier, T.; Arrouays, D. 2004. Carbon cycling and sequestration opportunities in temperate grasslands. *Soil Use and Management 20*: 219-230.

Steinbeiss, S.; Temperton, V.M.; Gleixner, G. 2008a. Mechanisms of short-term soil carbon storage in experimental grasslands. *Soil Biology and Bio-chemistry 40*: 2634-2642.

Steinbeiss. S.; Beβler, H.; Engels, C. 2008b. Plant diversity positively affects short-term soil carbon storage in experimental grasslands. *Global Change Biology 14*: 2937-2949.

Tate, K.R.; Scott, N.A.; Saggar, S.; Giltrap, D.J.; Baisden, W.T.; Newsome, P.F.; Trotter, C.M.; Wilde, R.H. 2003. Land-use change alters New Zealand's terrestrial carbon budget: uncertainties associated with estimates of soil carbon change between 1990-2000. *Tellus 55B*: 364-377.

Tate, K.R.; Wilde, R.H.; Giltrap, D.J.; Baisden, W.T.; Saggar, S.; Trustrum, N.A.; Scott, N.A.; Barton, J.P. 2005. Soil organic carbon stocks and flows in New Zealand: System development, measurement and modelling. *Canadian Journal of Soil Science 85*: 481-489.

Thornley, J.H.M. 1998. Grassland dynamics: An ecosystem simulation model. CAB International, Wallingford, UK. 271 pp.

Thornley, J.H.M.; Cannell, M.G.R. 1997. Temperate grassland responses to climate change: an analysis using the Hurley Pasture Model. *Annals of Botany 80*: 205-221.

Trewavas, A.J. 2001. The population/biodiversity paradox: agricultural efficiency to save wilderness. *Plant Physiology 125*: 174-179.

West, T.O.; Marland, G.; King, A.W.; Post, W.M.; Jain, A.K.; Andrasko, K. 2004. Carbon management response curves: estimates of temporal soil carbon dynamics. *Environmental Management 33*: 507-518.

Appendix 1

IPCC Articles
From Baisden *et al.* (2001).

The two articles of the Kyoto Protocol most relevant to this report are copied here with the key terms for consideration in this report underlined.

Article 3.3 of the Kyoto Protocol.
The net changes in greenhouse gas emissions from sources and removals by sinks resulting from <u>direct human-induced</u> land use change and forestry activities, limited to <u>afforestation, reforestation, and deforestation since 1990, measured as verifiable changes in stocks</u> in each commitment period shall be used to meet the commitments in this Article of each Party included in Annex I. The greenhouse gas emissions from sources and removals by sinks associated with those activities shall be reported in a transparent and verifiable manner and reviewed in accordance with Articles 7 and 8.

Article 3.4 of the Kyoto Protocol.
Prior to the first session of the Conference of the Parties serving as the meeting of the Parties to this Protocol, each Party included in Annex I shall <u>provide</u> for consideration by the Subsidiary Body for Scientific and Technological Advice <u>data to establish its level of C stocks in 1990</u> and to enable an estimate to be made of its changes in C stocks in subsequent years. The Conference of the Parties serving as the meeting of the Parties to this Protocol shall, at its first session or as soon as practicable thereafter, decide upon modalities, rules and guidelines as to how and <u>which additional human-induced activities related to changes in greenhouse gas emissions and removals in the agricultural soil and land use change and forestry</u> categories, shall be added to, or subtracted from, the assigned amount for Parties included in Annex I, taking into account uncertainties, transparency in reporting, verifiability, the methodological work of the Intergovernmental Panel on Climate Change, the advice provided by the Subsidiary Body for Scientific and Technological Advice in accordance with Article 5 and the decisions of the Conference of the Parties. Such a decision shall apply in the second and subsequent commitment periods. A Party may choose to apply such a decision on these additional human-induced activities for its first commitment period, provided that these activities have taken place since 1990.